Annex to document IPBES/4/INF/1/Rev.1

Thematic assessment on pollinators, pollination and food production (deliverable 3 (a)): Individual chapters and their executive summaries

Note by the secretariat

1. In section IV of its decision IPBES-2/5, the Plenary of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) approved the undertaking of a thematic assessment on pollinators, pollination and food production for consideration by the Plenary at its fourth session, as outlined in the scoping report set out in annex V to that decision.

2. In accordance with the decision, the thematic assessment was undertaken and a report on the assessment, comprising a summary for policymakers and a set of individual chapters and their executive summaries, was produced by an expert group according to the procedures for the preparation of the Platform's deliverables for consideration by the Plenary at its fourth session.

3. In section IV of decision IPBES-4/1, the Plenary approved the summary for policymakers as set out in annex II to that decision and accepted the individual chapters and their executive summaries (IPBES/4/INF/1), with the understanding that the chapters would be revised after the fourth session as document IPBES/4/INF/1/Rev.1 to reflect the summary for policymakers as approved**.**

4. A laid out version of the final thematic assessment on pollinators, pollination and food production (including a foreword, statements from key partners, a preface and acknowledgments, the summary for policymakers, the revised individual chapters and their executive summaries and annexes setting out a glossary and lists of acronyms, authors, review editors and expert reviewers) will be posted on the IPBES web site in time for IPBES-5.

5. This document contains the final set of chapters and their executive summaries. It corresponds to the Annex of IPBES/4/INF1/Rev.1.

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Chapter 1: Background to pollinators, pollination and food production

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Executive Summary

Most of the world's wild flowering plants (87.5%) are pollinated by insects and other animals (*established but incomplete*), more than three quarters of the leading types of global food crops can benefit, at least in part, from animal pollination (*well established*) and it is estimated that about one-third of global food volume produced similarly benefits from animal pollination (1.1). Pollination is an ecosystem function that is fundamental to plant reproduction, agricultural production and the maintenance of terrestrial biodiversity. Pollination is the movement of pollen within or between flowers (i.e., the transfer of pollen from an anther to a stigma) and is the precursor to sexual fertilization that results in the production of fruit and seed. Plants can be self-pollinated or pollinated by wind, water, or animal vectors. Self-pollination occurs when pollination happens within a single plant, sometimes with the aid of animal pollinators but it may also occur without a vector. Cross-pollination is the movement of pollen between different plants of the same species. Cross-pollination and self-pollination are not mutually exclusive; some plants have mixed pollination systems. Within these major pollination mechanisms there are many variations. Some plants can even produce seeds or fruits without pollination or sexual fertilization. The level of dependence of crops and wild flowers on pollination is highly variable (*established but incomplete*). Even within a single crop species, varieties may vary greatly in their dependence upon pollination. Of the leading global crop types (i.e. one or several similar crop species) that are directly consumed by humans and traded on the global market, 85% rely to varying degrees upon animal pollination, while 7% are not dependent on animal pollination and 8% remain of unknown dependence. In terms of global production volumes, 60 per cent of production comes from crops that do not depend on animal pollination (e.g., cereals and root crops), 35 per cent of production comes from crops that depend at least in part on animal pollination and 5 per cent have not been evaluated (*established but incomplete*). In the absence of animal pollination, crop production would decrease by more than 90 per cent in 12 per cent of the leading global crops, Moreover, 28 per cent of the crops would lose between 40 and 90 per cent of production, whereas the 45 per cent of the crops would lose between 1 and 40 per cent (*established but incomplete*). Of the world's wild flowering plants, 87.5% are pollinated by insects and other animals and most of the remainder use abiotic pollen vectors, mainly wind (*established but incomplete*). The complexities of plant-pollinator interactions, even in modern agricultural ecosystems, are poorly understood because usually more than one pollinator species is involved and they vary between seasons and locality (*established but incomplete*).

There are over 20,000 species of bees worldwide, they are the dominant pollinators in most ecosystems and nearly all bees are pollinators (*established but incomplete*). Flies are the second most frequent visitors to the majority of flowers with approximately 120,000 species. In addition, some butterflies, moths, wasps, beetles, thrips, birds and bats and vertebrates also pollinate plants, including crops (*established but incomplete*). Although managed honey bees such as the

western honey bee[1], *Apis mellifera,* and eastern honey bee, *Apis cerana*, are arguably the best known pollinators, other managed pollinators are important (2.4.2) and wild pollinators, for some crops, contribute more to global crop production than honey bees (*established but incomplete*) (1.3). Across 90 recent crop pollination studies conducted around the world, 785 bee species were identified as visitors to flowers of crop plants. Wild pollinators play a pivotal role in the pollination of wild plants (*well established*). Most animal pollinators are insects, of which bees are the best known. Flies outnumber bees in both diversity and abundance as pollinators in colder regions, such as at high altitudes and latitudes. Pollinating butterflies and moths are present worldwide, but are more abundant and diverse in the tropics. Beetles are important pollinators in many ecosystems and in some agricultural production, e.g., palm oil and Annonaceae (Custard apple family). Pollination by birds occurs mainly in warm (tropical/subtropical) regions, while pollination by bats is important in tropical forests and for some desert cacti. For a few plant species, less well known pollinators have been reported, including small mammals, lizards, cockroaches and snails. These less well known pollinators have small direct importance in food production (*established but incomplete*). At present, there is limited quantitative evaluation of the relative importance of the different flower visiting taxa that pollinate the world's flora (*established but incomplete*). Most pollinators are wild and a few pollinator species are managed (2.4.2). The western honey bee, *Apis mellifera*, is the most ubiquitous managed crop pollinator worldwide. *Apis cerana* is also managed for pollination in parts of Asia. Although most other pollinators are wild, there are other managed pollinators, including certain bumble bee and stingless bee species, and a few solitary bee and fly species, which also pollinate several crops. Managed pollinators may be introduced species, such as the western honey bee in the New World and the alfalfa leafcutter bee in North America. Wild pollinators of crops include bees (social and solitary), flies, butterflies, moths, wasps, beetles, thrips, birds, bats and other vertebrates (*established but incomplete*) and a few introduced species, such as the oil palm weevil (*Elaeidobius kamerunicus*), a West African species that was introduced into Malaysia. Wild insect pollinators are well known as important insect vectors to maximise pollination of certain crops (*well established*). Although the role of wild pollinators is becoming better understood and appreciated, the extent of their direct contributions across crops, fields and regions to food and fibre production remains poorly documented and experimental evidence is often lacking (*established but incomplete*).

High diversity (number of kinds) and abundance (size of populations) of pollinators in a single crop type can improve yields by maximizing the quantity and quality of the produce (*established but incomplete*) (1.4, 2.2, 2.3). Agricultural systems range from very high to low input practices. High-input agriculture (including inorganic fertilisers and pesticides) includes large fields dominated by monoculture and relatively few uncultivated areas. Low-input agriculture can be associated with polycultures, diversified crops, small fields and many uncultivated elements. Low-input agricultural

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[1] Also called the European honey bee, native to Africa, Europe and western Asia, but spread around the globe by beekeepers

practices that favour heterogeneity in landscapes and gardens and conserve natural vegetation are associated with greater flower visitation by wild pollinators (*established but incomplete*). Pollinator-dependent crop yields per unit area may be higher in low-input than high-input systems because pollinator abundance and species richness are generally higher where fields are smaller, pesticide use is limited and there is greater in-field density of pollinators (*established but incomplete*) (2.2, 2.3). Mixtures of different kinds of pollinators (including managed) have recently been shown to improve crop yields (quantity and quality) for various crops and regions of the world. A possible mechanism is via complementary pollination activities whereby species differ in their contribution to pollination. A high diversity of pollinators can result in high overall performance in crop production (*established but incomplete*) (1.4, 1.5, 1.6, 2.2).

Pollinator and pollination deficits resulting from globally prevalent drivers have been shown to cause reduced production locally, but these reductions are not reflected in global production statistics (*established but incomplete*) (1.1, 1.5, 5.0). Global analyses of food and fibre production indicate that more and more land is being placed into production (*well established*); for example, the total cultivated area increased almost 25% from 1961 to 2006 globally. In addition, more and more crops that depend completely or in part on animal pollination are being grown (*well established*). For example, the annual global crop production (measured in metric tons) attributed to pollinator-dependent crops increased by about 2-fold from 1961 to 2006 (Aizen et al., 2008) (*established but incomplete*). It is not understood why or how, in the global context, pollination deficits are presently not impacting global production when there is increasing documentation of local pollinator and pollination deficits coupled in some instances with economic loss (Aizen et al. 2008) (*inconclusive*) (3.8). Pollinators respond to several of the well-known drivers of environmental change that occur from local to global scales, namely climate change, land use change and management, chemicals (e.g. pesticides) and pollutants (e.g. heavy metals) in the environment, invasive alien species, parasites, and pathogens (*well established*) (2.1). A decline in diversity and/or abundance of pollinators could have cascading effects in biodiversity loss because many species of animals and micro-organisms depend on animal-pollinated plants for their survival (*established but incomplete*) (3.5). Pollinators contribute greatly to national and international economies because they are important for the production of food and fibre, including forage for livestock (*well established*) (4.2).

1.1 General introduction to this assessment

The scope of this assessment covers changes in animal pollination as a regulating service that underpins food production, and its contribution to gene flow and the restoration of ecosystems[2]. Thus, this document concerns food production that depends on pollination, and biodiversity related to plant-pollinator interactions. Seventy-five per cent of global food crop types benefit from animal pollination (Klein et al., 2007). The market price of additional crop production stemming from animal pollination services to agriculture was estimated to be $235-577 billion US$ in 2015 (Lautenbach et al., 2012), but this figure varies depending on market fluctuations, production volumes and the estimation methods used. The western honey bee, *Apis mellifera*, is a versatile and ubiquitous managed pollinator and the dominant visitor to more than half of the world's animal-pollinated crops (Klein et al., 2007; Kleijn et al., 2015). A few bumble bees (*Bombus* spp.) (Velthuis and van Doorn, 2006) and several solitary bees such as the alfalfa leafcutting bee (*Megachile rotundata*) (Bohart, 1962; Pitts-Singer and Cane, 2011) are also important managed pollinators. It is well known that managed pollinators suffer from a large number of serious problems, such as diseases, parasites and environmental stresses (2.4). Wild insect pollinators, which include native and introduced bees, flies, butterflies, moths and beetles, also contribute to the pollination of numerous leading global food crops (Klein et al., 2007; 2015; Garibaldi et al., 2013a). Many animal-pollinated crops provide vitamins and minerals (e.g. vitamin C, antioxidants, and lycopene) essential for good human and livestock health (Eilers et al., 2011; Chaplin-Kramer et al., 2014; Smith et al. 2015), even though some may comprise a small component of human diets.

Pollination is one of 15 ecosystem services identified as declining by the Millennium Ecosystem Assessment (2005). This is, in part, due to the growing demand for a diverse, nutritious diet (Klein et al., 2007; Eilers et al., 2011) and is resulting in more land being cultivated to satisfy global needs for food (Foley et al., 2011; Tilman et al., 2011). That, in turn, is increasing concern over security of food and other agricultural commodities (Gregory and George, 2011; Tilman et al., 2011; Breeze et al., 2014). Dependence upon crops that require pollination by animals is rising (Aizen et al., 2008). With the increase in agricultural intensification and cultivation, the demand for pollinators has grown, particularly in some developing countries (Aizen and Harder, 2009; Breeze et al., 2014) (see Chapters 3 and 4).

Of the world's wild flowering plants, it has been estimated that 87.5% are pollinated by insects and other animals and most of the remainder use abiotic pollen vectors, mainly wind (Ollerton et al., 2011). The level of dependence of crops and wild flowers on pollination is highly variable. Of the 107 leading global crop types, production from 91 (fruit, seed and nut) crops rely to varying degrees upon animal pollination. In terms of global production amounts, 60% does not depend on animal pollination (e.g. cereals and root crops), 35% does depend to some degree on pollinators and 5% have not been evaluated.

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[2] Annex V to IPBES decision-2/5

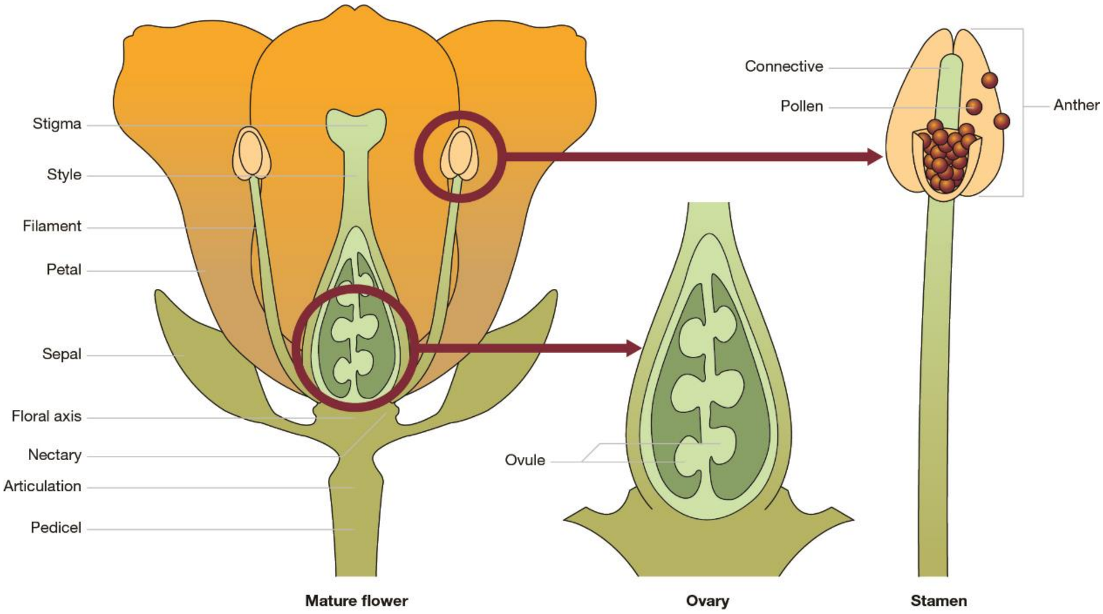
In the absence of animal pollination, crop production would decrease by more than 90 per cent in 12 per cent of the leading global crops, Moreover, 28 per cent of the crops would lose between 40 and 90 per cent of production, whereas the 45 per cent of the crops would lose between 1 and 40 per cent.

In view of growing demands for food and agricultural land, it is pertinent to recognize the interdependence between human needs and biodiversity conservation. This necessitates an assessment of the status of knowledge concerning pollinator population trends (Chapter 3). That would consider impacts of global change (Chapter 2), market and non-market values, and cultural use (see Chapters 4.1, 6.4, 6.6). Identification of the knowledge gaps (known risks and challenges) would help reduce uncertainty, facilitate decision-making and planning research to enable informed and appropriate management actions. Effective policy interventions would ensure that the social, cultural, environmental and economic values of pollinators are maintained.

1.2 Pollination and plant mating systems

1.2.1 What is pollination?

Pollination is an ecosystem process that is fundamental to the reproduction and persistence of flowering plants. Animal-mediated pollination is essential for about one-third of global food production (Klein et al., 2007). It occurs when animals move viable pollen grains from anthers (the male part of a flower) to receptive and compatible stigmas (the female part of a flower) of flowering plants and, when followed by fertilization, usually results in fruit and seed production (Figure 1, flower parts). Pollination may take place either between an anther and a stigma on the same flower, different flowers on the same plant individual (self-pollination), or between anthers and stigmas of different plants of the same species (cross-pollination) (Figures 1 and 2). Pollination is thus the main mechanism for sexual reproduction in flowering plants. As many plants do not self-pollinate or do so only to a certain degree to ensure seed production, most flowering plants depend on vectors for pollination, such as animal pollinators, wind, or water. As a precursor to fruit and seed production, pollination is crucial for the continued reproduction and evolution of flowering plants. Over 300,000 species (87.5%) of the world’s flowering plants have been estimated to be pollinated by animals (Ollerton et al., 2011).



1. Figure 1. A section of a flower showing the different parts
(modified from Pixabay Creative Commons Deed (2012).

Animals visit flowers to collect or consume rewards but do not visit them with the express purpose of pollination. These rewards include nectar (consumed by insects, bats, birds, non-flying mammals) as a source of sugar; pollen (used by most bees that collect it for provisioning their larval cells, and beetles, flies, birds, and some bats and non-flying mammals that eat it) for protein, vitamins, fatty acids and minerals; oils (collected by certain bees for provisioning their larval cells), fragrances (collected only by male euglossine bees (Apidae) for later attraction of receptive females) and resins (collected by various bees that use resin in nest construction). The mechanisms used by plants to ensure pollination are often complex, such as in wild figs. The books by Proctor et al. (1996) and Willmer (2011) describe and explain those pollination relationships.

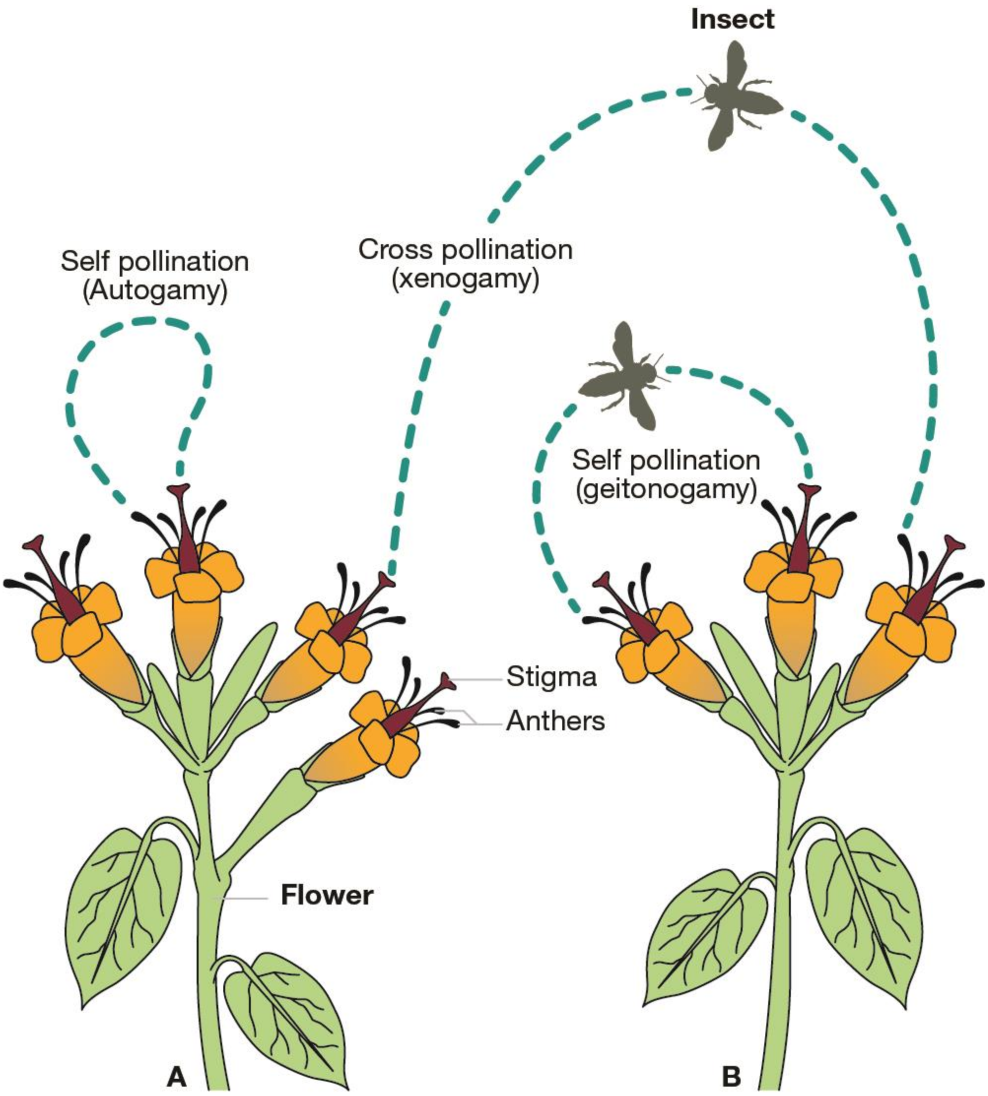
1.2.2 Plant mating and breeding systems

Pollination is a precursor to the sexual union of gametes. Following pollination pollen grains germinate on the stigma and the resulting pollen tubes grow through the tissues of the stigma to the ovule. The ovule develops into the seed and the ovary into the fruit. Even so, pollination alone does not assure sexual union (e.g. self-pollination on a plant that is self-infertile often occurs, but does not result in seed set). Pollination is crucial for reproduction, fruit and/or seed set in flowering plants whether they be crops, weeds or natural vegetation. Inadequate pollination may result from a shortage of viable pollen or limited pollinator activity.

Many plants have special mechanisms, some physiological and others morphological, that prevent or reduce the chances of self-pollination. An extreme strategy to avoid self-fertilization in plants is dioecy, the presence of male and female flowers on separate individuals. Some flowering plants that need to reproduce sexually cannot produce seeds without cross-pollination. Other plants readily self-pollinate and are self-fertile, and may rely on self-pollination and self-fertilization for seed production. Plant mating systems, as described and discussed in detail by Richards (1997) are defined in terms of a plant's form of reproduction (self-fertile to self-sterile) (Figure 2).

Plant breeding systems, of which there are many, explain the mechanisms that promote or dictate the particular mating system of a species of plant, or individual plant. They, like plant mating systems are an integral part of understanding pollination (see Proctor et al., 1996; Richards, 1997; Willmer, 2011).

There are four common mating systems that apply to plants that require pollinators for optimal fruit production. Obligate **xenogamy** (as in pome fruits, e.g. apples and pears) requires that the fruit/seed-bearing plant receives pollen from and is fertilized by pollen from an individual that is genetically different from the plant that receives the pollen. Self-pollination and self-fertilization can take place in two ways. In **autogamy**, pollen moves within the same flower whereas in **geitonogamy** the pollen moves between different flowers of the same plant. Pollen may move spontaneously or through pollinator activity. Facultative xenogamy, geitonogamy and autogamy together (**Mixed mating systems**) occur to various degrees in, and may differ among cultivars. The modern literature contains little information on the mating systems (and pollination requirements) that contribute to optimal yields for many important crop varieties. Even so, in several economically important crop cultivars capable of autogamy, such as sunflower (*Helianthus annuus*) (Carvalheiro et al., 2011), oil seed rape (*Brassica napus*) (Bartomeus et al., 2014), strawberries (*Fragaria vesca*) (Klatt et al., 2014), coffee (*Coffea* spp.) (Roubik, 2002) and soybean (*Glycine max*) (Milfont et al., 2013), significant yield boosts and improved quality have been documented when pollinators are involved (Garibaldi et al., 2013b).



2. Figure 2. Plant pollination systems that require and do not require pollinators for optimal crop production as represented by two plants A and B. The pollen vector is represented by a bee but could be any animal or abiotic pollinator (Drawn by Ian Smith, Guelph)

Some crops do not require a pollinator (such as in automatic self-pollination and subsequent self-fertilization (as described above)), and in **agamospermy** (apomixis) whereby flowers produce seed without the involvement of nuclei from pollen. Their embryo and endosperm are produced without fertilization (e.g. some cereals). **Parthenocarpy** occurs when flowers set seedless fruits without pollination or fertilization (e.g. banana, *Musa* spp., and various citrus cultivars). Some seedless melon (*Cucumis* sp.) cultivars, however, require pollination to initiate the hormonal influences needed for fruit production. **Stenospermy** happens if fertilization and some embryo growth are required for fruit production, e.g. in Brazilian seedless sugar apple (*Annona squamosa*) (Santos et al., 2014). In order to find out how much crop production of a focal crop species or variety can ideally be enhanced by optimal delivery of pollination, detailed studies of the crop mating system are required. Information is not available for many varieties and different varieties exhibit different degrees of the various types of mating systems (Garratt et al., 2014; Hudewenz et al., 2013).

1.3 The diversity of pollinators and their role in food production

Across the wider literature many species of flower visitors have been reported to visit blooming crop. For example, a meta-study including 90 recent worldwide crop pollination studies found that 785 bee species visit crop flowers (Kleijn et al., 2015). In most parts of the world bees are the most abundant and diverse pollinators, with over 20,000 species recorded around the globe (Neff and Simpson, 1993; Klein et al., 2007; Michener, 2007). Flies are also an important group in agriculture with approximately 120,000 species known to science, although only some families are effective pollinators (Larson et al., 2001). Flies outnumber bees in both diversity and abundance as pollinators in colder regions, such as high altitude/latitude habitats (Elberling and Olesen, 1999). Apart from bees and flies, some butterflies, moths, beetles, wasps, thrips, ants and vertebrates also pollinate plants, including some crops. Pollinating butterflies and moths are present worldwide, but are more abundant and diverse in the tropics (Scoble, 1995). Kevan and Baker (1983a) discuss the wide diversity of insect pollinators. Besides insects, some birds and bats are important pollinators (Proctor et al., 1996; Willmer, 2011). Bird pollinators occur mainly in warm (tropical/subtropical) regions, while bats are important pollinators in tropical forests and for some desert cacti. For a number of plant species less well known pollinators have also been reported. These include, among others, cockroaches (Nagamitsu and Inoue, 1997), mice (Wester et al., 2009), squirrels (Yumoto et al., 1999), lizards (Olesen and Valido, 2003; Hansen et al., 2006; Ortega-Olivencia et al., 2012) and snails (Sarma et al., 2007). The less well known pollinators are not known to have major roles in supporting agricultural production.

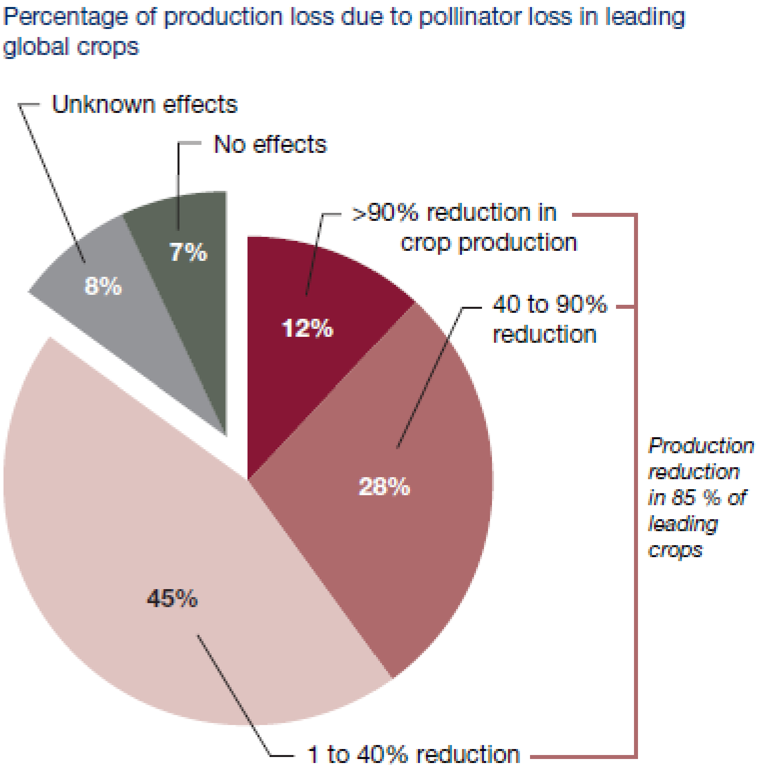
Most animal pollinators of agricultural importance are insects, of which bees, especially honey bees, bumble bees, stingless bees and solitary bees are the best known (Figure 4). The name honey bee refers to all bees in the genus *Apis*, of which two species are commonly managed: the western honey bee (*Apis mellifera*) and the eastern honey bee (*Apis cerana*) (Kevan, 1995; Kevan, 2007). Both those bee species have been managed for millennia in man-made hives and moved to follow nectar flows for honey production, or pollination (Crane, 1983; 1999). The western honey bee is native to Africa and Europe whereas eastern honey bees (*Apis* spp.) remain restricted to their native ranges. Both taxa have been moved by people around the globe (Moritz et al., 2005).

Modern beekeeping with honey bees arguably started with the invention of the top-opened movable frame hive designed by Langstroth in 1851 (Crane, 1999). This development allowed beekeepers to harvest honey and inspect colonies without destructively cutting out wax combs. Beekeepers could also inspect combs for disease and remove frames with bees and brood to start new colonies and thus increase hive numbers. With movable frames and standard-sized boxes for honey bees to occupy, beekeepers are able to trade honey bees, frames and boxes, and this type of beekeeping is now common. However, in many areas of the world (e.g. Africa and Asia), bees are still kept in simple boxes, straw skeps, hollow logs, walls of houses, bark tubes and clay pots, and entire honey combs are cut from these hives. One major advantage to this older, traditional method of beekeeping and honey harvesting is the low cost of inputs. Traditional beekeepers also report other advantages such as lower rates of colony migration (Joshi, 2000). Drawbacks include the destructive nature of comb harvesting and, when diseases are present, they are difficult to diagnose and treat. In modern, movable frame beekeeping the risk of disease spread is increased as combs and hive components are moved between colonies (Graham, 1992) (2.4).

Europeans introduced the western honey bee to the Americas soon after colonization and the species has since been imported to Asia, such that on every continent – Antarctica aside – beekeepers are practicing beekeeping with *A. mellifera* (2.4, 2.5). In all these cases, honey bees were originally managed mainly for honey and wax production. Management for pollination has subsequently grown and is now well developed in some intensively managed agricultural sectors worldwide (2.4). This reflects the fact that their overall value as pollinators far outweighs that of the honey harvested from them (Southwick and Southwick, 1992; Morse and Calderone, 2000; Kevan, 2007; National Research Council of the National Academies, 2007). *Apis mellifera* consequently remains the most abundant managed pollinator worldwide.

Bumble bees and stingless bees are also important pollinators for some high-value crops (2.4). The role of bumble bees, especially in tomato pollination, has led to the commercial production and international trade of over a million colonies per year worldwide (Velthuis and van Doorn, 2006). Bumble bee species are preferred to other managed pollinators because of their highly efficient "buzzing" behavior. They are effective in green houses and are easy to handle (Buchmann, 1985). The genus comprises around 250 species globally (Cameron et al., 2007), but commercial companies mainly breed two of them, *Bombus terrestris* in Europe and Asia, and *B. impatiens* in North America (see Chapter 3.3. for details of trends in managed numbers of colonies).

Figure 3 shows the percentage dependence on animal-mediated pollination of leading global crops that are directly consumed by humans and traded on the global market (Klein et al., 2007). Most crops are visited by more than one pollinator species. In the figure, bees are divided into honey bees, and other bees. Together they comprise over 50% of the organisms that pollinate the included crops.



3. Figure 3. Percentage dependence on animal-mediated pollination of leading global crops that are directly consumed by humans and traded on the global market.
Note that this graph and figures are taken from fig. 3 in Klein et al. (2007), and only include crops that produce fruits or seeds for direct human use as food (107 crops), but exclude crops for which seeds are only used for breeding or to grow vegetable parts for direct human use or for forage and crops known to be only wind-pollinated, passively self-pollinated or reproduced vegetatively.

The flowers of various plant species are visited and pollinated by arrays of flower-visiting animal species. Large animals such as birds, bats and other mammals frequently visit large flowers with copious and easily accessible nectar. For example, durian (*Durio zibethinus)* is a cultivated plant with large flowers pollinated by bats, birds and the giant honey bee, *Apis dorsata* (Lim and Luders, 1998). Cocoa, *Theobroma cacao*, on the other hand has small flowers, primarily pollinated by midges (Groeneveld et al., 2010).

Some plants need a specific pollinator and if moved outside their native range they either do not set seed or produce an inadequate crop (e.g., red clover in New Zealand and oil palm in South-East Asia) (Kevan and Phillips, 2001; Gemmill-Herren et al., 2007). Alternatively, exotic pollinators (i.e. those that are not in their native range), such as the western honey bee, pollinate many crops that are not from their home range. In the absence of an adequate pollinator, hand-pollination can be used. Human intervention through hand-pollination was used for oil palm pollination in South-East Asia for many years, but now the introduced oil palm weevil (*Elaeidobius kamerunicus*), native to tropical West Africa, is the main pollinator (Roubik, 1995; Gemmill-Herren et al., 2007). Pollination by hand has been practiced for millennia in the production of dates (*Phoenix dactylifera*) in the Middle East (Zaid and de Wet, 2002) and in the production of vanilla

(Arditti, 1992; Fouche and Coumans, 1992). Some farmers have turned to hand pollination in recent times to assure crop production, such as apple farmers in Maoxian County, China (Partap and Ya, 2012).

1.4 Defining different modes of agriculture

Different agricultural practices, from highly intensive greenhouse cultivation through annually cultivated field crops, to perennial cropping for fruit and nuts, to pastures and agroforestry all have different effects on pollinators, pollination and associated productivity (Kevan, 1999; 2001). Over the past half-century there has been both an expansion of agriculture around the world (Foley et al., 2005) and a change in agricultural strategies towards larger fields of monoculture crops that rely on high inputs of resources including synthetic fertilizers and pesticides (Pretty, 2008). This trend began with the Green Revolution and today is often referred to as *conventional agriculture*. However, many different kinds of agriculture still exist around the world that do not conform to this trend, and they have quite different effects on pollinators in particular (Kennedy *et al.* 2013), and biodiversity more broadly (Cunningham et al., 2013; Gonthier et al., 2014). These different kinds of agriculture include both many traditional farming approaches (Altieri, 2004; Koohafkan and Altieri, 2011) and others that integrate novel technologies or methods (Pickett et al., 2014). Because these alternative kinds of agriculture have different histories and origins, the meanings of the terms that different people use to classify them are complex and overlapping. Here we seek to define some of the terms in use in order to clarify how they are understood throughout the assessment, and to elucidate similarities and differences (Table 1).

Sustainable intensification was originally defined as increasing the yield output per unit of land while improving both environmental and social (livelihood) conditions (Pretty, 1997). It relied on sustainable agricultural practices, such as agroforestry, conservation agriculture, and conservation biological control, to establish low-input “resource-conserving systems” that (like agroecology, diversified farming systems and ecological intensification) are based on promoting favourable ecological interactions within the agro-ecosystem, rather than on purchased off-farm inputs. These approaches were found to improve yields and livelihoods in developing nations where they were widely practiced (Pretty et al., 2006). However, recent usage has shifted the focus toward capital- and input-intensive solutions to enhance resource use efficiencies, such as irrigation, precision agriculture, fertilizer application and GMOs (Parmentier, 2014), leading to critiques that the concept no longer promotes social equity (Loos et al., 2014).

Organic agriculture originated as a holistic system for building soil fertility, promoting water storage and the natural control of crop pests and diseases using management practices (FAO: http://www.fao.org/organicag/oa-faq/oa-faq1/en/, accessed 5 Aug 2015). Traditionally this farming strategy was associated with smaller-scale, low-input, diversified farms. A more recent development, *certified organic farming*, prohibits the use of almost all synthetic inputs of fertilizer and pesticides as well as genetically modified organisms, while allowing the use of organic fertilizers and pesticides. Certification allows marketing opportunities, which have been rapidly growing in Europe and North America. As the sales of certified organic products have increased in response to consumer demand, many organic farms today practice “input substitution” – in other words, similarly to conventional farms they are high-input, large-scale and highly simplified (low in crop and non-crop diversity), but use permitted organic products instead of synthetic fertilizers and pesticides (Kremen et al., 2012; Guthman, 2014). Thus today, organic agriculture includes a wide spectrum of farming styles, from those based on the original holistic concept, to those resembling conventional agriculture except for the choice of inputs.

Diversified farming describes farms that integrate the use of a mix of crops and/or animals in the production system, contrasting with the trend towards large areas of single crops in conventional agriculture. A *diversified farming system* is a newer concept (Kremen and Miles, 2012; Kremen et al., 2012) emphasizing use of a suite of farming practices that promote agro-biodiversity across scales (from within the farm to the surrounding landscape), leading to the generation and regeneration of key ecosystem functions (soil fertility, water use efficiency, pest and disease control, pollination, climate resilience, and others) and reducing the need for off-farm inputs. This concept is closely allied with concepts of *agroecology* and *ecological intensification* while emphasizing cross-scale diversification as the mechanism for sustainable production.

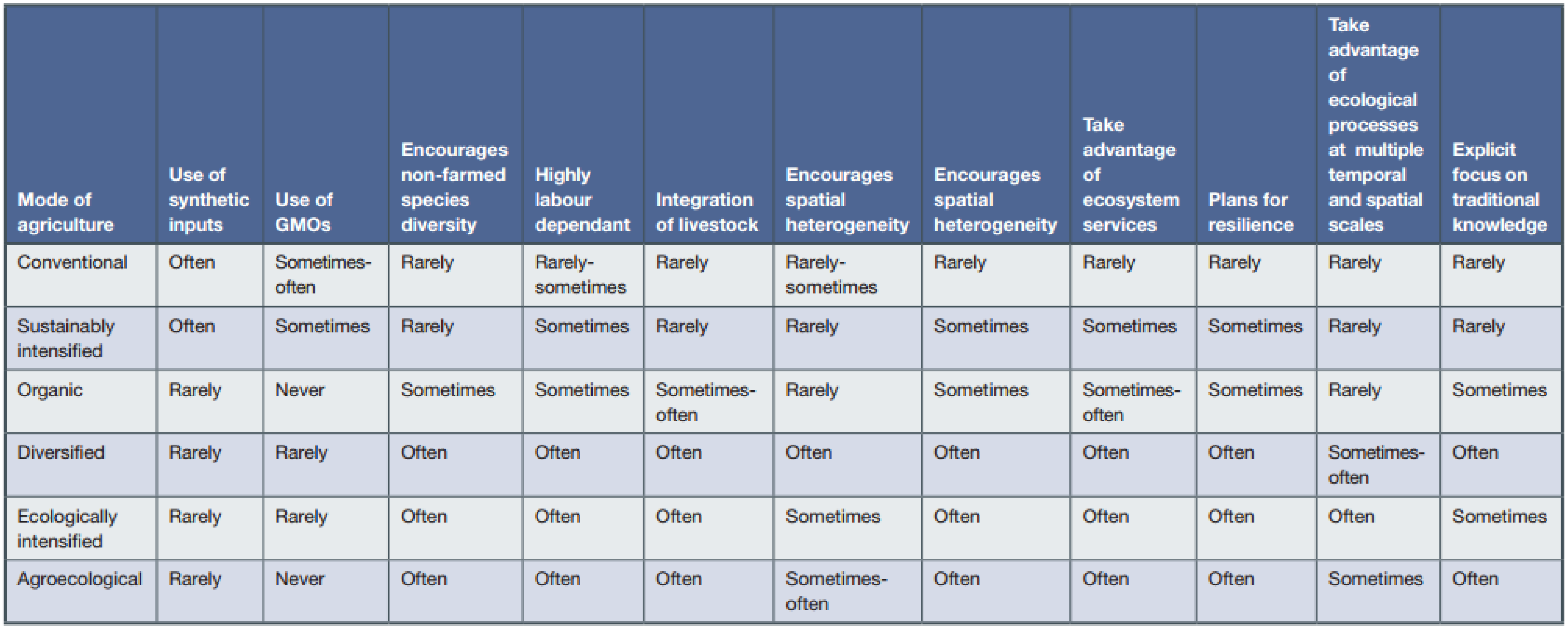
Ecological intensification describes a process rather than an end point. It provides one path toward intensified production for higher yield that would fit within the original broad sense of sustainable intensification. In contrast to current uses of the term sustainable intensification, ecological intensification emphasizes management that increases the intensity of ecological processes that support production, such as biotic pest regulation, nutrient cycling and pollination. In comparison with sustainable intensification, there is a more explicit focus on conserving and using functional biodiversity (Bommarco et al., 2013). The end point of ecological intensification is a farm that is likely to meet the definition of a *diversified farming system* (as defined above).

Agroecological agriculture is knowledge-, management- and labour-intensive rather than input-intensive, and aims to regenerate long-term agro-ecosystem properties (soil health, water storage, pest and disease resistance) by incorporating benefits of functional biodiversity (Tscharntke et al., 2012a), leading to sustainable, resilient systems (Altieri, 1999). Agroecological methods are often rooted in traditional farming practices and/or are co-developed by farmers and scientists working together (Altieri, 2004).

Different modes of agriculture that vary in management strategies will also differ in productivity, economic performance, labour requirements, and cultural values. An assessment of these differences is beyond the scope of this report but they are important to understand the risks and opportunities of adopting new strategies, independently of the values and ethical positions of different social actors.

1. Table 1: Similarities and differences among different term that define variations in the modes of agriculture

Each column identifies a characteristic, and scores qualitatively (often, sometimes, rarely, never) for the terms used in the assessment (rows). We include some characteristics (columns) that describe practice (the first six traits) and others that describe intention (the last five traits). There is still debate regarding definitions of different modes of agriculture, and within all kinds of agriculture there is a diversity of practice around the world. This Table reflects the definitions that we have adopted in this report, with frequency statements (i.e. never, rarely, sometimes, often) reflecting the most typical of the mode of agriculture in question.



Mode of agriculture	Use of synthetic inputs	Use of GMOs	Encourages non-farmed species diversity	Highly labour dependant	Integration of livestock	Encourages spatial heterogeneity	Encourages spatial heterogeneity	Take advantage of ecosystem services	Plans for resilience	Take advantage of ecological processes at multiple temporal and spatial scales	Explicit focus on traditional knowledge
Conventional	Often	Sometimes-often	Rarely	Rarely-sometimes	Rarely	Rarely-sometimes	Rarely	Rarely	Rarely	Rarely	Rarely
Sustainably intensified	Often	Sometimes	Rarely	Sometimes	Rarely	Rarely	Sometimes	Sometimes	Sometimes	Rarely	Rarely
Organic	Rarely	Never	Sometimes	Sometimes	Sometimes-often	Rarely	Sometimes	Sometimes-often	Sometimes	Rarely	Sometimes
Diversified	Rarely	Rarely	Often	Often	Often	Often	Often	Often	Often	Sometimes-often	Often
Ecologically intensified	Rarely	Rarely	Often	Often	Often	Sometimes	Often	Often	Often	Often	Sometimes
Agroecological	Rarely	Never	Often	Often	Often	Sometimes-often	Often	Often	Often	Sometimes	Often

1.5 The diversity of cropping systems

Conventional agriculture is characterized by large areas of monocultures, high rates of synthetic inputs such as fertilizers, insecticides, fungicides, herbicides and the use of heavy machinery (Hazell and Wood, 2008; Tilman et al., 2011; Tscharntke et al., 2012a). This form of agriculture produces large quantities of food, fiber and/or fuel per unit cropped area, but often at high social and environmental costs (Hazell and Wood, 2008; Godfray et al., 2010; Tscharntke et al., 2012a). Global analyses of food and fiber production indicate that more and more land is being placed into production. For example, the global area occupied by agricultural crops expanded by 23% from 1961 to 2006, with temporal trends differing greatly between the developed and developing world. The largest proportion of this increase can be attributed to pollinator-dependent crops. For example, pollinator-dependent crops contributed 13.7% to total agriculture production in the developing world in 1961, and this value increased to 22.6% by 2006 (Aizen et al., 2008).

Cropland has been expanding on most continents with an associated reduction in forests and grassland, and loss of habitat diversity (see Chapters 2 and 3). That, among other factors (see Chapter 2 and 3), has resulted in local declines in pollinator richness and abundance coupled with reduced flower visitation (Kevan, 2001; Kevan and Viana, 2003; Freitas et al., 2009; Partap, 2011; Garibaldi et al., 2011b; Clough et al., 2014; Rader et al., 2014). Such areas have impoverished pollinator faunas, if other solutions are not implemented, such as honey bee management, breeding of self-fertile varieties and hand pollination, crop failure may result.

Pollinator-friendly agricultural practices, such as management of set aside (fallow) areas, road edges and the establishment of insect/pollinator “hotels”, may be implemented. While conventional agriculture may increase food production, it may limit crop production over time due to the degradation of ecosystem services. Such reduced crop production is often compensated for by clearing new areas for crops, as is frequently seen in developing countries (Masuda and Goldsmith, 2009; Garibaldi et al., 2011a). Smallholder farmers may be more able to sustain the practices that favour pollinators for pollinator-dependent crops (Gemmill-Herren et al., 2014). However, insufficient pollination of pollinator-dependent crops results in poor yields or low quality fruit (Brittain et al., 2014; Klatt et al., 2014).

Agricultural systems range from monocultures to polycultures and other alternative forms of agriculture. Many polycultures, such as agro-forestry systems, are structurally closer to natural ecosystems than monocultures. In general, polycultures rely less on the use of fertilizers and pesticides than monocultures (Shackelford et al., 2013). Where environmentally-friendly farming practices, habitat heterogeneity, reduced pesticide use and the conservation of natural vegetation occur, flower visitation by wild bees and crop yield may be higher than in monocultures (Kremen et al., 2002; De Marco and Coelho, 2004; Kremen and Miles, 2012; Freitas et al., 2014). This is because bee abundance and species richness are generally higher on farms with more floral resources, such as organic farms where fields are smaller, inorganic fertilizer and pesticide use are limited and there is greater in-field density of pollinators and plants (Rundlöf et al., 2008; Holzschuh et al., 2010; Kennedy et al., 2013). Therefore, the more diverse a system, the more likely it is to host high pollinator diversity and greater harvest in pollinator-dependent crops (Tscharntke et al., 2012b; Tscharntke et al., 2015).

1.6 Pollinators, Indigenous and local knowledge and a good quality of life (see Chapter 5)

Throughout the world, local communities and indigenous people's knowledge systems about the functioning of complex ecosystems guide how they live and draw their livelihoods (Berkes, 2012). As a result, societies have developed unique biocultural associations with pollinators, both managed and wild, through diverse management, social and farming practices (Quezada-Euan et al., 2001; Stearman et al., 2008; Lyver et al., 2015). Local people, however, have also had a major destructive influence on biodiversity (Diamond, 2005) and hence on associated pollinators. Ostrom (1990) established that institutional arrangements that support common property systems of governance are critical determinants of whether or not sustainability results from local management systems.

Indigenous and local knowledge (ILK) therefore importantly includes knowledge of social institutions and governance systems as well as environmental observations, interpretations and practices (Berkes and Turner, 2006; Gómez-Baggethun et al., 2013). The contribution of ILK systems to pollination's role in ensuring nature's benefits to people and good quality of life is assessed in Chapter 5, guided by the following working definition (c.f. Berkes, 2012):

> *Indigenous and local knowledge systems (ILKS) are dynamic bodies of social-ecological knowledge, practice and belief, evolving by creative and adaptive processes, grounded in territory, intergenerational and cultural transmission, about the relationship and productive exchanges of living beings (including humans) with one another and with their environment. ILK is often an assemblage of different types of knowledge (written, oral, tacit, practical, and scientific) that is empirically tested, applied and validated by local communities.*

Understanding the interlinkages between pollinators and ILK-based management systems is important because substantial parts of the global terrestrial surface, including some of the highest-value biodiversity areas, are managed by ILK-holders (5.1). Pollinators in turn enrich livelihoods through additional income (e.g. beekeeping for honey production throughout the temperate and tropical world), food (e.g., honey hunting and gathering in Africa and Asia), medicine (e.g., human and veterinary remedies), ceremony and ritual (e.g., hummingbirds in Mesoamerica) and oral traditions (e.g., legends and songs in Oceania) (Buchmann and Nabhan, 1996; Silltoe, 1998; Nakashima and Roué, 2002; Mestre and Roussel, 2005). ILK is attuned to conditions of environmental change, for example through use of seasonal indicators to trigger crop-planting and honey-harvesting (Silva and Athayde, 2002; Berkes and Turner, 2006; Gómez-Baggethun et al., 2013; Césard and Heri, 2015) (5.2). In the Petalangan community in Indonesia, bees are managed to nest up to four times a year in the *sialang* trees through seasonal patterns of planting and harvesting, in accordance with flowering of corn, rice, and during the slash and burn period that opens the forest to start planting (Titinbk, 2013).

Modern science and indigenous knowledge can be mutually reinforcing (Tengö et al., 2014). For example, there are parallels between folk taxonomy of Abayanda indigenous people living around Bwindi National Park in Uganda, and modern systematics (Byarugaba, 2004).

By their practices of favoring heterogeneity in land-use as well as in their gardens, by tending to the conservation of nesting trees and flowering resources, by distinguishing the presence of a great range of wild bees and observing their habitat and food preferences, many indigenous peoples and local communities are contributing to maintaining an abundance and, even more importantly, a wide diversity in insect, bird and bat pollinators (Chapter 5).

1.7 Pollinator behaviour and interactions

Not all pollinators are equally efficient at servicing the pollination requirements of crops and wild flowers. Although honey bees, especially *Apis mellifera*, are the most frequently managed pollinators (Figure 4), other insect pollinators are more effective than the honeybee in some crops. For example, a common early-foraging sand bee, *Andrena cerasifolii*, and the blue orchard bee, *Osmia* sp., can pollinate some crops more effectively per flower visit than the western honey bee (Bosch and Kemp, 2001; Krunic and Stanisavljevic, 2006; Mader et al., 2010; Sheffield, 2014). The oil-collecting bee, *Centris tarsata*, is more effective than honey bees at pollinating cashew, *Anacardium occidentale,* in northeast Brazil (Freitas and Paxton, 1998). In New Zealand some flies, native bees and bumble bees are equally efficient pollinators of rape, *Brassica rapa*, as honey bees (Rader et al., 2009), but honey bees can be managed more easily. Pollinator behaviour can also be influenced by the presence of other pollinators, impacting fruit set through complementary activities (Garibaldi et al., 2013; Melendez et al., 2002, Pinkus-Rendon et al., 2005; Brittain et al., 2013b; 2006).

High diversity (number of kinds) and abundance (size of populations) of pollinators in a single crop type can improve crop yields by maximizing the quantity and quality of the produce. Pollinator behaviour under different conditions can result in variation in effectiveness across time and space. For example, wild pollinators were found to forage lower down on almond trees than managed honey bees, hence in conditions of high wind they were still able to provide pollination (Brittain et al., 2013a). Furthermore, in the absence of certain pollinator species the pollination of flowers at certain heights would be reduced, decreasing seed and fruit set (Hoehn et al., 2008). In strawberry, Chagnon et al. (1993), showed that large bees pollinate the pistils at the tip of the flower, whereas the smaller bees pollinate the pistils at the base of the flower leading to well-shaped fruit. These examples demonstrate that different pollinators can complement each other, often resulting in better pollination overall (Bluthgen and Klein, 2011; Brittain et al., 2013b). A global analysis of crop pollination data showed that wild pollinators play a central role in crop pollination, sometimes contributing more to fruit set than honey bees, even though they deposited fewer pollen grains on receptive stigmas than did honey bees (Garibaldi et al., 2013b). The mechanisms behind this finding are, however, not fully understood. Together these studies demonstrate that wild pollinators not only contribute to crop yield but, if they are sufficiently abundant, provide a degree of yield assurance to farmers growing insect-pollinated crops should honey bees falter.

Given that pollination is often not a simple association between plants and pollinators, consideration should be given to treating pollination as a complex web of interactions in any given ecosystem. Interactions include both different pollinating species interacting with a single crop during the same period, or one or more pollinators interacting with both crops and wild plants. Species that co-exist do not necessarily interact, and certain species interact more often with some than with others. These interactions can be investigated using ecological networks (Jordano, 1987; Bascompte and Jordano, 2007; Vázquez et al., 2009; Moreira et al., 2015).

A pollination network or web (most often and strictly speaking ‘visitation networks’) is a type of ecological network that contains information about which animals visits which flowers and how often (Memmott, 1999; Moreira, 2015) ((please see Figure 3.8, Chapter 3), and these associations may ultimately lead to pollination. Pollination networks allow visualization of the interactions among different species in a community. Such networks enable understanding of which species interact most often with others and whether they are specialists or generalists. Although the functionality of some pollination networks is resilient to the loss of species (at least up to a point where too many pollinators are lost from the system for it to function reliably), the efficiencies of pollinator species may differ, ultimately influencing plant survival and reproduction (Memmott et al., 2004). For example, removal of a single, dominant bee pollinator from subalpine meadows in Colorado permitted other species to become more general in their foraging. While the remaining bee species visited more plant species, they transferred less pollen between individual plants of the same species, resulting in lower seed set (Brosi and Briggs, 2013).

1.8 Local, landscape and global impacts upon pollinators

Modern ecosystem approaches to pollination are now examining the complexities of how pollinators and other flower visitors interact with each other on particular plants in both wild and managed ecosystems. Wild pollinator populations and their diversities wax and wane, as do abundances and diversities of flowers. The consequences of seasonal and annual variations can be offset in terms of the ecosystem function of pollination by various pollinators and flower visitors, and flowering plants, assuming each other’s roles under changing circumstances. Such complex dynamics play out differently within sites, across landscapes, habitats and ecosystems, as well as in their evolutionary consequences (Kevan and Baker, 1983b; Roulston and Goodell, 2011).

Understanding how individual pollinators that can actively move large distances and that have diverse life histories respond to global change drivers across different temporal and spatial scales remains a major challenge in food production (Roulston and Goodell, 2011; Tscharntke et al., 2012b). At the local scale, pollinator abundance and diversity are positively influenced by the diversity or proximity to non-crop floral resources and areas of low-intensity management methods (see Table 1) (Carvalheiro et al., 2011; Kennedy et al., 2013; Shackelford et al., 2013). Furthermore, land management practices with high inputs (e.g. pesticides) are often associated with local declines in diversity and abundance of pollinator populations (discussed further in Chapters 2.2 and 3.3). Declines in traditional beekeeping practices may also alter the biodiversity of pollinators at the local scale, with global reductions in the practice of stingless beekeeping impacting on local populations of these pollinators (see Chapter 5) (Cortopassi-Laurino et al., 2006).

At broader scales, pollinators respond to a number of global change drivers, including climate change, land use change and intensification, introduced species and pathogens (Cox-Foster et al., 2007; Tylianakis et al., 2008; Winfree et al., 2009; Potts et al., 2010; Burkle and Alarcón, 2011; Kennedy et al., 2013). Although these individual drivers have received some attention in relation to pollinators, studies addressing multiple drivers are few (Tylianakis et al., 2008; see Chapter 2.7; Schweiger et al., 2010; González-Varo et al., 2013; Vanbergen and The Insect Pollinators Initiative, 2013; Goulson et al., 2015). Pollinator populations are highly variable in time and space, therefore, it can be difficult to discern clearly trends in abundance as opposed to richness estimated from distribution records (Herrera, 1990; Petanidou et al., 2008; Rader et al., 2013a).

High pollinator diversity increases the chances that an effective pollinator is present and actively providing pollination at any given time and location. A diverse array of pollinators is therefore likely to buffer pollination against the effects of perturbations, such as land-use (Ricketts, 2004; Garibaldi et al., 2011b; Cariveau et al., 2013; Garibaldi et al., 2014) and climate change (Bartomeus et al., 2013; Rader et al., 2013b). This is because different pollinator species respond differently to changing conditions, due to their physiological, behavioral or other mechanisms (Petanidou et al., 2008; Winfree and Kremen, 2009). A long-term study of bees in the northeastern United States found that complementarity amongst bee species' periods of activity enabled synchrony between bee activity and peak apple flowering. This permitted a stable trend in pollination over time because various bee species displayed differential responses to climate change (Bartomeus et al., 2013). The effects of climate change on plant-pollinator interactions are still mostly unknown and the indirect effects upon interacting species and networks of species are poorly represented in the literature. However, one of the three key recommendations of the IPCC report for agriculture, in terms of adaptation measures to climate change, is the maintenance of biodiversity (IPCC, 2014).

Climate change is anticipated to bring about changes in rainfall distribution, wind patterns, temperature, air pollution and occurrence of extreme weather events, among other environmental changes (IPCC, 2014; Yuan et al., 2014). These changes may affect crop pollinators via changes in their spatial distribution, physiology and/or seasonal phenology through spatial and temporal mismatches between plants and their pollinators (Schweiger et al., 2008; Hegland et al., 2009; and see Chapter 2). Land use change, including intensification and extensification, is sometimes associated with local or regional declines in pollinator diversity, abundance and altered foraging behaviour (Westphal et al., 2003; Westphal et al., 2006; Kremen et al., 2007; Williams et al., 2012; Gonzalez-Varo et al., 2013; Kennedy et al., 2013; Woodcock et al., 2013; Rader et al., 2014). The landscape context can mediate these responses whereby local management factors may become important only in particular landscape contexts (Kleijn and van Langevelde, 2006; Rundlöf and Smith, 2006; Rundlöf et al., 2008). For example, pollinator richness and abundance can be high on organic farms in homogeneous landscapes, but not on organic farms in heterogeneous landscapes (Rundlöf and Smith, 2006). Landscape heterogeneity and less-intensive farm management methods are thus thought to mitigate pressures upon pollinators in some ecosystems (Kennedy et al., 2013). A strong relationship between bee diversity and heterogeneity of the urban landscape has also been found (Sattler et al., 2010).

As a consequence of global change (e.g. climate, land-use intensification and farming systems), pollinator communities may be changed in a non-random way, resulting in losses of particular functional guilds or species (Larsen et al., 2005; Flynn et al., 2009; Winfree et al., 2009; Williams et al., 2010; Rader et al., 2014). Individual taxa respond to land use change in different ways due to the varied morphological and behavioural characteristics within pollinator communities (Steffan-Dewenter, 2002; Tylianakis et al., 2005; Winfree et al., 2009; Shackelford et al., 2013). For example, social and solitary bees species may each respond differently to pesticide use (Williams et al., 2010) and dietary specialists and large-bodied taxa tend to be more strongly affected by habitat loss than less specialized and smaller-bodied taxa (Winfree et al., 2011a; Rader et al., 2014).

Different life history traits are associated with the quality and quantity of the pollination delivered. For example, body size measures correlate with pollination efficiency (Larsen et al., 2005; Vivarelli et al., 2011), foraging duration (Stone and Willmer, 1989; Stone, 1994) and foraging distance in some bees (Greenleaf et al., 2007). Frequent visitation may however also entail a cost (e.g., loss of pollen) to plants when pollinators are over abundant (Aizen et al., 2014). Within a given pollinator community, the variation in functional traits between species (i.e., functional diversity) itself improves the quality of pollination and reduces the variation in crop pollination and yield (Hoehn et al., 2008; Winfree and Kremen, 2009; Bluthgen and Klein, 2011).

1.9 The economics of pollination, risks and uncertainty (dealt with in more detail in Chapter 4)

The link between pollination and human quality of life is measured through the benefit that humans gain from this service. Due to the complexity of what a good quality life entails (Díaz et al., 2015), the benefit can have multiple dimensions depending on the type of contribution from pollination, such as the availability of basic foods or quality of food. This multidimensional benefit is called the value of pollination. However, values express a belief about a desired end, which guides action (de Vries and Petersen, 2009); this action can be individual or collective (Díaz et al., 2015). To date the emphasis among the literature has focused overwhelmingly on the economic value of pollinators, which may neglect the impact of changing pollinator populations other value dimensions.

Economic valuation of the conservation and sustainable use of pollination services can be highly informative for farmers and policy makers. Most early pollination valuation studies centered on managed western honey bees and farm gate prices of the crops they help produce. Valuation studies focused on pollination services typically used one of three major approaches (although more are detailed in Chapter 4, Section 4.4): Estimation of change to social welfare (Gallai et al., 2009); calculation of total market price of crop production that can be directly attributed to animal-mediated pollination (Gallai et al., 2009; Lautenbach et al., 2012); and replacement cost based on purchased inputs that substitute for natural pollination services (Allsopp et al., 2008; Winfree et al., 2011b). Most assessments have only examined the market price of additional crop productivity from crop pollination and have largely focused on national or regional analysis in the developed world (Chapter 4, Section 4.9).

1.10 Pollinators, traditional knowledge and a good quality of life

Ecological sciences and ethics together promote an opportunity to understand better the ways we can perceive and co-inhabit the world (Rozzi, 2013). Anthropocentrism with a utilitarian ethic is the dominant view in western societies, promoting globalization and neoliberal conditions for a dominant global development, with consequences for the environment and people (Cáceres, 2015). In this strategy the ethical position conceives the subject (human habits) separate from the environment (human actions disregarding the habitat). Here the economic growth, development and modernization that govern globalization neglect most people, biodiversity, ecosystems and humans with different cultures, which are disappearing from their native habitats and being excluded from the main discourses and laws that govern neoliberal global society. This dominant discourse determines a biotic, linguistic and cultural homogenization (biocultural homogenization, sensu Rozzi, 2013), which can be a ubiquitous driver for environmental change, biodiversity loss, and disruption of indigenous and traditional knowledge, promoting a small number of plant and animal species for nourishment.

Divergence from mainstream dominant utilitarian anthropocentrism (the other farthest view) is a biocultural ethical approach, defined as ethically connecting “human life with the diversity of beings, considered as co-inhabitants with whom humans co-constitute their identities and attain well-being” (Rozzi, 2013). It is not an extension of utilitarian ethics through the inclusion of animals, plants, etc. (intrinsic value assigned to pollinators), but includes interspecific relationships and how humans co-inhabit in the world. A biocultural perspective highlights planetary ecological and cultural heterogeneity, requiring an inter-cultural dialogue to solve environmental problems judiciously because it incorporates the views of marginalized people that should be respected and eventually adapted through intercultural exchanges (Rozzi, 2013). Biocultural ethics problematizes relationships among human conduct, habitats, and communities of co-inhabitants (Rozzi, 2013), embracing interrelatedness between different human groups and the environment. It includes different hierarchies of human values in decision making.

The consequences of decisions on biological and cultural diversity under different ethical approaches sharply differ because value and policy-making diverge. There are different environmental worldviews involving diverse ethics and hierarchical values that relate to ecological practices affecting pollination and pollinators. It is necessary to incorporate the diversity of worldviews, from indigenous and traditional knowledge on pollination and pollinators to sustainable ecological practices, into policy and education (see Chapter 5).

1.11 Legalities and governance of pollinators and pollination

Considering the importance of pollination for agricultural commodities, it is not surprising that there are a number of laws, directives, and decrees regulating various aspects related to pollination and the protection of pollinators. While many of these have been implemented for agricultural production, some policy instruments target the protection of natural or semi-natural ecosystems, due to their link with the provision of pollination. Much of the legislation is essentially designed to ensure the protection of pollinators against deleterious influences (e.g. pathogens, diseases, agrochemicals, habitat destruction, and in the case of managed pollinators, from inappropriate management practices; see Chapters 1.3, 2.3).

Many laws and regulations apply to an administrative unit within a country (e.g. federal state, province etc.). For instance in Canada, there is no legislation dealing with pollinators on a country level, but eight out of ten provinces have laws related to bees (Tang et al., 2007). In the United States, laws on hive inspection and disease treatment are likewise enforced at a state level (Michael, 1980). Supranational entities (e.g. the European Union) have also applied regulations to protect pollinators. Other regulations, for instance testing guidelines for agrochemicals, have international standards; however, adherence to them is only mandatory when stipulated in respective national legislation.

Regulations concerning managed pollinators are numerous. This is in particular true for the western honey bee, which is managed in colonies by people in many parts of the world. The honey bee has been domesticated by humans for thousands of years, and in contrast to most other pollinator species, it is a direct provider of honey and hive products like wax and propolis. The honey bee has been a subject of legislation for a long time before its role as a pollinator was appreciated. For example, early legal regulations of beekeeping practices trace back to the law of Solon in ancient Athens 594/593 B.C. (Crane, 1999). In modern agricultural systems the honey bee is known for its importance as a pollinator, hence the vast majority of existing regulations on pollinators are related to the honey bee and to beekeeping (See Chapter 6 for further details).

1.12 An overview of the report

Chapter 1 provides an introduction to the document by capturing its overall content. It presents an overview of existing knowledge and information on pollination, plant mating and breeding systems, diversity of pollinators and their contribution to crop production, global change drivers that directly or indirectly impact pollinators and pollination, market and non-market value of the contributions of pollinators and pollination, traditional and indigenous knowledge concerning pollinators, and institutional and policy mechanisms. The other chapters in this document deal, in greater detail, with specific aspects of the assessment introduced in this chapter.

Chapter 2 assesses the evidence for indirect and direct drivers of change in pollinators and pollination. It reviews in detail the impacts of direct drivers, focussing on land use change and management including GMOs, the use of toxic chemicals, environmental pollutants, climate change, the spread of invasive alien species, pests and pathogens, and interactions among these pressures. This chapter documents that pollination, especially by animals, is under threat as the world’s terrestrial ecosystems are changing at unprecedented rates.

In Chapter 3 the spatial and temporal status and trends in wild pollinators are reviewed. It deals with managed pollinators, including introduced and invasive pollinators, the structure of pollination networks, wild plant pollination, agricultural pollinator dependence, and the yields of animal-pollinated crops.

The economic and “non-marketed” values of pollinators are evaluated in Chapter 4, through the range of existing methodologies. In doing so, that Chapter identifies knowledge gaps and evaluates the assumptions, benefits, challenges and risks associated with each method of valuation and approach to economic analysis. Understanding how variations in pollinator population dynamics translate into monetary and other social benefits while identifying the costs incurred is a critical step forward in recognizing the spectrum of values that pollination services contribute to the agricultural sector and society at large. How the crops grown influence the benefits and costs of managed pollination services, and influence the availability and benefits of wild pollinators (ecological benefit: cost relations), are similarly important. Indeed, the values of wild pollinator services to agriculture are becoming increasingly recognized (see Chapter 4).

Chapter 5 includes indigenous and local knowledge perspectives on pollinators and pollinator systems and their benefits to those knowledge holders, as well as trade-offs between pollination processes and services and possible connections with disservices. Indigenous peoples and local communities' knowledge systems are based on different world-views (ontologies and epistemologies). Many indigenous peoples and local communities protect pollinators, directly and indirectly, through their own systems of governance, practices and understandings of the world. Their perceptions of pollinators and pollination are embedded in categories such as, fertility, reproduction and reciprocity.

Possible responses to reduce the risks and identify opportunities associated with pollinators and pollination are reviewed in Chapter 6. The responses are organised by policy sector, and grouped within each sector largely following the typology of action types developed for the Millennium Ecosystem Assessment (2005). The sectors are: 1) agricultural, agro-forestry and horticultural practices, 2) pesticides, pollutants and genetically modified organisms, 3) nature conservation, 4) pollinator management and beekeeping, and 5) urban and transport infrastructure. The action types are technical, legal, economic, social/behavioural and knowledge. Responses that apply across sectors are presented in a section on integrated responses. This chapter identifies those responses that are proposed, tested or established and summarises existing knowledge about whether or not each is an effective and appropriate response. The chapter also provides an overview of the tools and methods that have been used to understand and compare alternative responses and discusses what is known about trade-offs between them.

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Chapter 2: Drivers of change of pollinators, pollination networks and pollination

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Executive Summary

Indirect drivers (demographic, socio-economic, institutional, and technological) are producing environmental pressures (direct drivers) that alter pollinator biodiversity and pollination (*well established*). The growth in the global human population, economic wealth, globalized trade and commerce and technological developments (e.g. increased transport efficacy), has transformed the climate, land cover and management intensity, ecosystem nutrient balance, and biogeographical distribution of species (*well established*). This has had and continues to have consequences for pollinators and pollination worldwide (*well established*). International trade is an underlying driver of climate land-use change, species invasions and biodiversity loss (*well established*). The global expansion of industrialised agriculture driven by increased or changing consumption in the developed and emerging economies will continue to drive ecosystem changes in the developing world that are expected to affect pollinators and pollination (*established but incomplete*). The area of land devoted to growing pollinator-dependent crops has increased globally (*well established*) in response to market demands from a growing and increasingly wealthy population, albeit with regional variations (*well established*) (2.8).

Land use changes (including urbanization) that result in greater landscape fragmentation, lower connectivity, or the loss of resources for pollinators, will negatively affect wild pollinator diversity, abundance, and network structure (*well established*), potentially affecting community stability (*established but incomplete*). This land use change can also affect the potential for evolutionary adaptations of pollinator and plant species and their interactions (*established but incomplete*). Declines in plants and pollinators associated with land use are often only detected after a delay of several decades, but are linked to species traits governing the pollination interaction and sensitivity to environmental change (*well established*). Land use changes leading to losses in habitat diversity also reduce pollinator-dependent wild and cultivated seed and fruit set (*well established*) (2.2.1).

The creation or maintenance of more diverse agricultural landscapes may result in more diverse pollinator communities and enhanced crop and wild plant pollination (*established but incomplete*). Examples include use of intercropping, crop rotations (e.g., including pollinator forage crops), agroforestry, wild flower strips, and hedgerows. Local diversification and reduced intensity of land management will support pollinators and pollination, especially in landscapes dominated by large fields and conventional intensive management (*established but incomplete*). While some diversification methods may currently result in yield losses, these are counterbalanced by less inputs and the provision of further ecosystem services (*established but incomplete*) (2.2.2.1.1).

Intensive land management practices (such as high use of agrochemicals and intensively performed tillage, grazing or mowing) lead to a decline in pollinator richness at a local scale (*well established*). Monoculture systems with large, intensively-managed fields reduce both foraging (*well established*) and nesting (*established but incomplete*) resources for pollinators by removing weeds and reducing crop diversity and available nesting sites, such as suitable areas of soil (e.g., undisturbed), hollow stems of vegetation or dead wood. Certain mass-flowering crops provide huge food resources for some pollinators, but only for a short duration (*established but incomplete*) (2.2.2).

Extensively used traditional landscapes frequently contain high-quality habitats and species-rich pollinator communities (*well established*). These landscapes are often threatened by abandonment of farming (cessation of grazing or mowing of grasslands), which has been observed in temperate regions (*well established*) (2.2.2.2.1)**.**

The risk to pollinators from pesticides arises through a combination of toxicity (compounds vary in toxicity to different pollinator species) and the level of exposure (*well established*). Insecticides are toxic to insect pollinators and the direct lethal risk is increased, for example, if label information is insufficient or not respected, if application equipment is faulty or not fit-for-purpose, or the regulatory policy and risk assessment are deficient (*well established*). Pesticide application practices that reduce direct exposure reduce mortality accordingly (*well established*). Pollinators are likely to encounter combinations of pesticides applied in the field during foraging or flight (*well established*). These may result in unpredictable sometimes harmful effects; such combinations may interact in a complex and/or non-linear way (e.g., synergy) (*established but incomplete*). The level of exposure is significantly affected by factors including crop type, timing, rate and method of pesticide applications, as well as the ecological traits of managed and wild pollinators (*well established*) (2.3.1).

Pesticides, particularly insecticides, have been demonstrated to have a broad range of lethal and sublethal effects on pollinators in controlled experimental conditions *(well established)*. The few available field studies assessing effects of field-realistic exposure, provide conflicting evidence of effects based on the species studied and pesticide usage (*established but incomplete*). It is currently unresolved how sublethal effects of pesticide exposure recorded for individual insects affect colonies and populations of managed bees and wild pollinators, especially over the longer term. Most studies of sublethal impacts of insecticides on pollinators have tested a limited range of pesticides, recently focusing on neonicotinoids, and have been carried out using honey bees and bumble bees, with fewer studies on other insect pollinator taxa. Thus, significant gaps in our knowledge remain (*well established*) with potential implications for comprehensive risk assessment. Recent research focusing on neonicotinoid insecticides shows considerable evidence of lethal and sublethal effects on bees under controlled conditions (*well established*) and some evidence of impacts on the pollination they provide (*established but incomplete*). There is evidence from a recent study that shows impacts of neonicotinoids on wild pollinator survival and reproduction at actual field exposure (*established but incomplete*). Evidence, from this and other studies, of effects on managed honey bee colonies is conflicting (*unresolved*). What constitutes a field realistic exposure, as well as the potential synergistic and long term effects of pesticides (and their mixtures), remains unsettled (*unresolved*) (2.3.1.4).

Most genetically modified organisms (GMOs) used in agriculture carry traits for herbicide tolerance (HT) or insect resistance (IR). Though pollinators are considered non-target organisms of GMOs, there is potential for indirect and direct impacts (*well established*). Reduced weed populations are a likely result of the use of most herbicide-tolerant (HT) crops, diminishing food resources for pollinators (*established but incomplete*). The actual consequences for the abundance and diversity of pollinators foraging in herbicide-tolerant (HT)-crop fields are unknown (2.3.2.3.1). Insect-resistant (IR) crops result in the reduction of insecticide use, which varies regionally according to the prevalence of pests, and the emergence of secondary outbreaks of non-target pests or primary pest resistance (*well established*). If sustained, this reduction in insecticide use could reduce pressure on non-target insects (*established but incomplete*). No direct lethal effects of insect-resistant (IR) crops (e.g., producing *Bacillus thuringiensis* (*Bt*) toxins) on honey bees or other Hymenoptera have been reported, but some sub-lethal effects on honey bee behaviour. Lethal effects have been identified in some butterflies (*established but incomplete*), while data on other pollinator groups (e.g., hoverflies) are scarce (2.3.2.2).

Management of bees (honey bees, some species of bumble bees, solitary and stingless bees) is the basis for the provision of pollination for key parts of global food security, particularly for fruit and vegetable production (*well established*). Regional declines in managed colonies may be driven by socio-economic factors, e.g. low honey prices (*unresolved*). Mass breeding and large-scale transport of managed bees increases the risk of spread of pollinator diseases (*well established*). In the case of honey bees or bumble bees, these risks are well known for most regions (*well established*). The same risks may exist for intensively managed solitary and stingless bees (*inconclusive*), but as these species are generally managed on a smaller scale than honey bees, empirical evidence is still lacking. There are examples globally where the introduction of non-native managed bee species (e.g., honey bees, bumble bees) has resulted in escapes that subsequently led to competitive exclusion of native bee species (*established but incomplete*) (2.4.2, 2.5.4).

Insect pollinators suffer from a broad range of parasites, with *Varroa* mites attacking and transmitting viruses among honey bees being a notable example (*well established*). Emerging and re-emerging diseases (e.g. due to host shifts of both pathogens and parasites, sometimes arising from accidental transport by humans) are a significant threat to the health of honey bees (*well established*), bumble bees and solitary bees (*established but incomplete* for both groups) during the trade and management of commercial bees for pollination (2.4). These host shifts have been observed between different managed bees, between different wild pollinators, and from managed to wild pollinators and vice versa (*established but incomplete*). In managed social bees, disease outbreaks are often associated with colonies that are under stress (including poor nutrition, transportation, presence of other pests, pesticides, veterinary medicines, pollutants, and exposure or crowding (*established but incomplete*) (2.4.1).

The impact of invasive alien species on pollinators and pollination is highly contingent on the identity of the invader and the ecological and evolutionary context (*well established*). Alien plants or alien pollinators change native pollinator networks, but the effects on native species, diversity, or networks can be positive, negative or neutral depending on the species and ecosystem involved (2.5.1, 2.5.2, 2.5.5). Invasive alien predators affect pollination and plant fitness by consuming pollinators (*established but incomplete*). Invasive alien herbivores can affect pollinators and pollination, but this varies with the species and ecosystem concerned (*established but incomplete*). Alien plant pathogens are a potential but unquantified risk (*inconclusive*) (2.5.4). The impacts of invasive aliens are exacerbated or altered when they exist in combination with other threats such as disease, climate change and land-use change (*established but incomplete*) (2.5.6).

Several pollinator species have moved their ranges, altered their abundances and shifted their seasonal activities in response to observed climate change over recent decades (*well established*). These effects are expected to continue in response to forecasted climate change. The broad patterns of species and biome shifts toward the poles and higher altitudes in response to a warming climate have been observed over the last few decades in some well-studied species groups such as butterflies and bumble bees. A recent analysis has shown that bumble bees appear to be undergoing range contractions as climate changes across Europe and North America (*established but incomplete*). Climate change impacts on pollinators, pollination and agriculture may be manifested in the short-term (years) to longer-term (decades) depending on the pollinator species, but it is possible that the full impacts on nature and agriculture will not be apparent for many decades, due to long response times in and complexity of ecological systems (*well established*) (2.6.2.2).

Under all climate change scenarios for the second half of the 21st century, (i) pollinator community composition is expected to change as a result of decreases in the abundance of some species and increases in others (*well established*); and (ii) the seasonal activity of many species is predicted to change differentially, potentially disrupting life cycles and species interactions between plants and pollinators (*established but incomplete*). Changes in composition and seasonality are both projected to alter ecosystem function (*established but incomplete*). In high-altitude and high-latitude ecosystems, climate changes exceeding low-end scenarios (e.g. RCP 2.6)[3] are very likely to lead to major changes in species distributions and ecosystem function, especially in the second half of the 21st century (*well established*) (2.6.2.3).

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[3] low end scenarios are e.g., the Representative Concentration Pathway 2.6; http://sedac.ipcc-data.org/ddc/ar5_scenario_process/RCPs.html

The change in climatic conditions, especially under mid- and high-end scenarios, exceeds the maximum speed at which several groups of pollinators (e.g. many bumble bees or butterflies) can disperse or migrate (*well established*). Such species are predicted to find themselves in unfavorable climates and unable to reach areas of potentially suitable habitat (*established but incomplete*). To keep pace with shifting climates, species occupying extensive flat landscapes are particularly vulnerable because they must disperse over longer distances than species in mountainous regions (*well established*). Even if a species has the biological capacity to move fast enough to track suitable climates, those species with spatially restricted populations, such as those confined to small and isolated habitats or mountain tops, are expected to be particularly vulnerable to major climatic changes (*established but incomplete*). There is potential for differences in migration rate or ability to lead to a geographical or phenological dislocation of pollinator populations from populations of their historic food plants, which may present problems for pollination delivery (*established but incomplete*) (2.6.2.3).

Multiple pressures that individually impact the health, diversity and abundance of many pollinators across levels of biological organisation (from gene to biome scales), can combine in their effects and thereby increase the overall pressure on pollinators (*established but incomplete*). This variety of threats (often anthropogenic) to pollinators and pollination poses a potential risk to food security, human health and ecosystem function (*inconclusive*). The actual magnitude of interactions between these different pressures varies with location and among pollinator species, according to their biological attributes (*established but incomplete*). Nonetheless it is likely that changes in pollinator biodiversity and pollination are being exacerbated by both the individual and combined effects of multiple pressures (*established but incomplete*) (2.7).

2.1 Introduction

There are a number of potential drivers of changes in pollinators, pollination networks and pollination. In the present chapter, these drivers and their impacts are assessed, especially as they relate to the link of pollinators and pollination to food production, but also to semi-natural parts of the ecosystem. The pollinators under consideration here are mainly bees (honey bees, bumble bees, stingless bees and solitary bees), and to some extent other groups including syrphid flies, butterflies, moths, birds, mammals and reptiles.

The focus of the chapter is on the role of direct drivers of change in pollination, including the risks posed by:

(i) land-use and its changes (2.2), including changes in land cover and spatial configurations (2.2.1) and changes in land management and changing agricultural practices (2.2.2);

(ii) the use of chemicals, including fungicides, herbicides, and insecticides such as neonicotinoids (2.3.1);

(iii) the use of GMOs (2.3.2) and veterinary medicines (2.3.3);

(iv) environmental pollution from heavy metals, nitrogen and light (2.3.4);

(v) pollinator diseases (2.4.1);

(vi) pollinator management (2.4.2);
(vii) invasive alien species (2.5);
(viii) climate change (2.6); and
(ix) multiple additive or interacting threats (2.7).

It also includes assessments of the indirect effects of drivers of change (2.2.2.2.1: indirect effects of mowing; 2.2.2.2.3: indirect effects of fire; 2.3.1.2: indirect effects of pesticide applications; 2.3.2.3: indirect effects of GMO cultivation; 2.6.2.4: indirect effects of climate change; indirect effects also shown in Figure 2.2.1), including trade and policies in areas such as agriculture (2.8) and spatial planning (implicitly dealt with in section 2.2.1: “Changes in land cover and spatial configuration”). Possible responses and options to remediate effects of drivers, including tools or instruments are dealt with especially in Chapter 6, with specific discussions pertaining to scale (local, national, regional and global).

2.2 Land use and its changes

2.2.1 Changes in land cover and spatial configuration

Land cover has been defined by the UN FAO as the “observed (bio)physical cover on the earth's surface” (Di Gregorio and Jansen, 2005). Related to this concept is the idea of land use, namely “the arrangements, activities and inputs people undertake in a certain land cover type to produce, change or maintain it” (Di Gregorio and Jansen, 2005).

Human land use is the main current driver of changes in land cover (Foley et al., 2005), with the part of land exploited (see below) by humans being approximately 53% of the Earth’s terrestrial surface (Hooke and Martín-Duque, 2012; Goldewijk and Ramankutty, 2004). For instance, at a global scale, increased crop production has been generally associated with the replacement of forests (Goldewijk and Ramankutty, 2004), while it has been shown that grazing can lead to land degradation/desertification and scrub encroachment (Asner et al., 2004; but see also section 2.2.2.2). Logging is often followed by deforestation and conversion to crop- and grasslands (Haines-Young, 2009; Lambin et al., 2003). Urbanization generally involves conversion of agricultural land (Lambin et al., 2003). It is important to note that the type and speed of transition from one land cover type to another are dependent on the land management method (see 2.2.2), which has a cultural background and is thus influenced by local knowledge (see Uprety et al., 2012 for a discussion).

Since 1961, croplands have been expanding at the global scale and on most continents, with concomitant global reductions in forest and grasslands (http://faostat.fao.org/; a global annual average of 0.2% increase of croplands, accompanied by a reduction of 0.16% of forest land per year). This pattern was also revealed in modelled reconstructions of land cover using historical land use data for the last 300 years (Hansen et al., 2013; Hooke and Martín-Duque, 2012; Goldewijk and Ramankutty, 2004; Ramankutty and Foley, 1999). By

2030, most optimistic scenarios predict a net forest loss associated with a 10% increase in the area of agricultural land, mainly in the developing world (Haines-Young, 2009). Urban areas are also predicted to expand as a consequence of 66% (*vs.* 54% today) of the increasing global human population expected to be living in urban areas by 2050 (UN, 2014). Although forecasts suggest global increases, they are expected to be larger in developing countries, mainly in Asia and Africa (UN, 2014).

From an ecological perspective, changes in land cover involve shifts in the land cover composition and variations in its spatial configuration (e.g., fragmentation, isolation) (Fahrig et al., 2011; see Box 2.2.1), which directly affect the composition of biological communities and the relationships between pollinators and flowering plants (Ollerton et al., 2011; Vanbergen, 2014) (Figure 2.2.1). It is important to note that although the effect of changes in the composition and configuration of land covers on pollinators has been evaluated extensively, most studies focus on bees. Here, we present a review of how land cover modification through land use change can affect bee and non-bee pollinators and the pollination they provide.

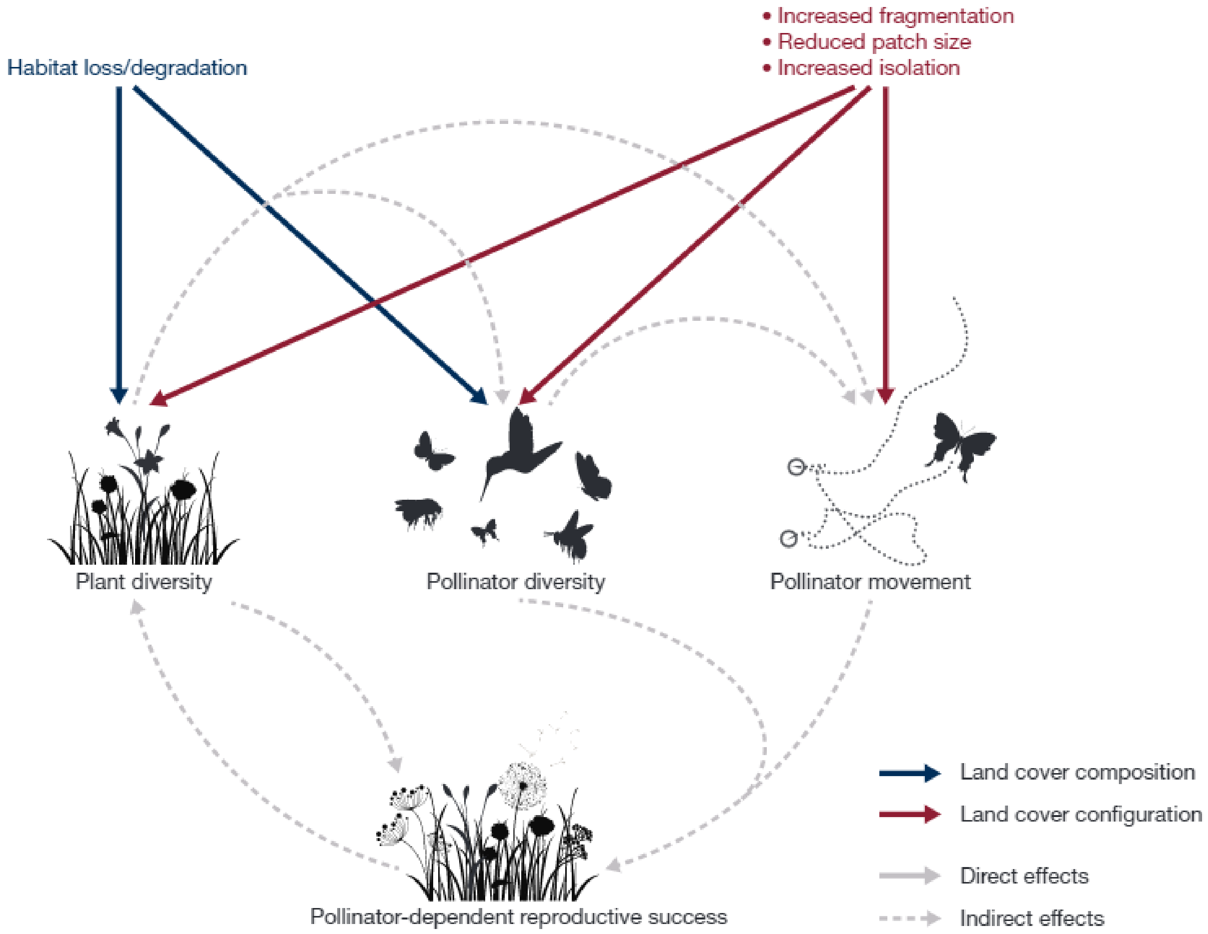


Figure 2.2.1 – Conceptual feedback loops between major components of pollination
and the effects of land cover composition (habitat loss or habitat degradation; black arrows) and configuration (measures of fragmentation, patch size, isolation; red arrows) on each component. These effects can be direct (solid arrows) or indirect (dashed grey arrows). Relative number of plant and pollinator types appearing in the figure does

not reflect their real proportion in nature. Modified from Hadley and Betts (2012). Refer to the text for directionality of these effects.

2.2.1.1 Changes in land cover composition

2.2.1.1.1 Habitat loss and degradation

Many types of land use (e.g., agriculture, urbanization) strongly change land cover types, leading to the disappearance of the habitats of many species, which is thus referred to as habitat loss (Fischer and Lindenmayer, 2007). Beyond habitat loss, land use change can induce a deterioration in habitat quality, termed habitat degradation. In these cases, species are still able to survive, but their populations may decline (Fischer and Lindenmayer, 2007).

An important body of research has investigated the effect of habitat loss and degradation on pollinators and pollination. Although the identified patterns appear to be consistent, they are incompletely documented in regions other than Europe and North America. It is well established that habitat destruction can reduce the population sizes, composition and species richness of pollinator communities (Hadley and Betts, 2012; Kennedy et al., 2013; Steffan-Dewenter and Westphal, 2008; Winfree et al., 2011; Figure 2.2.2; Table 2.2.1), affecting evolutionary processes at the species level (see below). Some pollinator groups (e.g., Hymenoptera, Lepidoptera) have already shown serious declines (reviewed in Potts et al., 2010), and this may be partly due to the habitat conversion history (i.e., historical landscape modification at a certain location, Bommarco et al., 2014), as well as the loss of particular habitat elements such as nesting or foraging sites (Ollerton et al., 2014; Potts et al., 2010; Scheper et al., 2014; Vanbergen, 2014). For example, a recent study indicated that agricultural expansion has reduced bee and wasp pollinator richness and composition in Great Britain, likely due to the reduction of floral diversity associated with monocultures (Senapathi et al., 2015). Further, the decrease of several bumble bee and butterfly species in Europe is probably attributable to the loss of unmanaged grasslands, heathlands, wetlands and bogs (Goulson et al., 2005), and key floral resources (Biesmeijer et al., 2006; Carvell et al., 2006). Similar responses have also been noted in honey bees by traditional beekeepers. For instance, in Southern France, beekeepers suggested that the reduction of flower populations, the expansion of the tree plantations and the decrease of pastures and meadows reduced the “vitality” [sic] of their honey bees, thus harming honey production (Elie, 2015; Velay and Velay, 2015).

Differences in ecological and morphological traits (e.g., feeding adaptations, mobility, body size, behaviour) can govern the response of pollinator species to changed environments, and their ability to persist in poor-quality environments (Hadley and Betts, 2012; Kennen et al., 2008; Marini et al., 2014; Morandin et al., 2007; Vanbergen, 2014). Pollinator species that are more specialised in habitat or food requirements (e.g., long-tongued bumble bees adapted to particular flower species) tend to be more vulnerable to land cover changes that alter the availability of food or nesting resources (Brosi, 2009; Goulson et al., 2008; Öckinger et al., 2010; Vanbergen, 2014; Vaudo et al., 2015; Weiner et al., 2014; however see Bommarco et al. (2010) for an exception), leading to the homogenization of pollinator communities dominated by common generalist species (Biesmeijer et al., 2006; Burkle et al., 2013; Carvalheiro et al., 2013; de Castro Solar, 2014; Grass et al., 2013; Marini et al., 2014; Weiner et al., 2014). Experimental studies in honey bees demonstrated one mechanistic basis for this; land use changes leading to the impoverishment of floral diversity (e.g., conversion of grassland into farmland, increase of monoculture) reduce the nutritional composition of pollen loads (Di Pasquale et al., 2013; Donkersley et al., 2014; Girard et al., 2012). Nesting behaviour influences pollinator response to land cover changes: above-ground nesters appear to be more sensitive to loss or isolation of high resource quality environments, such as natural or semi-natural land, than below-ground nesters (Williams et al., 2010; but see also section 2.2.2.4). This sensitivity may be because natural and semi-natural lands are richer in nesting resources for above-ground nesters (e.g., stems of perennial vegetation or dead wood) than converted areas; whereas the latter still harbour suitable patches of undisturbed soil available to below-ground nesters (e.g., field margins; Roulston and Goodell, 2011). However, note that some areas, although converted, may still contain nesting resources for above-ground nesters (see section 2.2.2).

Moderate levels of disturbance can sometimes increase habitat quality and availability, leading to increased pollinator diversity (Senapathi et al., 2015; Winfree et al., 2011). An example of this is the juxtaposition of different land cover types (an ecotone), which has been shown to affect pollinator diversity positively through edge effects (reviewed in Ries et al., 2004). These edges represent the transition zone between different environments (e.g., edges of a forest neighboring a crop field) and are biotically and abiotically distinct from their interiors. For this reason, they can support high pollinator diversity (e.g., Somme et al., 2014), although mainly due to a predominance of common species (Ries et al., 2004).

2.2.1.1.2 Effect of land cover composition on pollination

Studies have shown fruit set to be correlated with insect diversity in crops and wild plants (e.g., Garibaldi et al., 2013; Klein et al., 2002; Wilcock and Neiland, 2002). Thus, because changes in land cover composition negatively affect pollinator diversity (see above), and because greater pollinator diversity enhances pollination (Klein et al., 2009), habitat loss and habitat degradation should negatively affect fruit set, as has been shown in some crop systems (e.g., almonds, Klein et al., 2012; coffee, Klein et al., 2003a). Although it is difficult to demonstrate a direct relationship between changes in land cover composition *only* (i.e., without the common co-occurrence of changes in land cover configuration; see below), studies suggest that habitat loss affects wild plant reproduction. On this, it has been demonstrated that habitat loss more negatively affects insect- *vs.* self-pollinated plants (Aguilar et al., 2006; Batáry et al., 2013). Further, a recent study at the European scale (Clough et al., 2014) found strong correlations between the abundance of insect-pollinated plants and both bee pollinator abundance and diversity (positive correlation) and habitat loss/degradation (negative correlation).

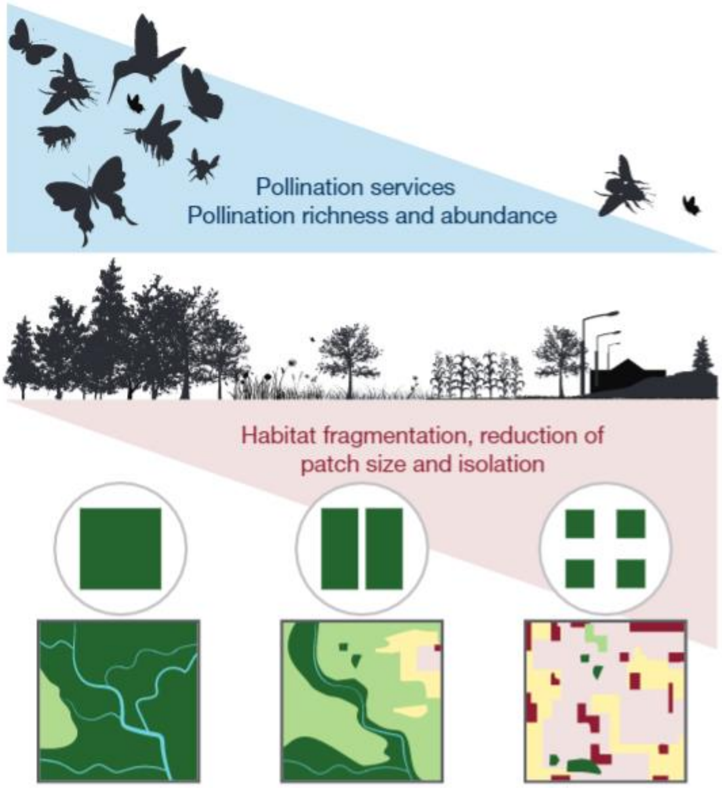
A recent modelling approach on the effect of habitat loss and fragmentation on pollination (Valiente-Banuet et al., 2015) proposed that these services should be considered as a function of the pollinator community as a whole. The study showed that pollination function is expected to decrease faster if generalist pollinators are lost or reduced, because these pollinators confer resilience to the pollination network (see section 2.2.1.2.1 and Chapters 1 and 3). Thus, they suggest that the maintenance of pollination interactions under habitat loss and degradation is also affected by the type of pollination network displayed by the community.

Related to the latter, a recent meta-analysis in crops (Kleijn et al., 2015) showed that although a more diverse landscape increases bee pollinator diversity, most of the crop pollination on average is likely provided by dominant (i.e., highly abundant) species. In agreement with this, Winfree et al. (2015) analysed a dataset from two regions and four crops in the USA, and found that the most abundant bee species are the ones contributing the most to crop pollination. Together, these studies suggest that decreases in bee habitat diversity may affect crop pollination less than it affects biodiversity. However, these studies contrast others identifying mechanistically how more diverse communities of pollinators can better support crop pollination by complementary or interacting behaviours among crops and over space or time (e.g., Brittain et al., 2013a; Greenleaf and Kremen, 2006; Hoehn et al., 2008; Klein et al., 2009; section 2.2.2). In addition, different pollinator species display “response diversity” (differential responses to the same environmental perturbations), and maintaining diverse pollinator communities, by enhancing response diversity, can increase the stability of pollination in the face of environmental or global change (Winfree and Kremen, 2009).

2.2.1.2 Changes in land cover spatial configuration

2.2.1.2.1 Effect of changes in land cover configuration on pollinators

Besides leading to habitat loss and degradation, changes in land use can fragment and alter the area and the spatial configuration of land cover and habitats. Thus, changes in land use can lead to habitat fragmentation (i.e., the sub-division of continuous habitat), affecting the size of habitat patches within an area, as well as their connectivity (Hadley and Betts, 2012; Hooke and Martín-Duque, 2012; Kearns et al., 1998). In these scenarios, although habitats are still available to pollinators, the fact that their relative spatial configuration has changed can lead to reductions in pollinator fitness (breeding success; Battin, 2004) and population sizes and thus can increase the chances of extinction.



4. **Figure 2.2.2 – Conceptual visualization of the effects of gradients of habitat fragmentation and natural and semi-natural land cover loss on pollinators and pollination.**
Landscape fragmentation (green rectangles) and increased loss of natural and semi-natural land cover (landscape cartoons) reduce patch sizes (smaller green rectangles with increased fragmentation) and inter-patch connectivity (more isolated green sections in cartoons with increased land cover change and fragmentation), negatively affecting pollinator richness and abundance, and pollination. Grey lateral triangles show gradients of landscape modification (right) and pollination, pollinator richness and abundance (left). Modified from Steffan-Dewenter and Westphal (2008).

Recent studies have shown that variation in landscape configuration can affect pollinator richness, species diversity and evenness in indirect and complex manners. A continental analysis of wild bees and butterflies in Europe (Marini et al., 2014) showed that species evenness and diversity were negatively correlated, and that while patch area related negatively to pollinator evenness, connectivity showed the opposite relationship. These results agree with what was observed by Winfree et al. (2011) for abundance and diversity of an array of pollinators. In that study, there were, however, differences among pollinator groups. On the one hand, bees were the most negatively affected by habitat fragmentation and loss (referred to as “land use” by the authors), followed by butterflies and hoverflies. On the other hand, larger vertebrate pollinators (i.e., birds, bats) were more positively affected by habitat fragmentation and loss (Table 2.2.1). This difference could be due to the greater dispersal ability of large vertebrates or to a bias in the analyzed datasets (Winfree et al., 2011). A more recent meta-analysis of bee species richness and abundance found little effect of landscape configuration (Kennedy et al., 2013), although it identified that loss of connectivity negatively affects social bee abundance. Overall, fragmented habitats may be able to maintain a greater level of pollinator diversity (related to this, see the concept of agricultural matrix, treated in section 2.2.2 and Chapter 6). However, although it is well established that landscape connectivity and especially the surrounding habitat availability correlate with components of biodiversity (e.g., Prugh et al., 2008), few studies have explicitly examined connectivity effects on pollinators, and this remains an important knowledge gap in the topic.

There is strong support for the hypothesis that the presence of resource-rich locations within fragmented landscapes increases pollinator diversity and richness (e.g., Klein et al., 2007). On this, a recent meta-analysis of 39 studies (605 sites) evaluated the effects of farm and landscape management on wild bees for 23 crops (Kennedy et al., 2013). The study showed that wild social and solitary bee species richness and abundance were higher in fields surrounded by environments considered by experts to provide more floral and nesting resources for pollinators ("high-quality habitats"). Similar results were also obtained in grasslands and almond plantations for other pollinator groups (e.g., butterflies and hoverflies, Öckinger et al., 2012; flies, wasps and non-*Apis* bees, Saunders and Luck, 2014), for invertebrates (including pollinators; Gonthier et al., 2014), for wild bee abundance and diversity (Winfree et al., 2009), and for (mainly bee) pollinator abundance and richness (Clough et al., 2014; Ricketts et al., 2008; Shackelford et al., 2013).

2. Table 2.2.1 – Directionality of changes in pollinator species richness and pollinator abundance with increasing values of land use change
(correlated positively with habitat loss/degradation and fragmentation). Values indicate proportion of experimental studies showing support for each of the negative, neutral and positive responses. Values in parentheses indicate number of studies (modified from Winfree et al., 2011).

	Directionality of pollinator response				
Group	**Negative**	**Neutral**	**Positive**	**Negative:Positive**	**Negative:All**
Bees	0.40 (81)	0.47 (94)	0.13 (27)	3.0:1	0.4:1
Butterflies	0.39 (88)	0.39 (88)	0.22 (47)	1.9:1	0.4:1
Flies (Syrphids)	0.40 (18)	0.30 (14)	0.30 (14)	1.3:1	0.4:1
Birds	0.32 (24)	0.27 (20)	0.41 (30)	0.8:1	0.3:1
Bats	0.22 (9)	0.29 (12)	0.49 (20)	0.5:1	0.2:1

Habitat loss, habitat degradation and fragmentation can lead to a cascade of species extinctions, often after a delay of several decades (Krauss et al., 2010; Kuussaari et al., 2009). Studies in grasslands have shown that extinction rates differ among pollinator groups, with bees declining faster than butterflies and hoverflies (Bommarco et al., 2014). Mobility also affects responses, with larger species being able to traverse greater distances (Greenleaf et al., 2007) and being thus less sensitive to changes in habitat area (Bommarco et al., 2010; Marini et al., 2014 – but see Williams et al., 2010). Sociality is a good predictor of the response of different pollinator taxa to variation in landscape structure, with social species being more sensitive to habitat loss and fragmentation. This may be due to the fact that most social insects are above-ground nesters, making substrate availability an important limiting factor for colony establishment and survival (Ricketts et al., 2008). A global meta-analysis indicated that social species are negatively affected by isolation from little-disturbed areas (Williams et al., 2010). However, even within social groups, interspecific variation in size, dispersal and foraging abilities plays a role in defining the ability of different taxa to survive under large landscape change (Vanbergen, 2014). For example, some *Bombus* species are able to forage over longer distances than their congeners, which is expected to improve their survival in fragmented landscapes (Carvell et al., 2012).

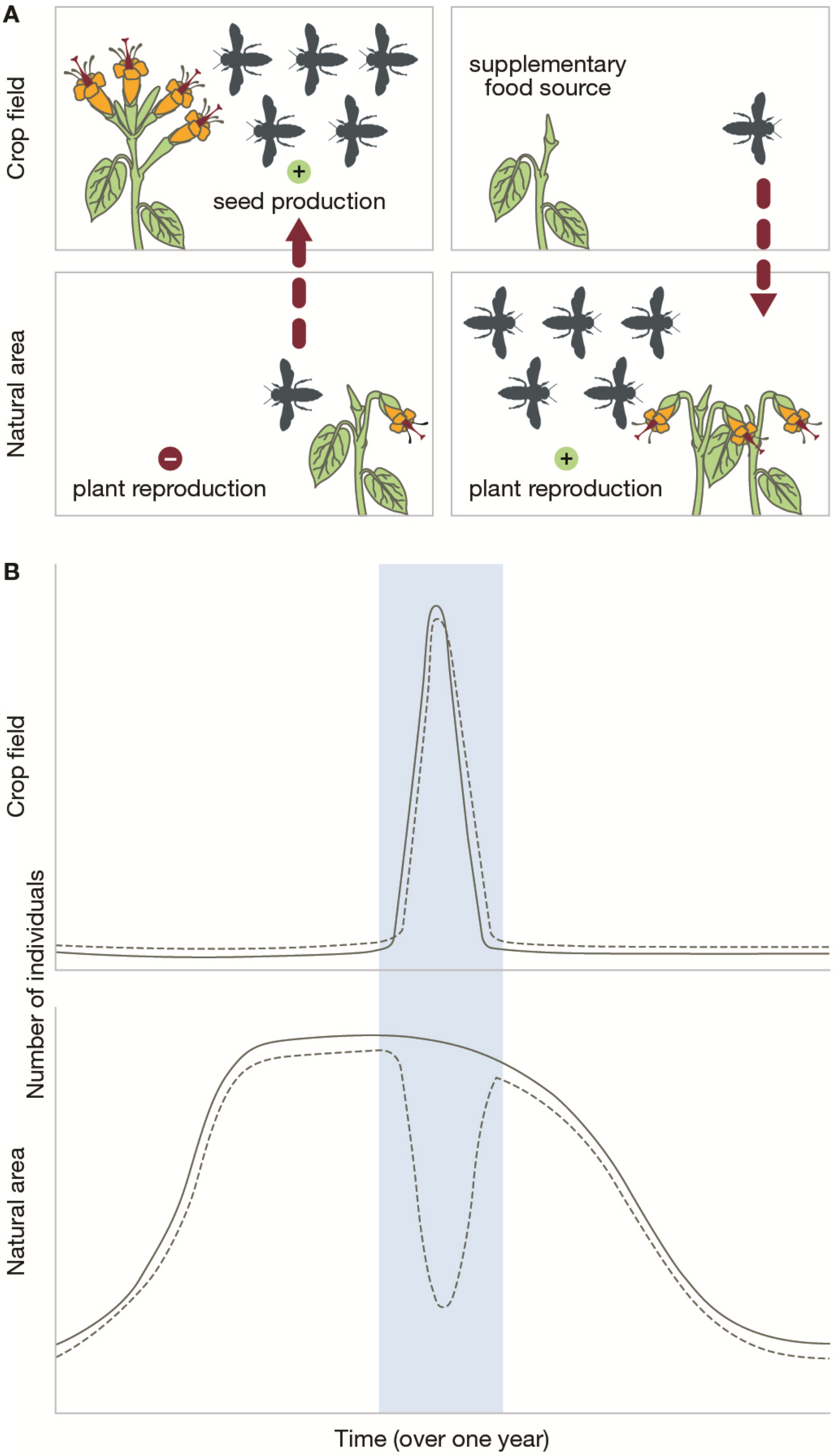
Habitat patch size reduction and fragmentation decrease species richness, and negatively affect the ecological network link richness (see Box 2.2.2), leading to network contraction (Sabatino et al., 2010; see also Chapters 1 and 3). Fragmentation reduces modularity, because with species loss, modules shrink, merge and finally disappear (Olesen *et al.*, 2007). Small patches have more unpredictable resources, which benefit generalists over specialists (Burkle and Knight, 2012). Indeed, generalists are better at changing resources (rewiring) and thus are less sensitive to extinction after the disappearance of other species in the network (i.e., secondary extinctions; Astegiano et al., 2015; Burkle et al., 2013; Memmott et al., 2004). The result of rewiring is that pollination networks in small fragments have higher connectance (i.e., more of the possible links are realized) and more homogenized pollinator communities (reviewed in Hagen et al., 2012). Pollinator networks often have a nested structure, in which dominant generalist species are connected to rarer species in the network; this nestedness is predicted to lend stability to the plant-pollinator community (see Vanbergen, 2014 and Chapter 3). Related to this, fewer species and links in pollination networks lower their resilience to disturbance (Lever et al., 2014). Larger, more interconnected patches improve the general survival of plant-pollinator communities due to increased ecological redundancy and decreased probability of extinction of keystone species (Burkle et al., 2013; Tscharntke et al., 2005). It is important to note that today most of our knowledge on this topic comes from network modelling (e.g., Memmott et al., 2004; Petanidou et al., 2008), because experimental data are only starting to become available (e.g., Aizen et al., 2012; Astegiano et al., 2015; Burkle et al., 2013; Winfree et al., 2014).

Finally, both the size of the patches and their connectivity can have evolutionary implications, because they affect the demography of pollinators and plants. Indeed, reduced population sizes and pollination specialization are generally associated with reduced genetic diversity (e.g., Goulson et al., 2008; Packer et al., 2005), which is exacerbated by lower migration between poorly connected habitat patches and lower chances of recolonization between fragments (Kremen et al., 2007). Small population sizes and low genetic diversities can reduce the mean individual fitness of the population (i.e., Allee effect), decrease the ability of a species to recover from stochastic events (e.g., diseases, climatic events), lower the possibility of adaptation, and/or increase the negative effects of strong genetic bottlenecks and inbreeding depression (Hartl and Clark, 2006). Only a few studies (e.g., Davis et al., 2010; Jha and Kremen, 2013a) have examined genetics changes in pollinators in response to landscape changes, and no studies have been done in non-temperate regions.

2.2.1.2.2 Effect of changes in land cover configuration on pollination

Higher land cover fragmentation has been shown to affect plant reproductive success negatively. On this, a review and meta-analysis of 53 studies and 89 wild plant species (Aguilar et al., 2006), indicated that plant sexual reproduction is strongly and negatively affected by habitat fragmentation. Further, the study indicated that this is particularly true for self-incompatible plants, thus demonstrating the role of fragmentation in reducing pollination. Along with these results, a recent experimental study (Blaauw and Isaacs, 2014) indicated that the most important factor for seed set is patch size, and that although a richer plant diversity increases insect pollinator diversity, high plant diversity in a small patch reduces seed set per flower. The authors suggest that in small patches a more diverse plant composition may reduce the efficiency and specificity of pollen transfer, thus negatively affecting seed production. Studies also confirmed that the level of pollination specialization does not define the sensitivity of plants to landscape fragmentation, thus rejecting the idea that specialist plants are more sensitive to habitat fragmentation than generalist ones (Aguilar et al., 2006; Aizen et al., 2002; Ashworth et al., 2004). Related to these results, a recent meta-analysis of animal-pollinated woody plants (Breed et al., 2015) showed that landscape fragmentation diminishes the genetic diversity of the received pollen, which contributes to genetic impoverishment. Although not yet investigated, it is also likely that dioecious animal-pollinated plants are more sensitive to fragmentation than their monoecious counterparts.

Fragmented landscapes and the presence of natural areas have also been shown to affect fruit set through pollinator spill-over, namely the movement of pollinator species and pollination from one land cover type to another (Figure 2.2.3). This spill-over occurs because of temporal and spatial variation of resource availability in the adjacent areas. Thus, pollination is no longer fulfilled in the location where the pollinator comes from but rather towards which it moves (Blitzer et al., 2012). Spill-over is expected to occur in fragmented landscapes where there is a relatively high proportion of resource-rich locations (e.g., parts of Europe, many tropical regions) and it has been shown to provide effective pollination for many crops, such as for instance watermelons (*Citrullus lanatus*), blueberries (*Vaccinium* sp.), coffee (*Coffea* sp.) and atemoya (*Annona × atemoya*; reviewed in Blitzer et al., 2012). The few studies that sought to evaluate the importance of spill-over towards natural areas (e.g., Cunningham et al., 2013; Lander et al., 2011) observed it occurring in many pollinator groups (e.g., bumble bees, solitary bees, hoverflies) and from several types of agricultural areas (e.g., fields, home-gardens, organic farms) towards diverse natural land-cover types (e.g., rainforests, grasslands, temperate semi-natural areas; Gabriel et al., 2010; Hagen and Kraemer, 2010; Holzschuh et al., 2011; Westphal et al., 2003).



5. Figure 2.2.3. - Pollinator spill-over.
A) From natural/semi-natural to managed areas during crop blooming and from managed to natural areas after crop blooming. Dashed arrows indicate direction of pollinator movement. B) Conceptual representation of changes in number of flowering plants (lines) and pollinators (dashed lines) during a year, in a crop field (top) and a neighboring natural area (bottom). Blue shaded area represents the moment of the year when pollinator spill-over occurs. Modified from Blitzer et al. (2012).

Habitat isolation and connectivity can also affect the delivery of crop pollination, measured as the relationship between fruit set and/or crop visitation rates of different pollinators and distance to resource-rich habitats

(Chapter 1). Synthesis of data across several pollinator taxa, pollinated crops and wild plant species from different biomes showed that pollinator diversity and abundance, and flower visitation, decrease with increasing distance from resource-rich locations (Garibaldi et al., 2011; Klein et al., 2007; Ricketts et al., 2008). Ricketts et al. (2008) synthesized results from 23 studies representing 16 crops on five continents and found exponential declines in pollinator richness and native visitation rate with increasing distance from resource-rich areas. This correlation was more negative for visitation rate than for pollination richness. Visitation rates dropped more steeply in tropical than in temperate regions, and were steeper but not significantly different for social compared to solitary bees (see also Klein et al., 2002). Despite the steep decrease in native pollinator visitation, no strong decline in crop fruit and seed set was found in this meta-analysis, probably due to sufficient pollination at the lowest visitation rates or to supplemental pollination by managed honey bees.

Managed species like some honey bees (e.g., *A. mellifera, A. cerana*) may provide sufficient pollination for several crops, even in fields distant from resource-rich areas. However, in the light of multiple environmental threats (Vanbergen et al., 2013), reliance on a single pollinator species for pollination delivery might be risky, compared to a diverse native pollinator community (Fontaine et al., 2006; Kremen et al., 2002; Ricketts, 2004). Supporting this view, a more recent meta-analysis (Garibaldi et al., 2011) used additional studies to those used in Ricketts et al. (2008) and indicated that diversity of wild pollinators and fruit set decreased with increasing distances to resource-rich areas in all crops evaluated. Such results had also been identified as a trend by Klein et al. (2007) and have been recently shown to hold regardless of the presence of managed honey bees, indicating that wild pollinators are important contributors to fruit set even in the presence of managed bees (Garibaldi et al., 2013). Some possible reasons for this might be that diverse wild pollinators provide a better pollination, for example through greater cross-pollination (Garibaldi et al., 2013; Woodcock et al., 2013), higher efficiency of pollination by complementarity of their foraging behavior (i.e., niche complementarity; Brittain et al., 2013a), or through positive effects of some pollinators on the pollination function of other pollinators (i.e., functional facilitation; Greenleaf and Kremen, 2006; Klein et al., 2009). The minimum proportion of resource-rich areas needed to maximize fruit set is likely to differ among plant species, based on their respective reproductive dependence on (wild) pollinators (Klein et al., 2007). Global and continental meta-analyses and syntheses (e.g., Garibaldi et al., 2014; Tscharntke et al., 2005; Westphal et al., 2003; Winfree et al., 2009) identified values of the minimum proportion of natural areas in close vicinity to crop fields necessary to maximize fruit set: 2-5% for Westphal et al. (2003) and Winfree et al. (2009), and 20-30% for Tscharntke et al. (2005; see also Kremen et al., 2004 and Morandin and Winston, 2006 for specific examples). The distance to fields in which these resource-rich areas should occur in order to increase pollinator abundance and fruit set were estimated to range from 200m (Garibaldi et al., 2014) to 2400m (Kremen et al., 2004).

The effects of land-use change on the structure of landscapes and their overall consequences for pollinators and pollination, and main sources of evidence, are summarised in Table 2.2.2.

3. Table 2.2.2 – Summary of the effects of several consequences of land use change on pollinator diversity and pollination.

Levels of evidence and main studies demonstrating the effect are given.

Factor	Effect	Review/Meta-analysis/Continental study
Increased landscape modification	Landscape modification that enhances heterogeneity increases diversity and pollinator spill-over (well established)	Winfree et al., 2011; Potts et al., 2010; Blitzer et al., 2012; Senapathi et al., 2015
	Landscape modification that increases uniformity homogenizes pollinator communities (well established)	Biesmeijer et al., 2006; Carvalheiro et al., 2013; Marini et al., 2014; Weiner et al., 2014
Presence of resource rich habitat	Increases nesting (established but incomplete) and foraging resources (well established)	Klein et al., 2007; Potts et al., 2010; Williams et al. 2010; Vaudo et al., 2015
	Increases pollinator diversity, richness and abundance (well established)	Kennedy et al., 2013; Gonthier et al., 2014; Winfree et al., 2009; Ricketts et al., 2008; Shackelford et al., 2013; Winfree et al., 2011; Marini et al., 2014
	Reduces chances of extinction (established but incomplete)	Goulson et al., 2008
	Increases fruit set (well established)	Garibaldi et al., 2011; Klein et al., 2007
High connectivity	Increases evenness (established but incomplete)	Marini et al., 2014
	Increases social bee abundance (established but incomplete)	Kennedy et al., 2013
	Increases diversity and richness (well established)	Winfree et al., 2011
	Decreases chances of extinction (established but incomplete)	Goulson et al., 2008
High isolation	Decreases diversity and richness (well established)	Ricketts et al., 2008; Garibaldi et al., 2011
	Reduces fruit set (well established)	Garibaldi et al., 2011
Increased fragmentation	Reduces diversity and abundance (well established)	Hadley and Betts, 2012; Kennedy et al., 2013; Steffan-Dewenter and Westphal, 2008; Winfree et al., 2011; Ollerton et al., 2014
	Reduces fitness of self-incompatible plants (well established)	Aguilar et al., 2006
	Increases selfing of outcrossing plants (established but incomplete)	Breed et al., 2015

Box 2.2.1: Network concepts

-link: ecological interaction, e.g. trophic or mutualistic interaction (Bascompte and Jordano, 2007).

-network: a set of nodes (species) connected through links. In the framework of pollination networks, they are graphical representations of which plant species interacts with which pollinator species, and how strong the interactions are.
-link richness: number of realized links in a network.
-connectance: the proportion of possible links that are realized (Bascompte and Jordano, 2007). Increased connectance confers higher network stability.
-modularity: Links between nodes are heterogeneously distributed. In networks, link-dense sections are termed modules, and species within a module are linked more tightly together than they are to species in other modules. The extent to which species interactions are organized into modules is termed the modularity of the network (Olesen et al., 2007).
-nestedness: measure that describes interactions in the network. It represents a pattern of interaction, in which the set of species with which specialists interact is a subset of the species with which generalists interact (Bascompte and Jordano, 2007).
-rewiring: link switching, usually after biotic and/or abiotic environmental changes that modify the plant-pollinator community (Hagen et al., 2012).

Box 2.2.2: Landscape concepts

-land cover: observed (bio)physical cover on the Earth's surface (Di Gregorio and Jansen, 2005)
-land use: the arrangements, activities and inputs people undertake in a certain land cover type to produce, change or maintain it (Di Gregorio and Jansen, 2005).
-habitat: the range of environments suitable for *a certain species* (Fischer and Lindenmayer, 2007). This is the range of locations in which the ecological conditions that allow a given species to establish and survive exist.
-habitat loss: Loss of habitat for a particular species (Fischer and Lindenmayer, 2007). In the case of pollinators, this relates mainly to the loss of nesting and floral resources.
-habitat degradation: gradual deterioration of habitat quality (Fischer and Lindenmayer, 2007). In these circumstances, a species can still occur, but may decline, occur at a lower density, or be unable to breed. For instance, in the case of pollinators, this can occur when the habitat harbors altered floral resources, which results in reduced flower numbers or diminished nutritional value.
-connectivity: measure of connectedness between patches harboring suitable conditions for a given species. (Fischer and Lindenmayer, 2007). The opposite of isolation.
-fragmentation: breaking apart of continuous suitable areas into multiple patches (Fischer and Lindenmayer, 2007).
-landscape: a mosaic of interacting ecosystems; an area spatially heterogeneous in at least one factor of interest (Turner, 2005). In the case of pollination and pollinators, this can be an area heterogeneous in the occurrence of habitats for different species.
-isolation: measure of separation between existing patches harboring suitable conditions for a given species (Fischer and Lindenmayer, 2007). The opposite of connectivity.

2.2.2 Land management

Land management such as agricultural and conservation practices has a great influence at both landscape and local scales on the nesting and foraging environment of pollinators. In this section we assess the main local-scale land management drivers, which determine pollinator community structure and associated pollination in arable, grassland, horticulture and agroforestry systems worldwide.

2.2.2.1 Contrasting forms of agricultural management systems

2.2.2.1.1 Organic or diversified farming systems versus conventional monoculture management

Increased land cover heterogeneity within the fields/farms can increase pollinator abundance, diversity and pollination effectiveness even in landscapes with few natural land cover types (Batáry et al., 2011; Holzschuh et al., 2008; Kennedy et al., 2013; Rundlöf et al., 2008; Williams and Kremen, 2007). The lower levels or lack of inorganic fertilisers, pesticides, increased number of cultivated crops, smaller field sizes, diverse edge vegetation and higher local complexity, which can be defined as within-field wild plants, crops or plant diversity in the crop margins, can have considerable positive effects on pollinators and pollination (Garibaldi et al., 2014; Kremen and Miles, 2012; Shackelford et al., 2013). Traditional land-use systems included classically low-input low-output systems with high variability throughout Europe in the form of livestock systems, arable and permanent crop systems, and mixed systems, persisted mainly in upland and remote areas (Plieninger et al., 2006). However, most of these traditionally managed landscapes have disappeared today due to intensification or land abandonment (Stoate et al., 2001). Environmentally friendly management methods, such as organic farming, diversified farming systems, polyculture farming, crop rotations, and conservation practices within agricultural management prescribed under policy instruments such as agri-environment schemes, are based on such practices (see more details in Chapter 6; see definitions in the glossary). Also integrated pest management (IPM), which combines biological and cultural control with informed use of chemicals as part of a system approach to provide targeted and efficient pest management solutions, could have beneficial effects on pollinators by improving habitat and minimizing the use of insecticides applied (Gentz et al., 2010; see also in section 2.3.1).

Several studies suggest that there are positive effects of diversified farming systems and organic management relative to conventional monocultures (Kennedy et al., 2013; Kremen and Miles, 2012; Nicholls and Altieri, 2013; for definitions and more details see the glossary). A large meta-analysis found that more than 70% higher total bee abundance and 50% higher total species richness of wild bees could result from diversified farming systems (Kennedy et al., 2013). Such differences were found for Mediterranean and temperate regions, with benefits being less accentuated in the tropics (Kennedy et al., 2013). Increased numbers of wild pollinators in organic fields was shown to correlate strongly with pollination success; for example, a study on canola seed set in Canada revealed 3 to 6 times lower seed set on conventional and GMO canola fields using insecticides and herbicides than on organic sites of similar field size (Morandin and Winston, 2005). Strawberry (*Fragaria × ananassa*) pollination was found to be higher at farms 2-4 years after conversion to organic farming (Andersson et al., 2012) (see more details in Chapter 6).

Effectiveness of organic management depends on the landscape context, the crop type, the management of the organic farms, soil conservation and the species considered (Arnhold et al., 2014; Brittain et al., 2010). Effects of local-scale conditions such as diversity in crops and management type may strongly interact in managed fields. Meta-analyses by Kennedy et al. (2013) found that both field-scale diversity and organic farming have distinct, positive impacts on wild bee abundance. Results suggested that higher vegetation diversity in conventional crop fields may increase pollinator abundance to the same extent as organically managed fields with low vegetation diversity (see also Winfree et al., 2008). However, organic management might produce richer bee communities than conventional management independently from the level of field diversification (Kennedy et al., 2013). Characteristics of agricultural disturbance may not always be mitigated by organic management, depending on the underlying mechanisms affecting pollinator populations (e.g., Forrest et al. (2015) found differences in diversity, but not in functional diversity of bees comparing organic and conventional fields, which functional diversity was lower in both farm types than in natural land cover types).

At the field scale organic management can enhance both continuity of wild plant distribution and flowering, providing continuous flower resources for pollinators. Rollin et al. (2013) and Sarthou et al. (2013) have demonstrated that in entomophilous crops where flower resources are very important but of short duration, wild flower diversity in the field (i.e. weeds with flowers) is more important for favouring diversity of wild bees, and is promoted by organic farming. Therefore, insect-pollinated plants might occur more evenly in organic fields and receive disproportionately higher pollination benefit from organic farming due to higher pollinator densities (Gabriel and Tscharntke, 2007).

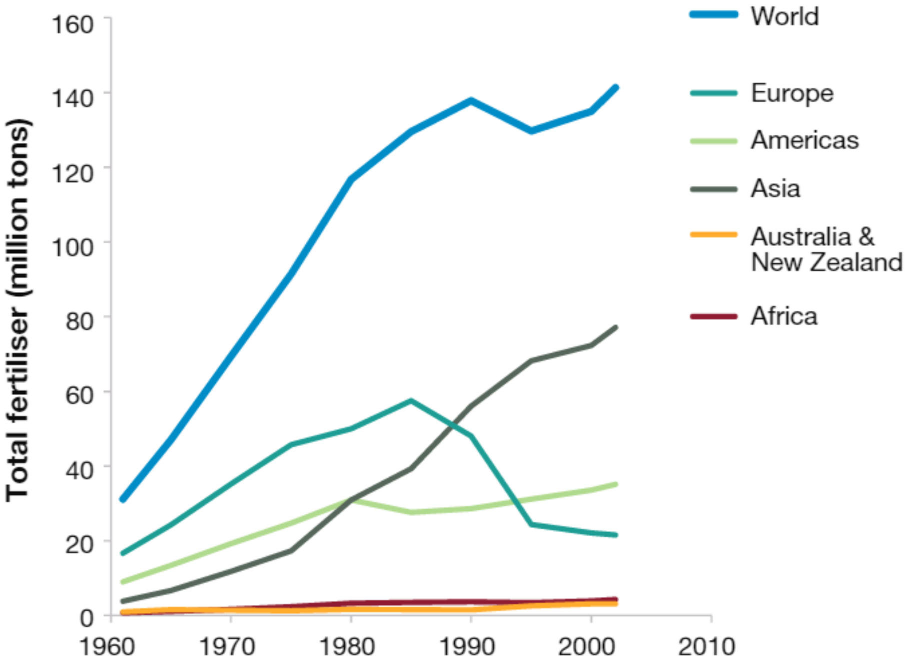
Benefits for biodiversity can be observed on organic farms at both farm and landscape scales; for example, greater bee, hoverfly and butterfly diversity was found in landscapes with a larger proportion of organic fields (Holzschuh et al., 2008; Gabriel et al., 2013; Rundlöf et al., 2008). Non-intensive field management using less chemicals and/or having more diversified farming system, e.g., organic farming, has positive effects more often in homogeneous rather than heterogeneous landscapes (Rundlöf and Smith, 2006; Tuck et al., 2014), however isolated organic farms may not provide any measurable benefit to local populations of pollinators and pollination (Brittain et al., 2010). Moreover, a recent study argues that observed differences in biodiversity between organic and conventional fields may be explained by greater cost-effectiveness of conservation efforts in low-productivity agricultural systems or on non-agricultural land, rather than organic management per se (Gabriel et al., 2013). However, Lüscher et al. (2014) showed a strong influence of local organic agricultural management on wild bees and a minor and inconsistent effect of the surrounding landscape, after accounting for the effect of geographic location. There might also be interacting effects of farming system and landscape heterogeneity on pollinator community composition and pollinator trait diversity. Decreasing landscape heterogeneity resulted in overall decline of species richness of hoverflies and wild bees, while taxonomic breadth only declined on conventionally managed farms (Andersson et al., 2013).

Not all studies found increased pollinator species richness/abundance or increased diversity of plants on organic farms. On 205 farms in Europe and Africa, Schneider et al. (2014) found that at farm scale, the diversity of bees was affected by the presence of non-productive land cover types rather than by the farming system (organic or not). Moreover, management type (organic vs. conventional) does not always match with plant or crop diversity. Conventional farms can be as diverse as organic ones (e.g., in Sweden – Andersson et al., 2005), while there are very large organic monocultures too (e.g., in South Africa – see Carvalheiro et al., 2012). In Europe, great differences exist in the implementation of organic farming or diversified agricultural management methods among EU-countries, resulting in a wide span of landscapes ranging from less intensively used and heterogeneous landscapes on the one hand to highly productive and monotonous landscapes on the other hand (Kleijn et al., 2006). Overall, there is a need for more careful experimental design to separate clearly the type of impacts that occur from organic and conventional agriculture (Roulston and Goodell, 2011).

Nevertheless, we can conclude that the creation or maintenance of more diverse agricultural landscapes may result in more diverse pollinator communities and enhanced crop and wild plant pollination. Local diversification and reduced intensity of land management will support pollinators and pollination, especially in simpler and more intensive landscapes.

2.2.2.1.2 Fertiliser use

Globally, agricultural management is increasingly using high levels of inorganic fertiliser in place of organic manures (e.g. Richards, 2001; Figure 2.2.4). Global demand for fertilizer is expected to show a successive growth of 1.8 per cent per year and to reach 200 million tonnes by the end of 2018 (FAO, 2014). Intensive fertiliser application per field can result in decreased diversity and cover of the less competitive wild plant species (Kleijn et al., 2009; Kovács-Hostyánszki et al., 2011). The lower number of flowering plant species, the lower flower abundance and the consequent reduction in floral resources decreases the number of pollinator species and their abundance, and the frequency of pollinator visits, which may have a negative impact on pollination success and plant reproduction (Ebeling et al., 2008). In plant-pollinator networks at small spatial scale the community structure may be relatively resistant to short-term bottom-up changes in the nitrogen supply, but sensitive to variation in the opportunistic behaviour and turnover of plant and pollinator species for years (Burkle and Irwin, 2009). For example, based on their larval host-plant characteristics, moths associated with plant species that are in decline, such as those associated with low nitrogen soil conditions, declined more rapidly (Fox et al., 2014).



6. Figure 2.2.4. Total fertiliser consumption worldwide and separately at the different continents during the last half century.
Data are shown in Million tonnes (FAO, 2014).

Nitrogen deposition (in interaction with air temperature and CO_2 level) may also change flower morphology, plant phenology and nectar chemistry, and through these pathways may alter pollinator mutualism. In a pumpkin case-study system, for example, bees tended to visit and consume nectar more frequently from plants grown under elevated N level, which significantly reduced worker bee longevity (Hoower et al., 2012). Nitrogen levels may affect flower number or size, which are important for pollinator attraction to plant individuals and communities; thus, nitrogen levels may influence plant biomass and reproduction directly as well as indirectly via changes in pollination (Burkle and Irwin, 2010).

2.2.2.1.3 Tillage

Around 70% of the bees are ground nesting (Michener, 2000). Soil surface disturbance caused by tillage practices may have destructive effects on pollinator species, destroying nests of below-ground nesting bees (Williams et al., 2010). It changes also the composition and abundance of wild plant species (see more in Chapter 6). There is still a research gap on the effects of tillage on pollinators. One study found no tillage effect on the abundance of flower-visiting *Peponapis pruinosa*, a bee species nesting within tillage depth in pumpkin fields (Julier and Roulston, 2009). While a no-tillage system was found to be beneficial for wild pollinators in squash and pumpkin fields, another study showed three times higher density of squash bees (*Peponapis* spp. and *Xenoglossa* spp.) in no-till fields than in tilled farms (Shuler et al., 2005). Ullmann et al. (2014) suggested that while tillage negatively impacted offspring survival of *P. pruinosa*, however, some individuals that probably nested below the tillage zone survived this disturbance allowing the population to persist.

Tillage systems have a great influence on topsoil organic matter content, and other soil properties, which influence erosion and water quality. A global literature review (Palm et al., 2014) found in many cases increased soil carbon sequestration with no-till compared to conventional tillage. Moreover, a global meta-analysis across 48 crops and 63 countries showed that overall no-till reduces yields, yet when no-till is combined with the other two conservation agriculture principles of residue retention and crop rotation, its negative impacts are minimized, and moreover it significantly increases rainfed crop productivity in dry climates (Pittelkow et al., 2015). No-till farming was adopted on 111 million ha worldwide in 2009, corresponding to the growth rate of 6 million ha per year (Derpsh et al., 2010).

2.2.2.1.4 Weed control management

Weeds provide important, often exclusive, foraging resources for pollinators in agricultural landscapes (Carvalheiro et al., 2011; Hawes et al., 2003). Their removal, by physical means (e.g. tillage) or chemical herbicides (see also effects of increased use of herbicides on herbicide-tolerant genetically modified (GMO) crops in section 2.3.2) can cause decline of native pollinators in agroecosystems (Richards, 2001; Steffan-Dewenter et al., 2005). North America and countries in Western Europe were the main market for herbicides during the second half of the 20th century (Schwinn ,1988). Herbicides have experienced a three-fold increase of use in Canada and two-fold increase in the USA since 1971 (Freemark and Boutin, 1995; see also section 2.3.1 on pesticide effects). From 1990-2010 applications of glyphosate on maize, soy and cotton in the US have increased from near zero to ~90,000 tons/yr (USDA-NASS, 2012).

Weed control may also be achieved by crop rotation, where sowing successive crops affects weed seed-banks and weed communities (Ball, 1992). Rotation on the one hand can lead to reduced weed populations, especially if small grain crops are sown or other crops that smother the weeds. On the other hand, when crops are rotated, diversity of weeds increases even as density decreases, creating more favourable food conditions for pollinators with crop rotation (Ball, 1992).

Conventional agricultural system monocultures (especially continuous monocultures without rotations) often result in uniformity of crop flowers and low diversity of weed species, restricting foraging resources to only to a few species that are visited by a depauperate pollinator community, mostly generalist species (Diekötter et al., 2010; Herrmann et al., 2007). Moreover, crop flowers usually bloom only for a short period, leaving pollinators without food in the rest of the season (Kovács-Hostyánszki et al., 2013). However, if crop species are left to grow together with ruderal plants (i.e. those plants that grow on disturbed lands), more diverse pollinator assemblages may benefit crop pollination, as was shown in sunflower fields, South Africa (Carvalheiro et al., 2011). Wild plants often reach highest diversity and cover in the field edge, through natural regeneration or sown flower strips (see more in Chapter 6), promoting pollinator abundance (Carvell et al., 2004; Lagerlöf et al., 1992).

2.2.2.1.5 Pesticides

Effects of pesticides are treated in detail in section 2.3.1.

2.2.2.1.6 Mono- versus polyculture systems

From the floral resources point of view, crop diversity in space, time and at a genetic level strongly influences pollinator communities and pollination success of crops and wild plants (see also Chapter 6). Like natural communities, polyculture systems can provide continuity of resources through time for pollinator communities when crops flower sequentially (Rundlöf et al., 2014). The diversity of agricultural crops tends to be greater in the developing than developed world (Aizen and Harder, 2009). Different cultivars planted together can help pollinator species and communities to persist more continuously during the vegetation season on the fields and provide efficient pollination for plant species flowering sequentially (Mayfield et al., 2008). Mixed cropping may also contribute positively towards pollination as well as its financial benefits to farmers, especially in developing world. For example, different maize varieties (short-cycle and long-cycle maize) in Yucatan, Mexico are planted together to supply bee communities with pollen during the wet season and sustain the bee populations until the next floral season of maize (Tuxill, 2005).

Facilitative interactions may occur among close relatives of plants or between phylogenetically unrelated but anatomically similar plant species. These species can jointly attract pollinators, which then experience decreased pollen limitation and increase reproductive success of both species (Moeller, 2004). Based on species-specific responses, floral traits such as similar flower colours contribute to interspecific facilitation of pollinator visitation (Hegland and Totland, 2012). On the other hand, the movement between conspecific and heterospecific flowers may lead to the deposition of more heterospecific pollen on stigmas, causing pollen clogging or chemical inhibition of pollen tube growth (Schüepp et al., 2013; Wilcock and Neiland, 2002). Such interspecific pollen transfer is a common phenomenon, with potential ecological and evolutionary consequences for the plants (Mitchell et al., 2009), but also for the crop yield.

2.2.2.1.7 Management of crop genetic diversity and cross pollination in hybrid systems

Genetic variability within a crop species can affect insect pollination. Increasing crop genetic diversity has the potential to enhance pollination by more viable cross-pollination (Hajjar et al., 2008). Pollinators often prefer one variety over another, and it is not always the commercially desirable variety. For example, in the case of an almond orchard studied by Jackson and Clarke (1991) it was found that honey bees predominantly visit only one cultivar and cross-pollination only results from accidental or rare visits involving two or more compatible cultivars. If one crop variety provides no or only low amounts of nectar or pollen, it has to be planted in fields mixed with better foraging varieties to provide sufficient pollination and promote cross-hybridization and better fruit set (e.g. melon, Bohn and Mann, 1960; almond, Jackson and Clarke, 1991). Many orchard crops need cross pollination between varieties to give optimal yield, e.g. many raspberry (*Rubus* spp.) varieties need pollen from a different variety to set fruit (Colbert and Oliveira, 1990).

Pollinator species can also increase the pollination efficiency of each other, resulting in higher pollination success for the pollinated plants (Greenleaf and Kremen, 2006). For example, in the case of hybrid seed production for some companies (e.g., the production of commercial seeds of sunflower) male-fertile and male-sterile sunflower plants are generally planted in alternating rows. The searching strategy of honey bees is generally more focused, foraging for either nectar or pollen, and therefore they do not cross between rows, until native bees, and other pollinators (moths, butterflies) collecting both resources alter the honey bees' behaviour as they try to avoid contact with the native pollinators and are chased to visit more frequently across rows, increasing cross-pollination rates (Carvalheiro et al., 2011; Greenleaf and Kremen, 2006). Wild flowers left in crop fields can also increase pollinator diversity, which can facilitate honey bee movement and therefore crop productivity (Carvalheiro et al., 2011).

2.2.2.1.8 Mass flowering crops

Monocultures that provide mass-flowering resources potentially have positive effects on pollinators. Edible oilseed crops provide large amounts of readily accessible pollen and nectar. For example, canola (oilseed rape, *Brassica napus*) is planted at a density of 350,000–700,000 plants per hectare, producing huge number of flowers (Hoyle et al., 2007), attracting many pollinators, and receives high numbers of flower visits per time unit, e.g. a single bumble bee visits on average over 400 canola flowers per bout and approximately 2,000 flowers per hour (Hoyle et al., 2007). Oilseed crop production is steadily increasing worldwide except in Africa (FAO, 2014). Mass-flowering crops receive important pollination from both managed and native pollinators; however, field management (e.g. pesticide and fertiliser use) in mass-flowering crops can have an important negative effect on pollinator richness and abundance (e.g. in pigeon pea (*Cajanus cajan*) fields in Kenya (Otieno et al., 2011), or on the reduction in floral diversity and floral resources over time, see 2.2.2.1.6).

There are diverse effects of mass-flowering crops on pollinators. Canola can have a positive effect on colony growth of bumble bees (mainly for short-tongued bumble bees; Diekötter et al., 2010) or number of brood cells of solitary bees at landscape and local scales, most likely depending on the species' foraging/dispersal distances (Holzschuh et al., 2013; Westphal et al., 2009). Other mass-flowering crops such as late-flowering red clover are important flower resources for bumble bees and enhance their reproduction by increasing temporal resource continuity, following bloom of other crops (Rundlöf et al., 2014). However, in a study of wood-nesting solitary bees, population growth of most species was not stimulated by the resource pulse provided by canola early in the year, but by persistent resources provided by wild flower patches after mass flowering (Diekötter et al., 2014). In the long run mass-flowering crops can enhance abundances of generalist pollinators and their pollination (Holzschuh et al., 2011). Mass-flowering crops may temporarily compensate for the effects of landscape change. In effect this may hide an increased vulnerability due to reduced heterogeneity of land uses and floral resources, which then becomes exposed when area devoted to mass-flowering crops diminishes (Jansson and Polasky, 2010).

Interactions between mass-flowering crop fields and wild flower patches occur at different spatial scales, altering resource use of pollinators and potentially reducing wild plant reproduction (Holzschuh et al., 2011). The bloom of flowers offered by mass-flowering crops may attract pollinators away from co-flowering wild plants in adjacent natural patches thereby reducing their reproductive success at the expense of improved crop yield (Holzschuh et al., 2011). The expansion of bee-attractive biofuel crops such as canola can result in transient dilution of crop pollinators and increased competition for pollinators between crops and wild plants, leading to reduced pollination of concurrently flowering wild plants (Holzschuh et al., 2011; see also Chapter 6). Although canola overlaps in pollinator niche with many co-flowering wild plants, and may compete with them via reduced flower visitation, crop pollen deposition on wild plant stigmas was found to be low, suggesting that stigma-clogging with heterospecific pollen is unlikely to be the cause of reductions in seed set of wild plants (Stanley and Stout, 2014). In contrast, plants in the adjacent areas that flower two to three weeks after blooming of canola, may benefit from enhanced local bee abundances (Kovács-Hostyánszki et al., 2011; see also spill-over in section 2.2.1). Pollinators often have to move back to the wild flower land cover elements at the end of crop flowering, becausethese elements provide the only – and in general more permanent – foraging resources (Kovács-Hostyánszki et al., 2013). At this time pollination of native plant species in the nearby wild flower patches can also profit from the spill-over of diverse and abundant pollinator communities, supplemented with efficient pollen and nectar gain in the adjacent crop fields (Kovács-Hostyánszki et al., 2013; see also section 2.2.1) (Figure 2.2.3). Thus, spatial and temporal changes in landscape composition can cause transient concentration or dilution of pollinator populations with functional consequences (Tscharntke et. al., 2012).

2.2.2.1.9 Orchards

Some of the economically most important fruit trees such as apples, almond, cherries (cross pollination essential) and pears (partly or entirely self-sterile) require insect pollination (Abrol, 2012), which affects both the quantity and quality of production, influencing size, shape, taste and seed number (Garratt et al., 2014). Pollination in orchards is usually supported by honey bees, while wild pollinators play also important role in fruit tree pollination (Brittain et al., 2013; Javorek et al., 2002; Vicens and Bosch, 2000). Pollinating efficiency of wild bees is often higher compared to honey bees (e.g. *Osmia* spp. in apple orchards, Vicens and Bosch, 2000). Unfortunately, there are already examples of the drastic consequences of decreased numbers of pollinators in orchard pollination. In Maoxian County of south-western China farmers apply hand-pollination by ‘‘human pollinators’’ to pollinate apple and other fruit crops to secure yields due to the loss of both wild pollinators and honey bees because of intensive management practices, e.g., intensive pesticide use (Partap and Ya, 2012). After pollinating 100% of apple in the County in 2001, recently, farmers tried to replace apples with plums, walnuts, loquats, and vegetables that do not require pollination by humans. However, hand pollination by human pollinators is still practiced with apples to a lesser degree. The number of bee colonies leased to pollinate the crops is still low, because the communication campaigns about the benefits of bee pollination for higher yield and better quality of Chinese crops are still yet to be done with a focus on major provinces, to improve awareness.

The within-orchard management has strong impact on the pollinator assemblages through both chemical and mechanical practices. The control of vegetation in the undergrowth by herbicides and/or mechanical means eliminated native flowers. However, undergrowth flowers are highly beneficial for insect pollinators through diversity of food resources that is important for flower visitor health (Alaux et al., 2010a), stability of pollinator assemblages (Ebeling et al., 2008), and they can even mitigate negative effects of land management and/or isolation from natural land cover types (Carvalheiro et al., 2011, 2012). Formerly it was recommended to remove the ground vegetation to avoid potential competition with fruit trees for pollinators (Somerville, 1999), however other studies emphasised the strong positive effects of additional flower resources on bee abundances, for example within cherry and almond orchards (Holzschuh et al., 2012; Saunders et al., 2013).

The heterogeneity of surrounding landscape around the orchards has great influence on pollinator assemblages and pollination efficiency of fruit trees within the orchards (Schüepp et al., 2014). The distance at which beneficial foraging and nesting resources out of the orchards may have a positive effect on the within-orchard assemblages depends on the flight and foraging distances of the pollinators. In the case of solitary bees maximum foraging range is between 150 and 600 m (Gathmann and Tscharntke, 2002), while Holzschuh et al. (2012) found increased wild bee visitation of cherry with the proportion of high-diversity bee habitats in the surrounding landscape in 1 km radius. Fruit set of almond was higher with increasing percentage of natural land cover types surrounding the orchards (Klein et al., 2012). In intensive orchard regions, however, orchard-dominated landscapes can drastically reduce wild bee species richness and abundance in the orchard compared to landscapes dominated by either grassland or forest (Marini et al., 2012).

2.2.2.1.10 Greenhouses

Greenhouse production increased worldwide over the past three decades (Pardossi et al., 2004). In China alone there are 2.7 million ha, in South Korea 57 thousand ha of greenhouses (University of Arizona Board of Regents 2012), and there are large areas of greenhouses also in the Mediterranean region, such as Spain, Turkey, Italy, Southern France, Israel and Greece (Jouet, 2001). Production of some greenhouse crops (e.g. tomatoes, melons, strawberries and beans) depends on insect pollination. Greenhouses can be closed systems with only introduced managed pollinators, or semi-open, which allows wild pollinators and managed pollinators from outside to enter. Bees and flies are among the most important pollinators, and honey bees and bumble bees are also commercially used for greenhouse pollination (James and Pitts-Singer, 2008). In the tropics stingless bees are used effectively for greenhouse crop pollination (see details in section 2.4.2.3). Moving of pollinator species and introduction for example of non-native bumble bee species into other continents for greenhouse crop pollination, however, caused severe problems, e.g. pathogen transfer between managed and wild bees (see section 2.4). More details on the importance of bumble bees in greenhouse crop production can be found in sections 2.4.2.2 and 2.5.5 and in Chapter 3.

Pollinating insects have to face several special circumstances in this artificial environment, influencing their fitness, reproduction and pollination efficiency. Plastic films that are used to cover greenhouses often reduce UV-transmission to reduce population levels of harmful insects, but can have an adverse effect on bee behaviour and orientation (Peitsch et al., 1992). The level of carbon dioxide (CO_2) is artificially increased in modern greenhouses to stimulate the growth of plants, but this increased CO_2 level could have a negative effect on the activity and development of bumble bee colonies placed close to the outlets of CO_2 (van Doorn, 2006). Bumble bees stop visiting flowers at higher temperature, which could reach sometimes around 40°C in greenhouses (see overview in James and Pitts-Singer, 2008).

2.2.2.2 Grasslands, shrublands and forests

2.2.2.2.1 Grazing and mowing management

Grazing livestock (e.g. cattle, sheep) alters ecosystems through selective vegetation consumption, soil enrichment by faeces, and soil compaction by trampling. These alterations affect plant production and the amount of floral and nesting resources available to pollinators, thus influencing their abundance or diversity (Kearns et al., 1998; Mayer, 2004). While some studies identified a positive effect of grazing on the overall pollinator diversity in Mediterranean, cold steppes and temperate forests (Vanbergen et al., 2014; Vulliamy et al., 2006; Yoshihara et al., 2008), no effect or a negative effect was found in temperate Andean forests (Vazquez and Simberloff, 2004) and strong negative effects were identified on pollinator richness in the Argentinean Pampas (Medan et al., 2011) and the US Pacific Northwest grasslands (Kimoto et al., 2012). A study on a steppe in eastern Mongolia shows that overgrazing weakens ecological function through the impoverishment of forbs and consequent pollination over a wide area, and by unexpectedly weakening the flower–pollinator network (Yoshira et al., 2008). The precise outcome of livestock grazing for pollinators and pollination likely depends on the land cover type, pool of plant species in the community as well as the grazing intensity, selectivity, timing, land-use history and climate (Asner et al., 2004). A recent experimental study (Kimoto et al., 2012) showed that the timing of grazing impacts bumble bee and other bee pollinator diversity, abundance and richness differently; grazing in the early season appeared to affect bumble bees more strongly than other bees (Kimoto et al., 2012) and grazing at flowering stage may have negative effects on the pollination process.

Grasslands, especially semi-natural ones in Europe, are endangered by overgrazing and mowing (OECD 2004). In northern Germany, changing grazing regimes alter plant-pollinator communities, leading to fewer pollinator species (Kruess and Tscharntke, 2002). Modern livestock farming in UK grasslands, for example, is characterized by high fertilizer application rates, frequent intensive grazing or cutting for silage to optimize harvested forage quality, resulting in low pollinator diversity and structurally homogenous, short vegetation (sward) (Potts et al., 2009). Overgrazing results in less efficient pollination of wild plants (McKechnie and Sargent, 2013). In contrast, careful grazing management can be beneficial for biodiversity in some places that have traditionally been grazed by native large herbivores (Fuhlendorf and Engle, 2001). Productive grasslands with an extensive grazing history peak in plant diversity when they are moderately grazed (Cingolani et al., 2005).

Leguminous species are major pollen resources for bumble bees, and the loss of leguminous species has been associated with reduced bumble bee colony densities at the local to regional scale (Goulson et al., 2005). Loss of leguminous species is partly due to the switch to silage as winter fodder for cattle, and consequent early cut of silage before blooming of leguminous herbaceous flowers (Goulson et al., 2005; Osborne et al., 1991). The impact of silage has been noted by traditional beekeepers from the Cevennes National Park in France, who are concerned that this agricultural practice is currently being promoted even though it deprives bees of nutritional resources (Clement, 2015).

Mowing can have a significant impact on pollinating insects through direct mortality, particularly for egg and larval stages that cannot avoid the mower (Di Giulio et al., 2001). Mortality due to mowing when eggs and larvae are present is a threat to the persistence of some butterfly species (Thomas, 1984; Wynhoff, 1998). Mowing can also disturb ant nests, which in turn affects the survival of butterflies that rely on particular ant species (their final instar larvae feed in the ant nests) (Wynhoff et al., 2011). Caterpillars on the ground as well as caterpillars on vegetation are vulnerable to direct mortality by mower (Humbert et al., 2010).

Mowing also creates a sward of uniform height and may destroy topographical features such as grass tussocks (Morris 2000) when care is not taken to avoid these features or the mower height is too low. Such features provide structural diversity and offer potential nesting sites for pollinator insects such as bumble bees (Hopwood et al., 2015). In addition to direct mortality and structural changes, mowing can result in a sudden removal of almost all floral resources for foraging pollinators and butterfly host plants (Johst et al., 2006). The reduction in host plants and foraging resources can reduce pollinator reproduction and survivorship (Boggs and Freeman, 2005), and pollinators will likely be forced to seek alternative habitat. Skórka et al. (2013) found that butterfly roadkill in Poland increased as mowing frequency increased; adult butterflies that dispersed to find new habitat after roadsides were mowed were more likely to collide with vehicles.

The frequency and timing of mowing influence the composition of vegetation over time (Forrester et al., 2005), thus indirectly influencing pollinator diversity and abundance. Frequent mowing during a growing season reduces native plant growth and the ability of forbs to compete with grasses. Excessive roadside mowing may have led to a decrease in flowers and a subsequent decrease in bumble bees in Belgium (Rasmont et al., 2006). Intensively-mowed roadsides generally have the shortest vegetation and lowest amount of nectar, which together result in decreased butterfly abundance (Gerell 1997; Saarinen et al., 2005). However, carefully timed roadside mowing can have positive effects on plant diversity (Parr and Way, 1988) that in turn benefit pollinators (e.g., Noordijk et al., 2009).

Mowing technique can have a great influence on the effects on pollinators. Frick and Flury (2001) estimated losses from rotary mowers as between 9,000 and 24,000 bees per hectare in flowering white clover fields and 90,000 per hectare in flowering *Phacelia*. Mowing without a conditioner, which processes hay so it dries more quickly, reduced the mortality by a factor of seven. In order to avoid significant bee losses, the researchers recommend refraining from mowing in periods of increased flight activity. Humbert et al. (2010) analysed the direct impact on invertebrates of different hay harvesting processes. The use of a conditioner reduced the survival rate of orthopterans from 32% to 18%. Leaving uncut refuges and delaying mowing mitigate the impact on pollinators (Buri et al., 2012; Humbert et al., 2012). Although there is no evidence about the effect of mowing mortality on local pollinator population dynamics and its impact on pollination, studies suggest mowing can have a negative impact.

2.2.2.2.2 Logging

Tree removal leads to alteration in the albedo (fraction of solar energy reflected back from earth), light regime, soil dynamics, hydrology, soil chemistry and plant composition (Foley et al., 2005), with profound effects on ecosystem structure. It is therefore expected that pollinators will also be affected by logging. Studies on logging indicate that the pollinator group and the biome play a role in the response of pollinators to logging disturbances. In tropical forests, forest fragmentation associated with logging leads to a rapid reduction in butterfly diversity and abundance (Daily and Ehrlich, 1995). In contrast, while selective logging negatively affects stingless bees (Eltz et al., 2002; Samejima et al., 2004), it can maintain the presence of some butterfly groups, at least if logging is associated with maintenance of land cover heterogeneity within the logged patch (Hamer et al., 2003; Lewis 2001). In the Western Amazon pollen deposition rate of some hardwood tree species was reduced, others were increased, while some species were unaffected at logged sites compared to non-logged forest (Maues et al., 2007). Moth diversity and abundance increased with levels of disturbance in montane rainforests (Axmacher et al., 2004), a result that agrees with works on several types of insect pollinators in temperate and boreal forests (Jackson et al., 2014; Pengelly and Cartar 2010; Romey et al., 2007). In the boreal forest of Canada there were generally more bumble bees, species of bumble bee-visited plants, and flowers in moderately (50–75% of trees remaining) logged sites, but logging affected the distribution of bumble bees across floral resources, with too many bumble bees in the flower-poor compartments and too few in the flower-rich ones than merited based on the quantity of flower resources (Cartar, 2005). Controlling for flower density, bumble bee density was significantly greater in clearcuts than in the highly (10-20% of remaining trees) or moderately logged (50–75% of trees remaining) plots. By disproportionately visiting plants in clearcuts (relative to flower density) bumble bees in clearcuts should experience higher levels of competition. Forests experiencing different levels of disturbance were also shown to harbour different plant and insect species, thus plant-pollinator networks also show different characteristics (Nielsen and Totland, 2014).

2.2.2.2.3 Fire

Fire is often used as a management tool for agricultural conversion and prescribed burning is used as a forest management strategy to suppress fires and improve land cover types in many regions of the world. These burnings have been shown to benefit the diversity of Lepidoptera in the Western US coniferous forests (Huntzinger, 2003), species richness of Hymenoptera and Lepidoptera in forest from the Southern Alps (Moretti et al., 2004), and species richness in central European forests (Bogusch et al., 2014). Fires in Mediterranean oak-pine forests lead to an initial strong reduction of bee diversity in recently burnt areas, with a recovery in the following years, which has been shown to be highly correlated to floral diversity (Potts et al., 2003).

Fire considerably changes vegetation and land cover conditions, and therefore can have an important effect on pollinators and plant pollination, which may be detrimental (e.g., Ne’eman et al., 2000; Panzer, 2002). Burns during the growing season remove floral resources, host plants, and nesting materials, and can be detrimental to species with life stages that cannot fly to safety at the time of the burn (Hopwood et al., 2015). Burns during the dormant season can kill overwintering pollinators such as butterflies, moths, syrphid flies, and soldier beetles that overwinter at the base of plants, in leaf litter, or underneath the surface of the soil (Hopwood et al. 2015). A recent study on prescribed burning and the imperiled mardon skipper (*Polites mardon*) in California showed substantially fewer butterflies in the burned areas of meadows compared to unburned areas after 1, 2, 3 and 5 years following the burn event (Black et al., 2014). Queen bumble bees overwintering in small cavities just below or on the ground surface are at risk, as are ground-nesting bee species that nest in shallow burrows (Cane and Neff, 2011). Solitary bees nesting in stems or twigs are unlikely to survive the heat of burns (Cane and Neff, 2011), and stem-nesting bee populations will only recover postfire when the availability of suitable stems increases over time (Potts et al., 2005). The loss of bees due to a burn can lead to reduced fruit set in plants in burned areas (Ne’eman et al., 2000).

Recovery of pollinators following a burn varies between guilds. Though losses of bees following a fire can be catastrophic, bees may be able to recolonize burned sites and recover within a few years (Potts et al., 2003). Habitat-dependent or -specialist species and those that are less mobile are most likely to be negatively affected immediately by a fire (Panzer 2002; Vogel et al., 2010). A pollinator species’ ability to cope with regular burns is dependent on there being adequate unburned adjacent areas that can provide sources of colonizers into the burned land cover type (e.g., Harper et al., 2000; Hartley et al., 2007; Panzer 2002;

Swengel 2001). Isolated populations of pollinators in small fragments may not survive repeated prescribed burns (Panzer 2002) because there are often no source populations available for recolonization once a population has been locally extirpated. Burning a small fragment in its entirety could risk eliminating some species because of limited recolonization from adjacent patches (Harper et al., 2000). This accentuates the need to leave substantial land cover patches when using fire as a management tool. Land cover patches should not be burned completely; rather, a mosaic of burned and unburned areas is ideal. Besides controlled grazing and mowing, prairies (ecosystems considered part of the temperate grasslands, savannas, and shrublands biome, typically in North America) can be managed through prescribed burning. A large experimental study demonstrated that different butterfly species have varied responses to prairie management through fire. While prairie specialists responded negatively to burning, generalists were largely benefited by this action (Swengel 2001). Moreover, greater durations without burning benefited specialists but reduced generalists (Swengel 1996). However, there may be some geographic variation in these results, as it has been shown that burns in oak savannas in the USA do not harm butterfly diversity (Siemann et al., 1997).

Fire can have significant, negative impact on plant reproductive success and is associated with statistically significant lower fruit set (McKechnie and Sargent, 2013). In the Cape Floristic Region of South Africa, nectar-feeding bird abundance and species richness was found to decrease in post-fire vegetation, and floral arrays within burnt vegetation received no visits by nectar-feeding birds (Geerts et al., 2012). Some studies, however, have shown that fire-dependent communities have indirectly and positively impacted pollinators by altering plant density and distribution (Van Nuland et al., 2013, Charnley and Hummel, 2011). Moreover, fires in Mediterranean climates are necessary for seed dispersal and germination (Pausas and Vallejo, 1999).

2.2.2.2.4 Transformation of agroforestry systems

Agroforestry refers to the practice of integrating trees and other large woody perennials into farming systems and throughout the agricultural landscape (Schroth et al., 2004). While a considerable number of papers show the positive effects of plant diversity in agroecosystems for bees and other insect pollinators (see Nicholls and Altieri, 2013, for a review), considerably less attention has been paid to understand the effects of agroforestry for bees and other pollinators. Willemen et al. (2013) revealed a high diversity of Tree-Based Ecosystem Approaches, including trees in croplands, trees in grasslands, forest-based systems, complex multi-strata agroforestry and homegardens. They report positive impacts for food security and climate change, but very few of these studies evaluated the impacts of these systems for pollinators.

Studies in temperate landscapes are particularly infrequent, although agroforestry has been flagged as a practice favourable to beekeeping (Hill and Webster, 1995). In Québec, Alam et al. (2014) estimated the value of ten ecosystem services in an agroforestry system (tree-based intercropping), in particular the value of pollinators, and found that yield and profit could be maximized with the presence of tree and shrub cover in agricultural landscapes.

Instead, the relatively few field studies on this topic have been performed in tropical landscapes, where agroforestry systems are the major agroecosystems that resemble natural forest, and potentially have high biodiversity and pollinator conservation value (Tscharntke et al., 2011). Agroforestry systems are a land use that might aid in enhancing connectivity between natural and semi-natural areas (Perfecto and Vandermeer, 2008). In the tropics, agroforestry may perhaps be one of the most important land management systems for pollinator conservation, because the majority of trees are animal pollinated and pollinators therefore may rely more on floral resources from trees compared to herbaceous wild flowers (Bawa, 1990).

The effects of agroforestry practices on the diversity of pollinators and pollination have been studied principally for two tropical crops, coffee and cacao, and show overall the positive effect of integrating agricultural landscapes with biodiversity conservation (Harvey et al., 2008). In the case of coffee, a shrub that benefits from shade from canopy trees, Ricketts (2004) showed in Costa Rica that the diversity of bees on coffee flowers decreased with distance to forest, where bees nest; this way, the forests increased coffee yields by 20%, due to pollination provided by bees. In Indonesia, Klein et al. (2002) found similar results, and in Mexico, Jha and Vandermeer (2010) showed the importance of in-farm tree diversity management, whereas Vergara and Badano (2009) established a link between diversity of bees and crop pollination in low-impact management systems in coffee plantation. Pollinator richness and abundance respond positively to increased species richness of shade trees, blossom cover of non-coffee flowering plants (Klein et al., 2003b), and increased canopy cover (Jha and Vandermeer, 2010). Recently, Bravo-Monroy et al. (2015) showed that forested landscape close to coffee farms appears to increase stability and resilience to the pollinating bees and insects. However, research is still needed to determine the relative effects of management interventions, as, for example, irrigation and addition of lime had more substantial positive effects on coffee production than tree cover (Boreux et al., 2013).

There are fewer studies on cacao crops, though Groenveld et al. (2010) showed experimentally that pollen limitation greatly reduces yields in Indonesia, indicating that practices that could increase the midge pollinator populations could have large impacts on yield and farmer income. Further, Hoehn et al. (2010) found in Indonesia that agroforestry systems increased bee species richness, especially on a regional scale due to high diversity in types of management.

2.2.2.3 Urban management

Given that urban areas are increasing globally (Seto et al., 2012), it is important to understand the effects of urbanization on pollinator communities. Urban areas are characterized by high heterogeneity, with fine-scale land cover variation (Cadenasso et al., 2007). Urban greenspaces can include private and public gardens, parklands, brownfield sites (land previously used for industrial purposes or some commercial uses), cemeteries and churchyards, green roofs and small-scale agroecosystems such as community or allotment gardens, market gardens, or urban farms (see Sadler et al., 2010).

Pollinators provide important pollination to urban flowers and crops (Lowenstein et al., 2014; Matteson and Langellotto, 2009; Potter and LeBuhn, 2015), and urban gardens on the rural-urban interface have the potential to provide pollination for neighbouring rural areas (Pereira-Peixoto et al., 2014). Little is known about pollinator efficiency of crops or wild plants in urban areas. Leong et al. (2014) suggest that reduced seed set in urban areas relative to natural areas could be due to reduced pollinator efficiency caused by higher plant species richness in urban areas, although Williams and Winfree (2013) found pollination in woodlands to be unrelated to the degree of urbanization along an urban-rural gradient.

The response of pollinators to urbanization is likely to be dependent on urban context, i.e. geographic location, surrounding landscape (agricultural vs. natural vs. semi-natural), size of the town or city and patterns of development (Wojcik, 2012) as well as local policies relating to green urban areas and the life history characteristics of different pollinator taxa, i.e. dispersal ability, reproductive strategy and foraging requirements. Studies have shown both positive and negative impacts of urbanization on pollinators, although it is difficult to ascertain properly the effect of urbanization as few studies have compared replicate urban and non-urban areas.

Studies conducted at a regional or local level have shown that urban areas can support species-rich pollinator communities relative to the regional (e.g., Fetridge et al., 2008) or national species pool (e.g., Owen, 2010; Saure, 1996) and that bee species richness (McIntyre and Hostetler, 2001; Sattler et al., 2011; Sirohi et al., 2015), bee abundance (McFrederick and LeBuhn, 2006; Winfree et al., 2007; Carper et al., 2014; Sirohi et al., 2015) and butterfly species richness (Restrepo and Halffter, 2013) are higher in urban or suburban sites compared to surrounding areas. Furthermore, experimental studies have shown that bumble bee colony growth rate and nest density in UK suburban gardens can exceed that found in the countryside (Goulson et al., 2002; Osborne et al., 2008).

In contrast, other studies show a decrease in the species richness of pollinating insects, including bees, hoverflies and butterflies, with increased urbanization (Ahrne et al., 2009; Bates et al., 2011; Clark et al., 2007; Hernandez et al., 2009). Bates et al. (2011) found decreased bee and hoverfly abundance with increased urbanization and Deguines et al. (2012) found urbanization to be the most detrimental land-use change for flower visitor communities in a country-wide study in France. Urbanization can also influence pollinator nesting opportunities; Jha and Kremen (2013b) found a negative effect of paved areas on bumble bee nesting density. Furthermore, urbanization might restrict gene flow for some pollinators. Jha and Kremen (2013a) found that impervious land use in urban areas negatively affects regional bumble bee gene flow.

The effect of urbanization can vary among taxa. For example, bee guilds may differ in their ability to adapt to urban environments; floral specialists are rare in cities (Frankie et al., 2009; Hernandez et al., 2009; Tonietto et al., 2011), whilst other studies have shown a positive effect of urbanization on bumble bees (Carré et al., 2009), cavity-nesting bees (Cane et al., 2006; Matteson and Langellotto 2009) and later-season small-bodied bees (Banaszak-Cibicka and Zmihorski, 2011; Wray et al., 2014). Hoverflies appear to be more negatively affected by urban development than bees (Baldock et al., 2015; Geslin et al., 2013; Verboven et al., 2014). Baldock et al. (2015) simultaneously sampled flower-visitor networks in triplets of urban, agricultural and natural sites located in and around 12 UK towns and cities. Sites were carefully selected to be representative of those land use types within each region. The study found no difference in overall flower-visitor abundance or species richness among the three land-use types. Bee species richness, however, was higher in cities compared to farmland, although there was no difference in abundance among landscapes. In contrast, fly abundance was higher in farmland and nature reserve sites, although species richness of these groups did not differ among land use types. In France, data from a citizen science monitoring scheme using photographs of insects on flowers showed that although most flower visitors had a negative affinity with urban areas and a positive affinity with agricultural and natural areas, hymenopterans (including bees) appeared tolerant of a range of landscapes (Deguines et al., 2012).

Positive effects of urbanization on pollinators are likely obtained through increased land cover diversity and heterogeneity in urban areas compared with some agricultural and natural land cover types (McKinney, 2008; Sattler et al., 2010). Further, built structures, ex-industrial areas, disturbed and gravelled surfaces and warm microclimates may create nesting opportunities rare in more thickly vegetated terrain. Although pollinator data are lacking from such land uses, Kattwinkel et al. (2011) suggest that brownfield sites can be important for the conservation of other taxa, including plants and insects. Urban areas could also provide a refuge from the impacts of insecticides applied in croplands, although neonicotinoid insecticide use in urban lawns has the potential to have a detrimental effect on bumble bee colony growth and new queen production if applied to blooming plants (Larson et al., 2013), and a study using citizen science data from French gardens found a negative correlation between butterfly and bumble bee abundance and use of insecticides and herbicides (Muratet and Fontaine, 2015). Muratet and Fontaine (2015) also found that the negative effect of insecticides was stronger in highly urbanised areas. Floral abundance and richness appear to play an important role in pollinator diversity. Studies have shown increases in plant species richness in urban areas compared to surrounding agricultural, semi-natural and natural areas, due to the large number of non-native species, longer flowering seasons which provide continuity of floral resources over a longer period and the high heterogeneity of urban areas (Angold et al., 2006; Hope et al., 2003; Kuhn et al., 2004; Neil and Wu, 2006; Stelzer et al., 2010).Urban areas that provide high levels of floral resources can support more flower-visiting insects (Matteson et al., 2013). However, the importance of floral resources may not hold for all pollinator taxa or across all areas (e.g., Neame et al., 2013; Wojcik and McBride, 2012).

There are comparatively fewer studies of pollinators in urban areas than in agricultural or natural landscapes and many knowledge gaps exist, particularly regarding beneficial urban management approaches for pollinators (but see Blackmore and Goulson, 2014; Garbuzov et al., 2015). Although studies are emerging in neotropical cities (e.g. Aleixo et al., 2014; Frankie et al., 2013; Nates-Parra et al., 2006; Zotarelli et al., 2014), there remains a research bias towards northern temperate cities. Thus it is difficult to apply the findings from many current studies to tropical and arid countries, many of which are experiencing rapid growth in urban development. As for all landscapes, the lack of standardised long-term data on pollinators from urban areas makes it difficult to infer anything about the long-term effect of urbanization on pollinators.

2.2.3 Conclusions

Land use is currently the main driver of land cover change, leading to changes in land cover composition and configuration. It is well established that habitat loss and degradation, as well as loss of connectivity, reduction in patch sizes, and fragmentation negatively affect pollinator diversity, abundance and richness. These changes can negatively affect community stability, pollination networks and the survival and evolutionary potential of pollinator and plant species. Finally, these changes also result in a reduction of plant fruit set, which is of critical importance for food security, ecosystem services and human welfare in wild and agricultural environments.

Land management alters most ecosystems, having considerable impact on pollinator communities, and crop and wild plant pollination. Large-scale, chemically-intensive agricultural systems that simplify the agroecosystem through specialization on one or several crops are among the most serious threats to natural and managed ecosystems. Agricultural management practices such as increased fertiliser use, intensive tillage systems, heavy use of pesticides, high grazing/mowing intensity or badly-timed management actions decrease pollinator diversity dramatically, while influencing and reducing the effectiveness of ecological functions and services, like pollination.

Large monoculture systems reduce both foraging and nesting resources for pollinators by removing flowering weeds and native plants and reducing crop diversity, and decreasing availability of undisturbed soil patches, hollow stems, shrubs, trees and dead wood that are needed for nesting sites. While certain mass-flowering crops provide large amounts of foraging resources (i.e. nectar and/or pollen) for some pollinators, these pulsed resources provide only temporary benefits that cannot sustain most pollinators throughout their life cycle.

Creating a more diversified agricultural landscape based on principles from sustainable agriculture, agroecology and organic farming management (i.e. intercropping, polyculture, crop rotations, cover-cropping, fallowing, agroforestry, insectary strips and hedgerows), has the potential to maintain rich pollinator communities, promote connectivity, and increase pollination of crops and wild plants, as well as improve livelihoods for smallholder farmers that make up the majority of the farming community and provide an estimated 50 – 70% of the world’s food (Altieri et al., 2012; Herrero et al., 2010). However, concerns have been raised as to whether such techniques can be equally productive. Existing evidence suggests that organic farming methods are on average 10 – 25% less productive than conventional farming methods (established; Badgley et al., 2007; de Ponti et al., 2012; Seufert et al., 2012; Ponisio et al., 2015), although these yield gaps are reduced to 5 – 9% in organic farming that takes full advantage of diversification practices (intercropping and crop rotations) (Ponisio et al., 2015). Although organic farming suffers relatively small yield gaps, these yield gaps are balanced by enhancements that they provide to multiple aspects of sustainability (Kremen and Miles, 2012). A meta-analysis by Crowder and Reganold (2015) showed first, that organic systems with price premiums were significantly more profitable (22–35%) and had higher benefit/cost ratios (20–24%) than conventional agriculture, and second, that price premiums were far higher than necessary to establish equal profitability with conventional systems. Given their multiple sustainability benefits, these results suggest that organic farming systems could contribute a larger share in feeding the world at a lower price premium. A major gap in our understanding is how to reduce yield gaps in these more sustainable systems. Research, extension and infrastructure investment in sustainable agriculture, agroecology and organic farming management methods has been orders of magnitude less than in conventional scale agriculture (Ponisio et al., 2015; Carlisle and Miles, 2013), suggesting that increased investment in these techniques could lead to greater yields and profits, and to broader adoption (Parmentier, 2014). The lack of sustainability of monoculture systems that are highly dependent on chemical inputs, however, indicates the urgent priority for improving the productivity of more sustainable systems that will also promote pollinators.

Specifically, diversified farming systems are beneficial for biodiversity and ecosystem services, including pollinators and crop pollination. Provision of different crops and crop varieties not only benefits pollinators but also increases crop genetic diversity, potentially enhancing pollination. Maintenance of diverse wild plant communities within the crop fields and orchards provides a high variety of foraging resources before and after the crop flowering period that supports wild and managed bee health, and increases wild pollinator diversity and abundance on these fields with positive effects on crop pollination. Within-field diversification and application of less intensive management practices, will be more effective if wild flower patches and a diverse landscape structure is available nearby or around the managed sites. Furthermore, the conservation of pollinator habitat can enhance overall biodiversity and other ecosystem services such as biological pest control, soil and water quality protection (Kremen et al., 2012; Kremen and Miles 2012), and these secondary benefits should be incorporated into decision making (Wratten et al., 2012).

Traditional landscapes maintain wild flower patches that are often threatened by abandonment of these management practices, especially in remote sites. Cessation of management, such as grazing, mowing on grasslands, leads to vegetation succession that can have considerable negative consequences on the pollinator fauna. Therefore, maintenance of ecosystem healthy and optimal management at such valuable, traditionally managed systems is highly beneficial.

2.3 Pesticides, GMOs, veterinary medicines and pollutants

2.3.1 Pesticides

Pesticides (fungicides, herbicides, insecticides, acaricides, etc. (see Box 2.3.1)) are primarily used in crop and plant protection against a range of pests and diseases and include synthetic chemicals, biologicals, e.g., *Bacillus thuringiensis* (*Bt*) or other chemicals of biological origin such as spider venom peptides (Windley et al., 2012). Pesticides also play a key role in public health, for example in the control of disease vectors such as mosquitoes, e.g. application of larvacides, adulticides and use of treated bednets (Casida, 2012). Broad-spectrum insecticides, which are generally seen as higher risk to pollinators, are used on agricultural areas, in urban environments such as gardens, parks and golf-courses, and in controlling nuisance insects and disease vectors such as mosquitos (Goulds, 2012). Some pesticides, particularly insecticides, and especially when not used in accordance with effective risk management/mitigation to reduce/remove exposure, for example using them only outside the flowering period in bee-attractive crops, have the potential to affect pollinator abundance and diversity directly by causing mortality. Sublethal effects, such as impaired foraging ability or reduced immune function, may affect pollinator populations (Rundlöf et al., 2015). A recent experiment suggests that sub-lethal exposure in the laboratory can adversely impact on subsequent pollination provision to apple (Stanley et al., 2015), although there is no evidence to date of impacts on pollination under field conditions resulting from sublethal effects (Brittain and Potts, 2011). The role of sub-lethal effects of pesticides, particularly the neonicotinoid group, as a driver of pollinator decline has undergone increasing scrutiny (Blacquière et al., 2012; Van der Sluijs et al., 2013; Godfray et al., 2014). This scrutiny is in part caused by their high level of use combined with their high toxicity and systemic properties resulting in the potential for exposure to pollinators. Despite this high level of scrutiny, some knowledge gaps remain (Blacquière et al., 2012; Godfray et al., 2014; Lundin et al., 2015) which, combined with sometimes conflicting research results, has led to a much polarised debate.

Box 2.3.1: Types of pesticides

"Pesticides" is a collective term for a range of synthetic and natural chemical plant (crop) protection products. They are broadly classified into three main groups: herbicides for weed control, fungicides for fungal disease control, and insecticides for insect pest control. Other classes of pesticides include plant growth regulators, acaracides and molluscicides, and in some countries, varroacides for controlling honey bee *Varroa* parasites are classed as pesticides.

The insecticides include a wide range of chemistries with differing modes of action but the main chemical classes often referred to are:

Organophosphorus and carbamate insecticides: these inhibit the acetylcholinesterase enzyme that terminates the action of the excitatory neurotransmitter acetylcholine at nerve synapses. Globally the use of these insecticides is declining.

Organochlorines and pyrethroid insecticides: These are sodium channel modulators that keep sodium channels in neurones open causing hyperexcitation and in some cases nerve block. Sodium channels are involved in the propagation of action potentials along nerve axons. Organochlorine insecticides are no longer widely used; the use of pyrethroids is stable.

Neonicotinoid insecticides: These are nicotinic acetylcholine receptor (nAChR) agonists mimicking the action of acetylcholine at nAChR and resulting in hyperexcitation. The use of neonicotinoids has increased globally since their introduction in the 1990s.

For more information, see Table 2.3.2

Box 2.3.2: **Pesticide Risk "in a nutshell"**

The risk posed by pesticides is driven 1) by the toxicity (hazard) of the chemical, e.g. as the active substance, metabolite, or formulation, which has a fixed value to a given developmental stage of the species under specified conditions, and 2) by the level and duration of exposure of the pollinator, which is highly variable depending on its behaviour, e.g. bee foragers versus in-hive adults versus larvae. Toxicity can be measured as lethality (i.e. median Lethal Dose (LD_{50}) or median Lethal Concentration (LC_{50}) which is the amount of the substance required to kill 50% of the test population) or as sublethal effects, e.g. memory impairment, reduced foraging, reduced brood production, etc., measured by an effective dose (EC_x) or No Observed Effect Level or Concentration (NOEL or NOEC). The challenge is to understand the magnitude and duration of effects on the individual, colony and/or community of pollinators.

Pesticide exposure varies according to pesticide use and the properties of the component chemicals, the behaviour and biology of the pollinator species and the local environment. Exposure may be to one, or a combination of pesticides, which can be directly applied sequentially or in combination, e.g., tank mix, to a wide range of crops visited by pollinators or through pollinators foraging on non-crop floral resources (e.g. wild flowers) that have also been exposed to pesticides. The behaviour of the pollinator may affect exposure, for instance depending upon whether they forage on a single or limited number of plants, store pollen and/or nectar, collect water, use plant material or soil to construct their nests, or are active at particular times of year. The environment may affect exposure and includes factors such as the size of fields, cropping management and availability of alternative untreated forage, e.g., flowering headlands.

Risk is typically estimated by examining the ratio of exposure to effects. At a deterministic level point estimates of exposure and effects are used, whereas probabilistic risk estimation methods consider the distribution of exposure and effect endpoints and are considered better suited to estimating the likelihood and magnitude of an adverse effect. The Hazard Quotient (HQ) approach is deterministic and can be used for foliar-applied pesticides and is defined as the application rate (grams of active ingredient applied per hectare) divided by the acute contact or oral LD_{50} (whichever is the lower) (EPPO, 2010). Similar approaches are taken for systemic pesticides where the exposure (e.g. via ingestion of contaminated nectar and pollen) is compared to the LC_{50} or NOEL. Additionally, where potential risks are identified, further refinement options are available to understand the potential risk under field conditions, e.g. semi-field and field studies (see also Chapter 6).

The challenge, and areas of greatest debate, are to understand:

- the magnitude and duration of direct sublethal effects on pollinator populations from exposure to pesticides at levels found in the field under typical use conditions; and
- whether honey bees are a suitable surrogate for other pollinator species in risk assessment, e.g. due to differences in physiology, ecology and behavior.

2.3.1.1 Pesticide use

Globally, pesticide use on agricultural land varies according to the regional or local pest and disease pressures (FAOSTAT, 2014) as well as factors such as the purchasing capacity and cultural practices of the farmers (Schreinemachers and Tipraqsa, 2012; Heong et al., 2013; Heong et al., 2014). Although in many countries for which data are available (e.g., in the USA, Brazil and Europe) the total tonnage of pesticides used in agriculture is stable or increasing over time since the 1990s (OECD, 2013; FAOSTAT, 2014). For many other countries (e.g., in Africa and Asia) data are incomplete or absent. Some variations in pesticide use are driven by changing agricultural practices, for example, herbicide application in the USA has increased and insecticide tonnage decreased, both associated with the increase in cultivation of genetically modified crops and with changes in efficacy (USDA, 2014).

Where data are available for developing countries pesticide use has been seen to increase rapidly, sometimes against a low base level. However, international consensus over the level of risk posed by some of these pesticides has often not been reflected in reductions in the use of these chemicals in developing countries (Schreinemachers and Tipraqsa, 2012). Thus in many high- and middle-income countries enforced restrictions on the use of organochlorine, organophosphate and carbamate insecticides that pose a high risk to human and environmental health have resulted in their replacement by neonicotinoids and pyrethroids (e.g., see Figure 2.3.1). For example, one of the significant changes in pesticide application methodology in the EU/US over the last 20 – 30 years has been the development of soil- or seed-applied systemic insecticides (e.g., the neonicotinoids) as an alternative to multiple foliar/spray applications (Foster and Dewar, 2013). This class of systemic insecticides is now used on a wide range of different crops/application combinations in field and tree crops including foliar sprays, soil drenches and seed treatments in over 120 countries, accounting for at least 30% of the world insecticide market (Nauen and Jeschke, 2011; Simon-Delso et al., 2015). Their persistence in water and soil, uptake into crops and wild plants and subsequent transfer into pollen and nectar (Krupke et al. 2012, Johnson and Pettis 2014) potentially representing a significant source of exposure, has led to concerns that they pose a unique, chronic sublethal risk to pollinator health (Van der

Sluijs et al., 2013). In contrast, in low- and lower-middle income countries many of the older classes of insecticides are still widely used and excessive use due to lack of user training and stewardship is a significant concern (see Africa case study, Box 2.3.4) (Tomlin, 2009; Schreinemachers and Tipraqsa, 2012; Heong et al., 2013).

Where pesticides are used they should be applied in accordance with integrated pest management practices (IPM, see Chapter 6, sections 6.4.2 and 6.8.2). In this assessment appropriate use of insecticides (which as a class have the greatest potential to direct impact pollinators) has been defined as also including, but not limited to, ensuring mitigation has been identified to minimise exposure of pollinators, that the label provides clear instructions on how to protect pollinators and the applicators are aware of, and follow, the label instructions (for example see the FAO Code of Conduct (FAO, 2013)). Pesticides, when used appropriately and in an integrated pest management program (IPM, see Chapter 6, sections 6.4.2 and 6.8.2) (Korsten, 2004; Mani et al., 2005; Gentz et al., 2010), may be considered an important tool for the sustainable intensification of crop production (Tilman et al., 2002; Godfray et al., 2010; Tilman et al., 2011; Andersons, 2014). Although the range of pesticides available may be limited by market demand in some sectors, e.g. in organic farming (Box 2.3.3), such restrictions do not necessarily equate to reduced risk to pollinators as many of these authorised pesticides are also toxic to pollinators, (e.g., Barbosa et al., 2014). Overlaid on this discussion is that around the world the classes of pesticide authorized, the level of risk assessment/regulation and the scale of education, understanding, implementation and enforcement of responsible and careful use by pesticide end-users vary widely from ineffective regulation and/or enforcement to highly enforced systems such as in the EU and North America (see Appendix of Chapter 6, examples include Ecobichon, 2001; Hordzi et al., 2010; Sahu, 2011; Al Zadjali et al., 2014; Deihimfard et al., 2014; EC, 2014; CropLife, 2015). Regulation of pesticide use may be undertaken directly through environmental risk assessments (see Chapter 6) but also may occur indirectly through ensuring other requirements such as MRLs for human safety (Maximum Residue Limits set by importing countries such as the US, EU, Australia, Japan, Taiwan) are met in crops for export (Sun et al., 2012). Even when data on total pesticide usage are available these rarely provide the detailed information relevant to this assessment, e.g. the potency of different insecticides and their use pattern (application method, rate, crop, area treated and timing), making comparisons based solely on total tonnage sold or value of sales complex and difficult to interpret. Improving the detail of pesticide usage data would significantly enhance our understanding of the potential risks posed to pollinators from pesticide use. Beyond agricultural uses, data are also lacking for use in residential properties with limited data from amenity use (e.g., Goulds, 2012).

Box 2.3.3: ***Pesticides in organic farming***

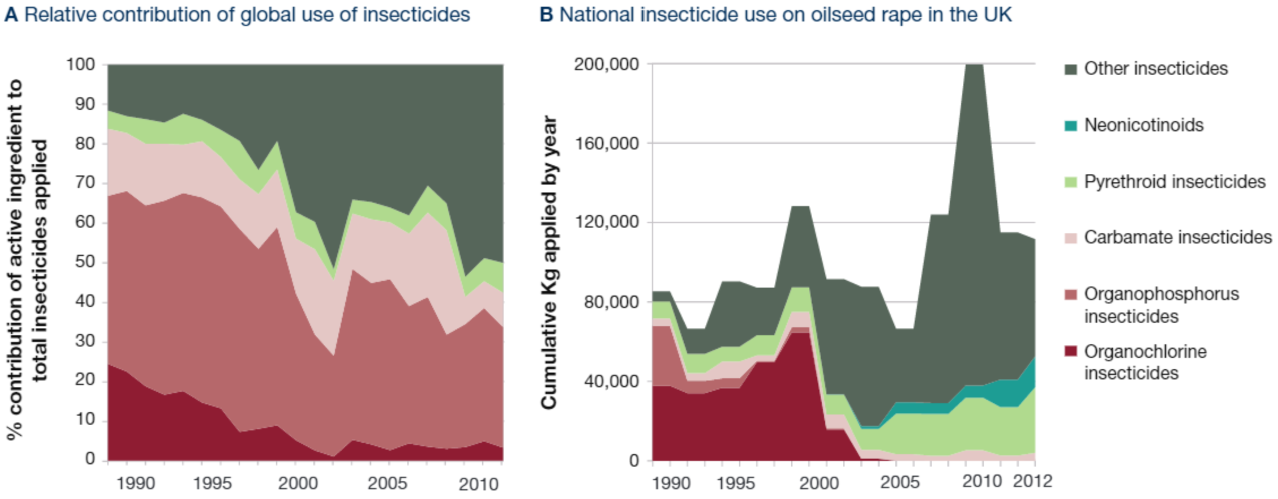
This example is derived from the FAO (http://www.fao.org/docrep/015/an905e/an905e00.pdf; http://www.fao.org/docrep/015/an765e/an765e00.pdf). However, definitions of organic farming vary widely.

Organic crop production uses only pesticides for pest/disease/growth management that are on lists referenced by the relevant international/regional organic standards and also requires that co-formulants (e.g. inerts and synergists) are not carcinogens, mutagens, teratogens or neurotoxins.

If the pesticide is used for plant protection, growth regulation or weed control:

- it must be essential for the control of a harmful organism or a particular disease for which other biological, physical, or plant breeding alternatives and/or other management practices are not effective.
- it has the least harmful impact (compared to alternatives) on the environment, the ecological balance (in particular non-target organisms) and the health of humans, livestock, aquatic animals and bees.
- substances must be of biological or mineral origin and may undergo the following physical (e.g., mechanical, thermal) or biological (e.g., enzymatic or microbial composting or digestion) processes in formulation.
- synthetic substances may be used by exception, such as the use in traps or dispensers, or substances that do not come into direct contact with produce, or those for which no natural or nature-identical alternative are available provided that all other criteria are met.
- use may be restricted to specific target organisms, conditions, specific regions or specific commodities;

Examples of chemicals allowed include preparations/products from Neem *(Azadirachta indica)*, rotenone, spinosad, copper salts (e.g., sulfate, hydroxide, oxychloride, octanoate, cuprous oxide, Bordeaux mixture and Burgundy mixture), fungal and bacterial preparations (e.g., *Bacillus thuringiensis*).



7. Figure 2.3.1: Global use of insecticides
(OECD, 2013), shown as relative contribution to sales (tonnes active ingredient) as data are incomplete across years and countries; data for neonicotinoids are not separately identified in the dataset) and an example of national insecticide use on oilseed rape in the UK [UK Pesticide Usage Survey; total usage data to 2012; areas where less than a total of 100 ha were treated have been excluded; for pesticides in each class see PUS data (FERA/Defra, 2015)].

Total mass applied may be affected by crop area grown, which increased from 415,000 to 615,000 ha between 1996 and 2010 (http://www.ukagriculture.com/statistics/farming_statistics.cfm?strsection=Oilseed%20Rape).

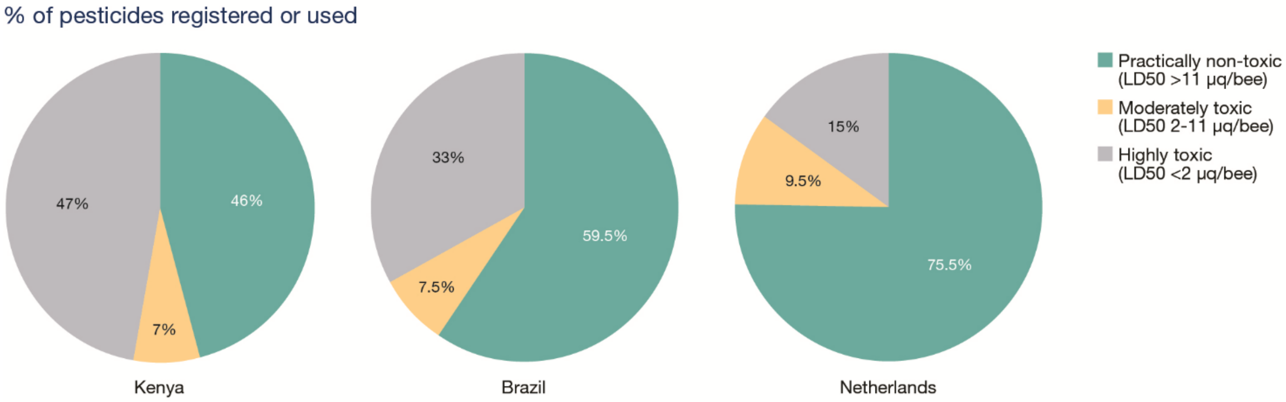
Box 2.3.4: Case Study: Africa

In Africa, there is a high demand for pollination for many crops (Gemmill-Herren et al., 2014). At the same time, pollinators are exposed to similar environmental pressures that have been associated with declines elsewhere in the world, e.g., habitat transformation or fragmentation (Ricketts et al., 2008; Kennedy et al., 2013), loss of diversity and abundance of floral resources (Biesmeijer et al. 2006; Carvell et al. 2006; Kremen et al., 2007), inappropriate use of pesticides (Pettis et al., 2013), spread of pests and diseases (Aebi and Neumann, 2011; Cameron et al., 2011b), and climate change (Schweiger et al., 2010). Despite the economic importance of insect-pollinated crops (Garibaldi et al., 2013) data on the pattern and amount of pesticide use in Africa are also very difficult to obtain and almost impossible to estimate for any single African country due to a lack of detailed lists of imports into these countries (Youm et al., 1990).

The environmental impact of pesticides on pollinators has been reported by local farmers through the observation of the abundance of bees that populate their hives or through fluctuations in honey production. Efforts to evaluate pesticide impacts on pollinators are needed throughout the African continent, as existing studies are limited and geographically widely spread, and some of these raise great concerns. For example, traditional beekeepers in Burkina Faso have noted that their hives situated near cotton fields treated with pesticides had lower numbers of adult bees and were less productive than those which were kept farther away (Gomgnimbou et al., 2010). Similarly Otieno et al. (2011) found pesticide use was negatively related to pollinator abundance in fields in Eastern Kenya. However, another study (Muli et al., 2014) suggested impacts may not be severe in all cases; relatively low levels of residues of up to four pesticides were detected in 14 out of 15 honey bee hives sampled across Kenya. In South Africa, pesticide use and isolation from natural habitat were associated with declines in flying pollinators and in mango production (Carvalheiro et al., 2012), although this effect was not consistent between years (Carvalheiro et al., 2010). There is a clear need for more studies of impacts of pesticides on pollinators and pollination given the economic importance of insect-pollinated crops throughout the African continent (Archer et al., 2014; Steward et al., 2014) and indeed across many developing countries.

2.3.1.2 Potential impacts of pesticides on pollinators

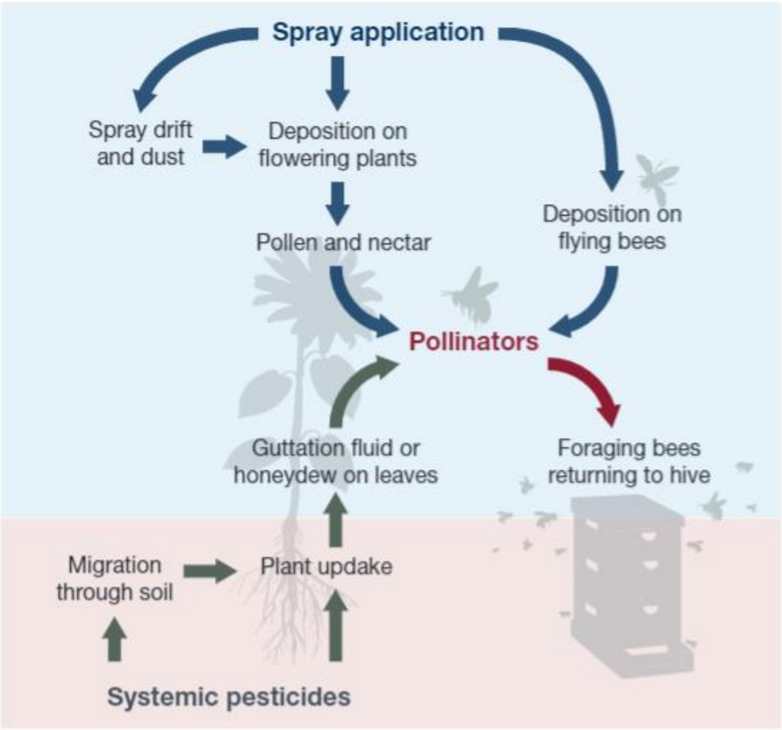
The use of insecticides is of particular concern due to their potential for effects on non-target insect pollinators due to their inherent toxicity (UNEP, 2010; EASAC, 2015). Although there is also evidence that some pesticide co-formulants such as adjuvants (used to enhance application and uptake of the pesticide) or synergists may also show toxicity at high doses (Donovan and Elliott, 2001; Ciarlo et al., 2012; Zhu et al., 2014; Mullin et al., 2015). Insecticides vary widely in their mode of action from molecules interacting with nerve receptors (see examples in Table 2.3.1) to those affecting energy metabolism and development (e.g. insect growth regulators). Novel pesticides and modes of action are continually sought to address rapid development of resistance in target pests (Ohta and Ozoe, 2014). There are very limited data available globally on actual usage of insecticides (as opposed to sales data) by farmers on crops attractive to pollinators from which to base a global assessment of potential risk. However, data from Kenya, Brazil and the Netherlands demonstrate the differences among countries in the availability of pesticides that are inherently toxic to bees (Figure 2.3.2; van der Valk et al., 2013).



8. Figure 2.3.2: Hazard (LD50) of pesticides used on bee-attractive focal crops
in Brazil (melon and tomato), Kenya (coffee, curcurbits, French bean and tomato) and the Netherlands (apple and tomato) (% pesticides refers to number registered or used) (data from van der Valk et al., 2013).

Risk assessment (which considers both toxicity and exposure, Box 2.3.2) is considered more relevant in defining the potential impact of pesticides than hazard (toxicity) identification alone (van der Valk et al., 2013). Pollinator exposure to insecticides, their impact and the potential for population response is affected by a wide range of factors including crop type, the timing, rate and method of pesticide application and the ecological traits (e.g. diurnal activity, foraging specialisation, life history) of managed and wild pollinators (Defra, 2008).

The direct exposure of pollinators to pesticides may occur through a number of routes including ingestion of contaminated pollen, nectar, aphid honeydew or water (e.g. from contaminated puddles within fields), contact with drift or overspray during foliar applications, and contact with residues on foliage and flowers (e.g. Figure 2.3.3 for bees (EFSA, 2012)). Solitary bees may also be exposed via residues in soil and on plant nesting material (EFSA, 2012). In flowering crops, systemic pesticide residues may be transferred into pollen and nectar collected and consumed by pollinators with the potential for adverse effects from chronic low-level exposure (Goulson, 2013; Pilling et al., 2013; Cutler and Scott-Dupree, 2014); although there is some evidence that systemicity is not a property limited to the new classes of insecticide, with similar properties reported for dimethoate, an organophosphorus pesticide (Davis and Shuel, 1988). Pesticide residues (parent molecule and any toxic metabolites) in nectar and pollen vary considerably depending on the mode of application. For example, a collation of studies on oilseed rape found average maximum values vary from around 1.9 µg/kg in nectar and 6.1 µg/kg in pollen following neonicotinoid seed treatment, but that residues are 10-20 fold greater when the same compounds are applied as foliar sprays at a similar rate per hectare (10 g a.i./Ha) (EFSA, 2013; Godfray et al., 2014) or as soil drenches (Dively and Kamel, 2012; Stoner and Eitzer, 2012). Pollinators may be exposed to residues is via guttation fluid (plant xylem fluid exuded through specialised pores onto the leaf surface during periods of high root pressure), where neonicotinoid residues can be extremely high in the early stages of crop growth (Bonmatin et al., 2015). Other sources of contaminated water include puddles in fields (Samson-Robert et al., 2014). However, this is not currently considered a significant route of exposure for honey bees, although data on water sources are more limited for other bee species (Pistorius et al., 2012; Godfray et al., 2014). Another potential route of exposure is the generation of dust, containing insecticide, that may drift onto nearby flowering crops or weeds during drilling of treated seed (Krupke et al., 2012; Pisa et al., 2014). There have been a number of studies demonstrating the lethal effects of dusts generated from neonicotinoid-treated seeds during drilling (Bonmatin et al., 2015) and large-scale honey bee mortality has resulted from treated seed when the seed contained high levels of dust particularly when it was incorrectly coated or dust based seed lubricants were added during drilling when dust drifted onto flowering crops and weeds (Pistorius et al., 2009; PMRA, 2014). There is evidence that appropriate technical measures can be adopted to reduce the associated risk of dust although no single measure has currently been shown to be totally effective (Kubiak et al., 2012; Nuyttens et al., 2013).



9. Figure 2.3.3: Summary of key identified routes of exposure in honey bees
(EFSA, 2012); similar routes of exposure are likely for other bees and other pollinators.

There is evidence that the identity of pesticides present and scale of the exposure of honey bee colonies (levels in pollen, nectar/honey and wax) differ between crop type (Pettis et al., 2013) and regions reflecting differences in pesticide approval and use (Bogdanov, 2006; Johnson et al., 2010; Mullin et al., 2010; Chauzat et al., 2011; Al-Waili et al., 2012). However, quantitative data on an individual pollinator's exposure to pesticides is limited, i.e. actual ingestion by a foraging bee, not measured residues. Pollen and nectar consumption has been almost entirely studied in honey bees and often extrapolated from estimated nutritional requirements as a proxy for foraging rate (Thompson, 2012) rather than measured directly. Exposure factors have been evaluated for wild bees on focal crops in Brazil, Kenya and the Netherlands by (van der Valk et al., 2013). The overall likelihood of exposure of wild bees to pesticides were evaluated as "probably similar" to *Apis mellifera* in the case of *Apis mellifera scutellata* and *Xylocopa*, but due to a lack of information were "unclear" for *Patellapis* and *Megachile* and "possibly greater" for Halictidae. However, from a review of the literature it is clear there is a lack of accurate data on key aspects of the biology of non-*Apis* species (e.g. nectar consumption by foraging bees) to allow exposure under field conditions to be quantified.

Pesticides may result in impacts on pollinators without direct exposure. Indirect effects on pollinators include the removal of nectar/pollen sources and/or nest sites by herbicides (Potts et al., 2010). Together both direct and indirect effects of pesticides, in combination with other aspects of monoculture agriculture, may contribute to observations at the landscape scale of a tendency for reduced wild bee and butterfly species richness in response to pesticide application (Brittain et al., 2010; Brittain and Potts, 2011; Vanbergen et al., 2013).

4. Table 2.3.1: Factors affecting pesticide risk to pollinators
(adapted from van der Valk et al., 2013)

Risk factor	Pesticide use has increased risk when:	Comments
Exposure		
Crop factors	Overall crop area is high	Decreased risk with crop patchiness
	Application timing overlaps with:	
	Crop flowering	
	Flowering of attractive weeds	
	Seasonal timing of bee foraging and collection of nesting materials	
	Crop has extrafloral nectaries	
	Crop is regularly infested with high numbers of aphids producing honeydew	
	Drinking water only available in-crop, e.g. guttation, puddles	
Bee biology	Nest sites located in field or field border	Low persistence pesticides applied out of foraging period decrease risk
	Short foraging range for in-field/field border nests	
	Extensive time spent out of nest/hive	If collective pollen/honey storage (social bees) due to mixing/maturation/microbial action risk decreased
	Foraging period when pesticides applied	
	Number of days spent foraging on crop	
	Few crop/plant species used as forage	
	High quantity of pollen and nectar collected per day	
	High quantity of nectar consumed per day	
	Small bodyweight; relatively higher exposure	
	If forage not stored prior to consumption	
Pesticide use/application practices	Some formulation types e.g. micro-encapsulated, sugary baits	If systemic specific exposure/impact assessment
	Some modes of application, e.g. aerial, dusting, dusty seeds without adapted machinery	If insect growth regulator specific impact on brood
	Increased application rate for same pesticide product	Decreased risk for soil/ seed treatments with non-systemic pesticides
	Increased application frequency	
	Persistent systemic pesticides applied as soil treatment to seed treatment to a previous rotational crop	
Impact and recovery		

Risk factor	Pesticide use has increased risk when:	Comments
Pesticide properties	Low acute LD50 (for similar exposure levels) Higher foliar residual toxicity (persistence of residues on leaf/flower surface)	Foliar residual toxicity affects impact and likelihood of recovery
Life history and population dynamics	Lower metabolic rate of adults (decreased detoxication) Low degree of sociality with no/few foragers Higher proportion of population of colony active out of the nest (= high impact for colony/population Longer development time of queen/reproductive female increases exposure (if development overlaps with flowering) Small number of offspring per female decreases likelihood of population recovery after impact Fewer generations per season decreases likelihood of population recovery after impact Decreased number of swarms per colony –less likelihood of population maintenance/recovery Lower swarm migration distance lower likelihood of population recovery after pesticide impact	High degree of sociality decreases impact as to population/colony as pesticide effects mainly on foragers (except IGRs)

2.3.1.3 Evidence of lethal effects during pesticide use

Insecticides vary widely (several orders of magnitude) in toxicity to pollinators depending on their mode of action (see Table 2.3.2) and target life-stage (e.g. insect growth regulators only directly affect larvae/pupae). Even within an insecticide class, toxicity can vary from a few nanograms (ng) per bee to several thousand micrograms (µg) per bee, as in the case of the neonicotinoids (Blacquière et al., 2012). There is evidence that the detoxification enzymes in honey bees are less diverse than in other insects making them less well adapted to respond to exposure to a range of chemicals (Johnson et al., 2010; Mao et al., 2013) and even this limited range of enzymes is also affected by the age of the bee, the time of year, etc. (Smirle and Winston, 1987). However, there is also evidence that *Apis mellifera* is no more sensitive to insecticides than other insect species (Hardstone and Scott, 2010). The relative sensitivity of different bee species to the acute (single exposure) effects of insecticides and other pesticides is similar, i.e., the acute toxicity (LD_{50}) is within an order of magnitude (Arena and Sgolastra, 2014), particularly if body mass (80-300mg) is taken into account (Arena and Sgolastra, 2014; Fischer and Moriarty, 2014). However, the chronic toxicity (LC_{50}) of pesticides may be more variable; some evidence suggests clearance of insecticides may differ among species of bees (Cresswell et al., 2014). Other factors have also been identified as affecting the toxicity of insecticides to honey bees, including nutrition (Godfray et al., 2014; Schmehl et al., 2014) and disease (Vidau et al., 2011) (see section 2.4.1).

The largest published databases on acute pesticide effects under real-use field conditions are formal incident monitoring schemes that are limited to honey bees (only a handful of reported incidents have involved bumble bees). These schemes have been instigated by national governments in a number of European countries, Australia, Canada, USA and Japan (OECD, 2010) and are reliant on notification of honey bee deaths either on a voluntary basis by beekeepers or as a requirement for pesticide registrants. A single incident may range from a few bees to several thousand bees but has rarely been linked to an assessment of the longer-term impact on the colony, e.g., the neonicotinoid seed treatment dust incident in Germany (Wurfel, 2008). Where voluntary reporting exists there is potential for under-reporting due to reticence of beekeepers to report incidents and risk the loss of apiary sites with good forage often on land belonging to farmers (Fischer and Moriarty, 2014). The longest-running incident schemes are primarily in Europe (Germany, Netherlands and UK), where the number of incidents where pesticides have been identified as a cause declined from circa 200 incidents per year in the 1980s to around 50 by 2006 (Barnett et al., 2007; Thompson and Thorbahn, 2009); more recent data from the UK show a decline from up to an annual average of 48 incidents between 1981 and 1991 to an average of 7 per year between 2010 and 2014 (http://www.pesticides.gov.uk/guidance/industries/pesticides/topics/reducing-environmental-impact/wildlife/WIIS-Quarterly-Reports.htm). Similar schemes have been established in Japan (http://www.maff.go.jp/j/press/syouan/nouyaku/150623.html), where recent incidents of honey bee mortality have centred around neonicotinoid insecticide sprays to control rice stink bug.

Of more than 8,500 detections of pesticides in bee and incident-related plant samples submitted to the European pesticide poisoning incident (bee-kill) schemes, between 1981 and 2007 nearly 50% contained insecticides, 40% contained fungicides and 11% contained varroacides (a sample may contain more than one pesticide and several samples may relate to a single incident). Identifying whether pesticides are a cause of acute bee losses can be challenging because detection of a pesticide residue may not necessarily be related to an adverse effect and residues may decline in dead bees depending on the persistence of the chemical. Data linking lethal exposure to the resulting residues in bees are limited to a few insecticides (Greig-Smith et al., 1994; Thompson, 2012). Of the separate incidents of honey bee mortality in Europe where insecticides were detected (an incident may include more than one coloniy or more than one apiary site), 27% contained organophosphorus insecticides or carbamate compounds, 14% contained organochlorine insecticide (gamma-HCH (lindane), and dieldrin) and pyrethroid insecticides were present in 7.8% of incidents; none were associated with neonicotinoids (Thompson and Thorbahn, 2009). Between 1981 and 1991 around 65-70% of the 545 incidents in the UK were identified as due to farmers not complying with label instructions and applying insecticides in flowering beans, peas and oilseed rape, or crops containing flowering weeds (Greig-Smith et al., 1994); of the remainder 3% were associated with aerial applications (no longer permitted in the UK), 2% with use in feral bee control and in the remainder of reported incidents the use often could not be clearly identified from the information available. These incidents have resulted in improved regulation and enforcement in Europe (e.g. Directive 91/414 EEC) with subsequent reduction in incidents as well as providing information where uses according to the label require further education of farmers (Thompson and Thorbahn, 2009). For example, the reduction in the number of reported aphid honeydew-related insecticide incidents in the Netherlands from 119 in 2003 to 17 in 2006 was attributed to the reduction in the aphid control threshold for insecticide use in potatoes, which limited the availability of aphid honeydew (a source of sugar) and thus attraction of honey bees to the crop (Thompson and Thorbahn, 2009).

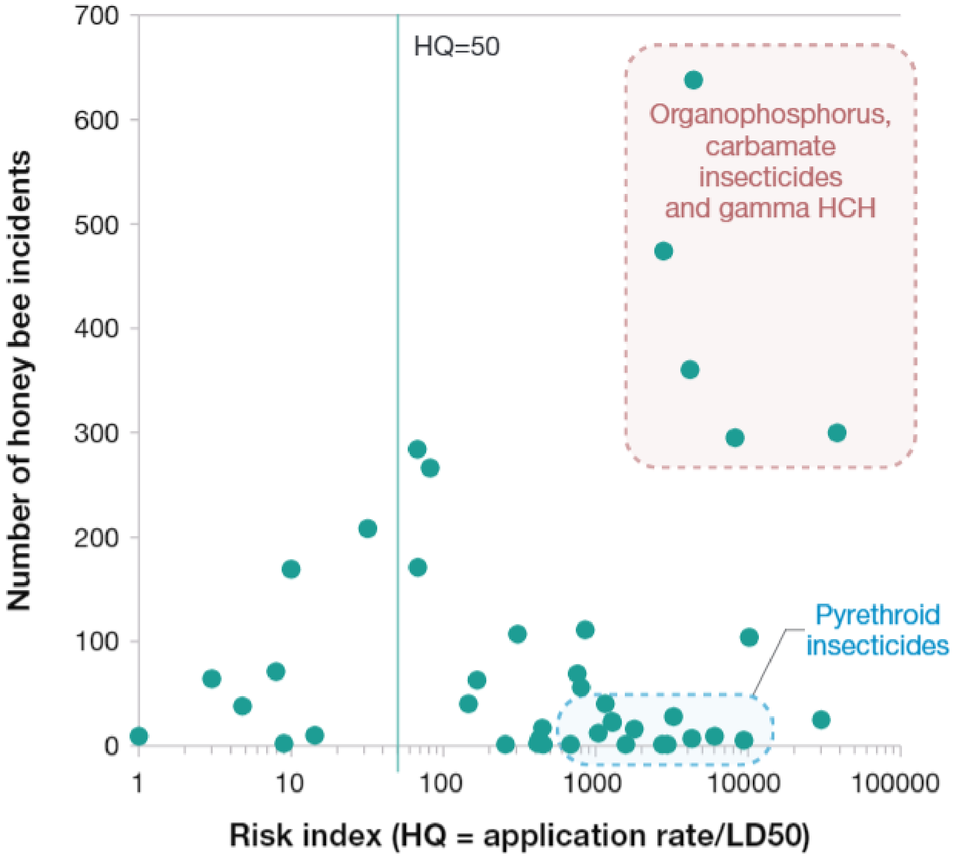
These experiences from countries with incident schemes suggest that where there is no effective regulation or enforcement of key mitigation (Heong et al., 2013), it is likely that incidents of insecticide-associated honey bee mortality are occurring; in the absence of honey bee impacts effects on wild bees are unclear. Mitigation measures to protect honey bees include only applying insecticides outside the flowering period, and closing or removing beehives or use of low-drift technology to reduce drift of spray onto nearby flowering crops or hedgerows containing flowering plants. To be effective, mitigation needs to take account of local practices and also apply to other insect pollinators. For example, beekeepers keeping native bees in Korea have reported that impacts of pesticide spray cannot be avoided because their hives cannot be moved (Park and Youn, 2012).

These incident data have also been used to derive the hazard quotient (application rate (g active ingredient/ha)/LD_{50} (µg active ingredient/bee)) threshold of 50 to identify those uses of foliar applied pesticides with a risk of resulting in acute honey bee mortality and requiring further evaluation in the risk assessment, e.g., semi-field and field studies (EPPO, 2010). A comparison of the hazard quotient (HQ) with the number of incidents reported is shown in Figure 2.3.4. Although the HQ for pyrethroid insecticides is far greater than 50 there is good evidence that, when applied according to the label, particularly at lower application rates, and not mixed with ergosterol biosynthesis-inhibiting (EBI) fungicides (see pesticide mixtures section) honey bee incidents are rarely observed due to the repellent properties of some of this class of insecticide (Thompson and Thorbahn, 2009). These national monitoring schemes have shown a decrease in the overall number of incidents reported over the last 20 years following reactive changes to product registrations and stewardship, e.g. limiting applications to non bee-attractive crops. However, high profile incidents are still reported such as the off-label use of neonicotinoid dinotefuran on linden trees in the USA, which resulted in a significant bumble bee kill (Katchadoorian, 2013), dust generated during planting of a poor-quality neonicotinoid seed treatment in Germany that affected over 11,000 honey bee colonies (Pistorius et al., 2009), a similar problem in Italy (APENET, 2011), and dust generation during planting of neonicotinoid-treated seed in the presence of seed lubricants in Ontario, Canada (PMRA, 2013; Cutler et al., 2014b; see http://www.hc-sc.gc.ca/cps-spc/pubs/pest/_fact-fiche/bee_mortality-mortalite_abeille-eng.php).

It is well established that insecticides can affect individuals and populations of bees, and the impact will increase with increased exposure, e.g. if the label does not provide clear and effective mitigation measures (mitigation selected for honey bees may not always protect other pollinator species (Thompson and Hunt, 1999), or the user does not comply with the label (Johansen, 1977; Kevan et al., 1990; Thompson and Thorbahn, 2009; Brittain et al., 2010; Hordzi et al., 2010). However, beyond the small number of country-level incident schemes there are few data available on incidents occurring following approved uses or on the scale of poor practice/non-compliance. There is evidence of deliberate misuse, i.e., intentional poisoning (Thompson and Thorbahn, 2009). Albert and Cruz (2006) present the testimony of owners of an organic farm where traditional and local knowledge about agricultural practices were being regenerated in Valencia, Spain. They explained the problems with a law (called the "pinyolà" decree) that forbids pollinators in certain areas in this community, where plantations of clementines (non-native) have been introduced. Pollination generates seed in clementines, reducing their market value, therefore pesticides are being used in order to kill pollinators. There is also evidence that home and garden pesticide use can impact butterfly and bumble bee populations (Muratet and Fontaine, 2015). However, there is also good evidence both from national incident schemes (Thompson and Thorbahn, 2009) and from field trials (Gels et al., 2002; Stadler et al., 2003; Shuler et al., 2005; Larson et al., 2013) that the effects of insecticides on individuals and populations of honey bees can be reduced by appropriate mitigation measures, although the effectiveness of these mitigation measures for wild bee populations is unclear.

There is limited evidence that increasing the proportion of natural habitat in the surrounding landscape can buffer the effects of pesticide use on wild bee abundance and species richness. For example, Park et al. (2015) observed pesticide effects on a wild bee community visiting an apple (*Malus domestica*) orchard were buffered by increasing proportion of natural habitat in the surrounding landscape. The direct consequences for crop yield from pesticide-induced pollinator losses under field conditions are unresolved (Kevan et al., 1990; Partap et al., 2001; Richards, 2001). In the presence of pest pressure, pesticides can enhance crop yield (Oerke, 2006) but a more limited evidence base also demonstrates that pesticides used in combination with managed pollinators can enhance crop yield (Lundin et al., 2013; Melathopoulos et al., 2014) and environmental health (Scriber, 2004) and may even improve abundance of butterflies and bumble bees in urban situations (Muratet and Fontaine, 2015). More recent reviews have specifically questioned the widespread use of the neonicotinoid seed treatments and suggested there is little to no published evidence to demonstrate economic benefits of these for farmers (EPA, 2015; Van der Sluijs et al., 2015), although the number of published trials evaluating this directly is very small and conflicting data also exist (Afifi et al., 2015; AgInformatics, 2004). In a recent survey on neonicotinoid seed treatments (Budge et al., 2015) the benefits of these seed treatments to crop production in the UK were shown through reduced applications of other insecticides in autumn and increased yield in the presence of pest pressure, although this was variable between years. However, it also showed an apparent correlation between the scale of use of imidacloprid as a seed treatment on oilseed rape seed and increased honey bee colony loss. There was no apparent correlation with total neonicotinoid use (making the underlying mechanism of the correlation unclear) and a number of other factors, such as beekeeping practices and presence of other forage sources, were not included. Further large-scale studies are required to develop a greater understanding of the balance between the benefits of pesticide use in crop production and the potential risks to pollinator or other non-target populations.

There have been suggestions that chronic exposure to certain insecticides (particularly neonicotinoids) may result in delayed but direct mortality of honey bees (Rondeau et al., 2014; Sanchez-Bayo and Goka, 2014). However individual honey bees have been shown to clear imidacloprid rapidly (Cresswell et al., 2014) and although honey bee colonies fed high levels of imidacloprid resulted in high adult mortality and colony failure (Dively et al., 2015), feeding with more field-realistic exposure levels over an extended period did not result in increased adult mortality or colony failure (Faucon et al., 2005; Dively et al., 2015). A similar lack of adult honey bee mortality following long-term (2 – 6 weeks) exposure of colonies has been reported for thiamethoxam and clothianidin (Pilling et al., 2013; Cutler et al., 2014a; Sandrock et al., 2014). Recent approaches of using chronic toxicity (LC_{50}) data to assess cumulative toxicity may directly address such concerns for a wider range of pesticides (EFSA, 2013).



10. Figure 2.3.4: Comparison of the risk index (HQ = application rate/LD50) with the number of honey bee incidents in which the pesticide was detected.

Data are from the UK, Germany and the Netherlands 1985-2007. An HQ of 50 is used in risk assessments for pesticides to identify those uses that require further evaluation. Incidents may also contain pesticides not related to bee mortality, e.g. 1) fluvalinate used as a varroacide (to control the *Varroa* mite) and 2) captan, a fungicide applied at high rates (Thompson and Thorbahn 2009). The circles highlight the groupings of incidents involving different classes of insecticides. For reference the HQ of the neonicotinoids imidacloprid, thiamethoxam and clothiandin are >1000 but no incidents were reported.

Class	Examples (chemical subgroup or exemplifying active ingredient	Mode of action	application rate + 10's g ai/ha ++ 100's of g ai/ha	Example honeybee LD50 µg a.i. (active ingredient)/bee [1]
Acetylcholinesterase (AChE) inhibitors	Organophosphates, carbamates	Inhibits enzyme which terminates the action of the excitatory neurotransmitter acetylcholine at nerve synapses. Acetylcholine is the major excitatory neurotransmitter in insects.	++	Dimethoate 0.1
GABA-gated chloride channel antagonists	Cyclodiene organochlorines; phenylpyrazoles	Blocks GABA-activated chloride channel; GABA is the major inhibitory neurotransmitter in insects	+	Fipronil 0.004 (oral lowest)
Sodium channel modulators	Pyrethroids, pyrethrins; DDT/methoxychlor	Keep sodium channels open causing hyperexcitation and in some cases nerve block. Sodium channels are involved in the propagation of action potentials along nerve axons.	+	Deltamethrin 0.0015 (contact lowest)
Nicotinic acetylcholine receptor (nAChR) agonists	Neonicotinoids; nicotine; sulfoxaflor; butenolides	Mimic the agonist action of acetylcholine at nAChR causing hyperexcitation. Acetylcholine is the major excitatory neurotransmitter in insects.	+	Thiacloprid 17.3 (oral lowest) Imidacloprid 0.0037 (oral lowest)
Nicotinic acetylcholine receptor (nAChR) allosteric modulators	Spinosyns	Allosterically activate nAChRs causing hyperexcitation. Acetylcholine is the major excitatory neurotransmitter in insects.	+	Spinosad 0.0036 (contact lowest)
Chloride channel activators	Avermectins, milbemectins	Allosterically activate glutamate-gated chloride channels causing paralysis. Glutamate is an important inhibitory neurotransmitter in insects	+	Abamectin 0.002 (contact)
Modulators of chlordotonal organs	Pymetrozine; flonicamid	Stimulate chlordotonal proprioceptors by an unknown mechanism; impairs fine motor control, resulting in disruption of feeding and other behaviours of Hemiptera and certain other insects	+	Pymetrozine >117 (oral lowest)
Voltage dependent sodium channel blockers	Indoxacarb; metaflumizone	Block sodium channels causing nervous system shutdown and paralysis. Sodium channels are involved in the propagation of action potentials along nerve axons.	+	Indoxacarb 0.07 (contact lowest)
Ryanodine receptor modulators	Diamides	Activate muscle ryanodine receptors leading to contraction and paralysis. Ryanodine receptors mediate calcium release into the cytoplasm from intracellular stores.	+	Chlorantraniprole >4 (contact lowest)

5. Table 2.3.2: Examples of classes. Mode of action and toxicity of insecticides acting on nerve/muscle targets (from IRAC MoA Classification v7.3 February 2014 http://www.irac-online.org/documents/moa-classification/?ext=pdf)

[#1] toxicity data from http://www.agritox.anses.fr/php/fiches.php

2.3.1.4 Sublethal effects of pesticides on bees

2.3.1.4.1 Importance of sublethal effects

In addition to the traditional measurements of lethal effects happening during acute exposure to pesticides, an increasing number of studies have focused on the sublethal effects of pesticides on pollinators, since the 1970's. Sublethal effects are defined as the effects on individuals that survive exposure (Desneux et al., 2007). They mainly follow chronic exposure to pesticides, but can also be a consequence of acute exposure. A pioneering study by Schricker and Stephen (1970) showed that when honey bees were exposed to a sublethal dose of parathion, an organophosphate insecticide, they were unable to communicate the direction of a food source to other bees. Using a variety of methods, many studies have shown the effects of newer classes of insecticides, for instance pyrethroids (Vandame et al., 1995) and neonicotinoids (Henry et al., 2012), causing alterations in the navigation of bees and their orientation to food resources and colony location, resulting in bee losses. After reviewing the documented sublethal effects of pesticides on bees, we examine the conclusions of the principal reviews on this topic with respect to the role of sublethal effects of these pesticides in the decline of bees, and the pollination they provide.

2.3.1.4.2 Range of sublethal effects

An extensive variety of sublethal effects has been studied, and can be classified into effects at the individual (physiological and behavioral) and colony levels. We provide several examples of each detected effect, based on the principal reviews (Thompson, 2003; Desneux et al., 2007; Belzunces et al., 2012; Van der Sluijs et al., 2013; Godfray et al., 2014; Pisa et al., 2014) (see Table 2.3.3).

6. **Table 2.3.3: Non-exhaustive list of sublethal effects of different classes of insecticides and acaricides** (Bz: benzamides; Oc: organochlorines; Nn: neonicotinoids; Op: organophosphates; Py: pyrethroids), fungicides (Az: azoles) and herbicides (Ph: Phosphonoglycines) on individual (physiology and behavior) and colony levels for various species of bees (Ac: *Apis cerana*; Am: *Apis mellifera*; Bt: *Bombus terrestris*; Mq: *Melipona quadrifasciata*; Mr: *Megachile rotundata*; Ob: *Osmia bicornis*).

	Species	Compound	Effect	Reference
1. Physiology				
Neurophysiology	*Am*	Op-fenitrothion Op - coumaphos Py-cypermethrin	Enzyme inhibition	(Bendahou et al., 1999; Palmer et al., 2013)
Immunity	*Am*	Nn-clothianidin	Decreased immunity, increased viral pathogen replication	(Di Prisco et al., 2013)
Thermoregulation	*Am*	Az-prochloraz Az-difenoconazole Py-deltamethrin	Hypothermia (separately and in synergistic action)	(Vandame and Belzunces, 1998)

	Species	Compound	Effect	Reference
Reproduction	*Ac, Am*	Bz-diflubenzuron Bz-penfluron	Decreased brood production	(Chandel and Gupta, 1992)
	Bt	Nn-imidacloprid	Decreased brood production	(Tasei et al., 2000)
	Ob	Nn-thiamethoxam Nn-clothianidin	Reduced offspring production, male biased offspring sex-ratio	(Sandrock et al., 2014)
Longevity	*Am*	Py-deltamethrin Nn-imidacloprid	Reduced adult longevity	(Dechaume et al., 2003)
	Bt	Nn-thiamethoxam Nn-clothianidin	Truncated worker production, reduced worker longevity	(Fauser-Misslin et al., 2014)
Fecundity	*Mr*	Py-deltamethrin	Reduced egg laying	(Tasei et al., 1988)
2. Behavior				
Feeding	*Bt*	Py-deltamethrin	Reduced feeding stimulation	(Tasei, 1994)
Mobility	*Am*	Py-permethrin	Increased self-cleaning, trembling, decreased walking and food giving	(Cox and Wilson, 1984)
	Mq	Nn-imidacloprid	Affected mushroom bodies development, impaired walking behavior	(Tomé et al., 2012)
	Am	Nn-thiamethoxam Nn-imidacloprid Nn-clothianidin	Loss of posture control, failure to right body	(Williamson et al., 2014)
Learning	*Am*	Az-prochloraz Py-deltamethrin Oc-endosulfan Nn-fipronil	Decreased olfactory performance, impaired memory and brain performance	(Decourtye et al., 2004; Decourtye et al., 2005)
	Am	Nn-imidacloprid	Impaired olfactory associative behavior	(Yang et al., 2012)
	Am	Op-coumpahos Nn-imidacloprid	Impaired conditioning of proboscis extension	(Williamson and Wright, 2013)
	Bt	Nn-imidacloprid	Chronic behavioral impairment	(Gill and Raine, 2014)
	Am	Ph-glyphosate	Reduced sensitivity to sucrose and reduced learning performance	(Herbert et al., 2014)*
Navigation	*Am*	Py-deltamethrin	Failure in returning to the colony	(Vandame et al., 1995)
	Am	Nn-imidacloprid	Failure in returning to the colony	(Bortolotti et al., 2003)
	Am	Nn-thiamethoxam	Failure in returning to the colony	(Henry et al., 2012)
Communication	*Am*	Op-parathion	Incorrect communication of information during dance	(Schricker and Stephen, 1970)
Defense	*Ac*	Nn-imidacloprid	Decreased avoidance of predators	(Tan et al., 2014)
3. Colony				
Foraging	*Bt*	Nn-imidacloprid	Reduced pollen foraging	(Feltham et al., 2014)
	Am	Nn-fipronil Nn-imidacloprid	Reduced rate active/total bees, decreased foraging	(Colin et al., 2004)

	Species	Compound	Effect	Reference
Colony performance	*Bt*	Nn-imidacloprid	Reduced growth rate, reduced queen production	(Whitehorn et al., 2012)
	Bt	Py-cyhalotrin Nn-imidacloprid	Increased worker mortality and pollen collection, reduced brood development	(Gill et al., 2012)
	Am	Nn-thiamethoxam Nn-clothianidin	Declining number of bees, queen failure, reduced propensity to swarm	(Sandrock et al., 2014)
	Bt	Nn-imidacloprid	Decreasing birth rate, colony failure	(Bryden et al., 2013)

* Note that Thompson et al. (2014) found no sublethal effects of glyphosate on honeybees.

As shown in Table 2.3.3, there exist a broad variety of sublethal effects, including individual physiological and behavioral effects as well as colony-level effects. Most of these effects have been shown with the honey bee, and most of the recent studies look at neonicotinoid insecticides effects. Despite this research, important gaps of knowledge remain; for example: 1) most studies have been carried out with honey bees, a few with the bumble bee, *Bombus terrestris,* but very few with other social or solitary bee species (Sandrock et al., 2014) (Table 2.3.3, Figure 2.3.6). Consequently, the actual effects on pollinator communities are not clear; 2) most research has been performed with insecticides, particularly of the neonicotinoid class; therefore less is known of the sublethal effects of other insecticides, herbicides, or fungicides; 3) the synergistic effects of pesticides at sublethal doses have been little studied, despite the possibility of severe effects (Colin and Belzunces, 1992; Vandame and Belzunces, 1998); 4) the interaction of pesticides at sublethal doses with other key pressures on pollinators (land-use intensification, climate change, alien species, pests and pathogens), while largely unknown (Vanbergen et al., 2013), is likely to contribute to the overall pressure on pollinators (Goulson et al., 2015) (see section 2.7.2. case study 2: pathogens and chemicals in the environment).

2.3.1.4.3 Sublethal effects and the threat to bees

The overview given in Table 2.3.3 raises an important question: what is the current role of these numerous sublethal effects in terms of the decline of bees worldwide? Nine reviews have provided a variety of responses to this question.

One set of three reviews deals with sublethal effects of pesticides in general. The first (Thompson, 2003) reviewed 75 studies dealing with behavioural effects of pesticides on bees, ranging from effects on odour discrimination to disruption of the homing behavior, showing that these effects occur at pesticide levels at or below those estimated to occur following field applications. It states that long-term impact on the colony of the behavioural effects is rarely reported. It calls for using laboratory studies to address sublethal effects for compounds with low acute toxicities and low application rates. The second review (Desneux et al., 2007), based on 147 studies, showed a wide range of sublethal effects, principally perturbation of individual development, foraging patterns, feeding behavior, and learning processes; it concludes that the consequences of sublethal effects on populations and communities of pollinators are not well understood, and calls for development of methods to test these effects, and their inclusion in regulatory procedures. A third review (Belzunces et al., 2012), based on approximately 250 studies, focused on neural effects of insecticides on honey bees, highlighting the fact that the mechanisms by which insecticides elicit their effects are not restricted to the interaction between the active substance and the molecular target responsible for the insecticidal action. It also showed that synergistic effects between different insecticides are poorly understood in bees, and very likely underestimated.

Another set of six reviews more specifically addressed the sublethal effects of neonicotinoid insecticides. The first (Blacquière et al., 2012) reviewed approximately 110 studies, reporting a wide variety of sublethal effects. It showed that levels of neonicotinoid residues in plants (specifically in nectar and pollen) tended to be lower than levels required to produce toxic effects (either acute or chronic) on bees. Blacquière et al. (2012) also highlighted that there was a lack of reliable data with most analyses having been conducted near the limits of detection and for just a few crops. Despite a number of sublethal effects documented in laboratory studies, Blacquière et al. (2012) found that no effects were observed in field studies at field-realistic dosages. This result then can be partly due to the fact that most studies were conducted after seed treatments, a mode of exposure that generates lower levels of residues than other ways of application. A further review of 259 studies (Godfray et al., 2014) focussed on the sublethal effects in laboratory and semi-field experiments. This review also highlighted the need to understand further whether these effects corresponded to sub-lethal doses received by pollinators in the field leading to significant impairment of individual performance, whether there is a cumulative effect on colonies and populations affecting pollination in farm and non-farm landscapes, and what the consequences are for the viability of pollinator populations.

The other four of these six reviews (three of which (Van der Sluijs et al., 2013, Simon-Delso et al., 2014, Pisa et al., 2015) have common contributors) conclude in a different way and state that the sublethal effects of neonicotinoids very likely have a negative impact in individual and social performances of bees. A meta-analysis of 14 studies on the effects of imidacloprid on honey bees (Cresswell, 2011) estimated that field-realistic levels in nectar will have no lethal effects, but will reduce expected performance in adult honey bees under laboratory and semi-field conditions by 6 to 20%. This author's statistical power analysis showed that the field trials published at this time (up to 2011), which reported no effects of neonicotinoids on honey bees, were incapable of detecting these predicted sublethal effects with conventionally-accepted levels of certainty. Therefore, this study raised concern regarding the ability of the reviewed studies to detect a sublethal impact of imidacloprid under field conditions, a view supported by the more recent study by Rundlöf et al. (2015), who used a study design with sufficient replication (8 pairs of fields) to detect a 20% effect on honey bee colony strength if it had occurred. Similarly, a review of 163 scientific studies (Van der Sluijs et al., 2013) concluded that at field realistic doses, neonicotinoids produce a wide variety of adverse sublethal effects in honey bee and bumble bee colonies, affecting colony performance. These authors also warn that long-term effects are not taken into account by tests for marketing authorization, and in general field tests have a low reliability due to the number of environmental variables involved. Recently, the International Task Force on Systemic Pesticides, a group of 29 independent scientists set by the IUCN (International Union for the Conservation of Nature), published a series of complementary reviews. In one of these reviews, Simon-Delso et al. (2015) summarize the high number of metabolites derived from neonicotinoid and fipronil, and underline how limited is the knowledge about their toxicity profiles. A different analysis by the same team (Pisa et al., 2015), based on more than 350 studies, reviews the effects of these compounds on invertebrates, including honey bees, showing a wide range of sublethal effects on activity, locomotion, metabolism, ontogenetic development, behaviour, learning and memory. In contrast with Blacquière et al. (2012), they conclude that there is a clear body of evidence showing that existing levels of pollution, resulting from authorized uses, frequently exceed the lowest observed adverse effect concentrations and are thus likely to have large-scale and wide-ranging negative biological and ecological impacts. They finally suggest that regulatory agencies apply more precautionary principles and tighten regulations on neonicotinoids and fipronil.

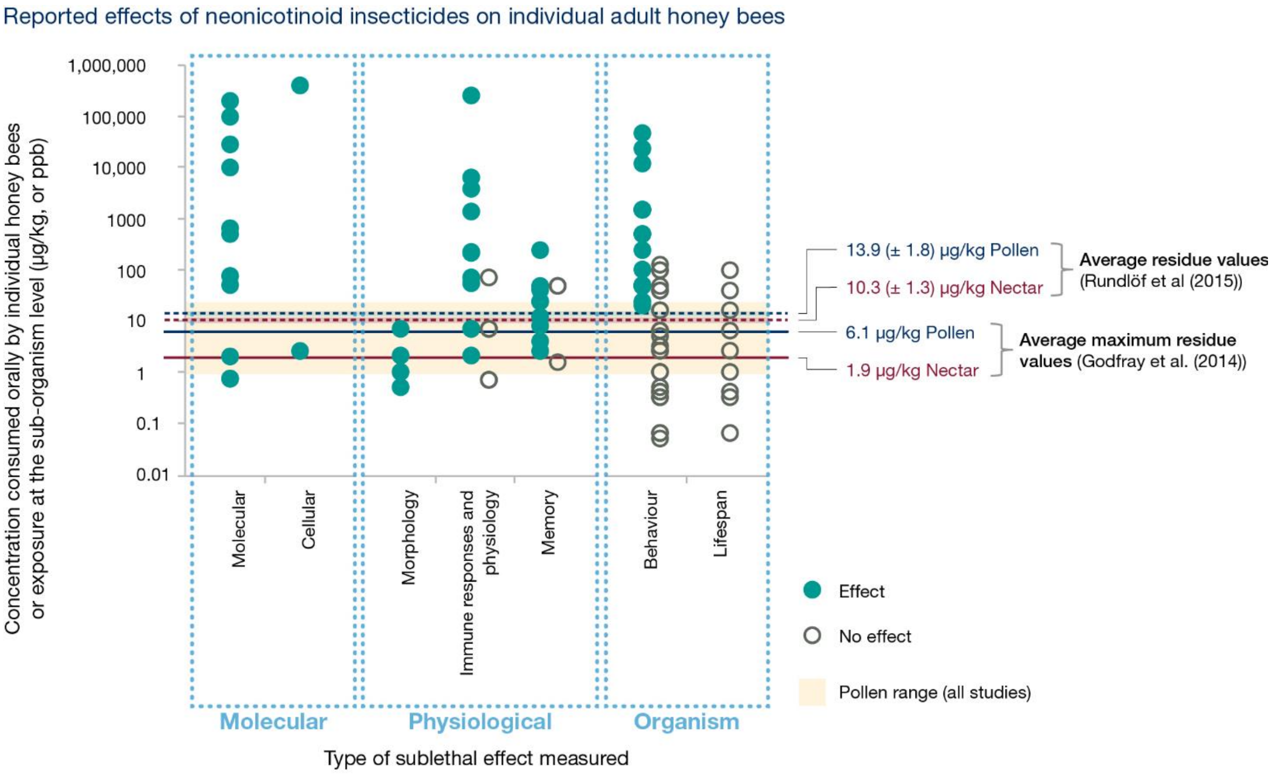
Despite the fact that these nine reviews overlap with respect to the papers they include, their conclusions are quite varied, though there is some commonality in the authors of the opposing views. Clear consensus exists regarding the fact that both wild and managed bees are exposed to pesticides (mainly through nectar and pollen, in the case of the neonicotinoids), and that the range of sublethal effects is quite broad. There is significant evidence and rather high agreement on the highly negative impacts of sublethal effects in controlled conditions.

However, some other topics are a matter of disagreement between the reviews, and in particular, over what constituted a field-realistic dose given pollinator traits, environmental context and management (Van der Sluijs et al., 2013; Carreck and Ratnieks, 2014). Thus, there are divergent views around the real effects of pesticides in field conditions, a knowledge gap that is attracting interest of different recent studies. In particular, Goulson (2015), when reanalyzing a study of the impacts of exposure of bumble bee colonies to neonicotinoids, showed a negative relationship between both colony growth and queen production and the levels of neonicotinoids in the food stores collected by the bees. Another study at wide field scale observed the effects of the clothianidin applied on spring sown oilseed rape in Sweden on managed honey bees and different wild bees (Rundlöf et al., 2015). They showed that this insecticide had no impact on managed honey bees but was reducing the density of wild bees, the nesting of the solitary bee *Osmia bicornis*, and the growth and reproduction of the bumble bee *B. terrestris* colonies. Though it is unclear whether the same results would be observed under different conditions (e.g. different crops, climates, or modes of agriculture) these results do show for the first time the effects of neonicotinoid insecticides in field conditions. These new data have a considerable importance, considering that oilseed rape is one of the main crops worldwide, and is highly attractive to bees, such that it competes successfully with other co-flowering vegetation for pollinator visits (Holzschuh et al., 2011; section 2.2.2.1.7).

Among the reviews published to date, four out of six (Cresswell, 2011; Van der Sluijs et al., 2013; Simon-Delso et al., 2014; Pisa et al., 2014) do conclude that sublethal effects of neonicotinoid insecticides on bees have negative consequences on their individual and social performances, suggesting their contribution to the decline of bees. Such consequences are potentially worsened by the fact that bees can be attracted by foods contaminated by neonicotinoid insecticides (Kessler et al., 2015). There is overall considerable evidence of sublethal effects of neonicotinoids on bees, but still low agreement on their in-field exposure levels and subsequent consequences, resulting in considerable uncertainty about how sublethal effects recorded on individuals (Figure 2.3.5) might affect the populations of wild pollinators over the long term. This knowledge gap makes it particularly difficult to assess how sublethal pesticide impacts affect the delivery and economic value of pollination services (Rundlöf et al., 2015; Raine and Gill, 2015). As highlighted by Johnson (2015) modeling may provide an approach to improve our understanding of the potential impact of sublethal effects on honey bee colonies (Becher et al., 2014) and other pollinators (Bryden et al., 2013).

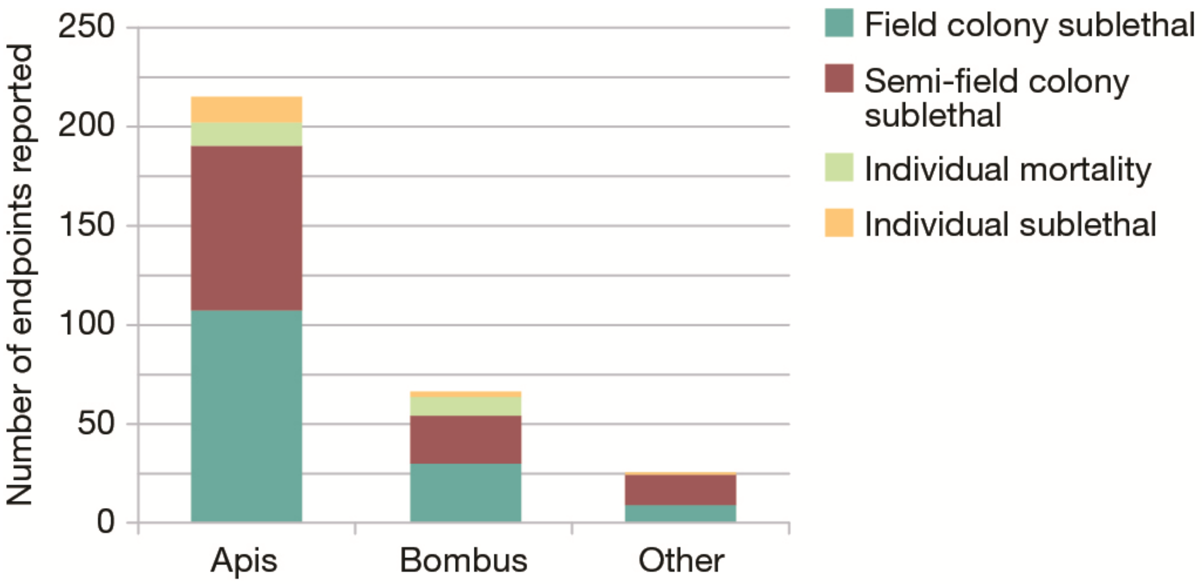
Finally, some of the reviews consider that synergistic and chronic effects have been widely underestimated, and should be studied much more.

Another issue is whether sublethal effects of pesticide exposure affect the provision of pollination. A recent study by Stanley et al. (2015) provided the first experimental evidence that neonicotinoid exposure can reduce the pollination delivered by bumblebees (*B. terrestris*) to apple crops. Flower visitation rates, amounts of pollen collected and seed set were all significantly lower for colonies exposed to 10 ppb thiamethoxam than untreated controls in flight cages. These findings suggest that sublethal effects of pesticide exposure can impair the ability of bees to provide pollination, which could have wider implications for sustained production of pollinator-dependent crops and the reproduction of many wild plants. Although currently there is no evidence of such impacts on pollination under field conditions (Brittain and Potts, 2011).



Reported effects of neonicotinoid insecticides on individual adult honey bees

11. Figure 2.3.5: Analysis of the reported oral exposure levels for three neonicotinoid insecticides (imidacloprid, clothianidin and thiamethoxam) resulting in sublethal effects or no effects at varying levels of organization in individual honey bees (sublethal effects data from literature reported in Fryday et al., 2015). Colony-level effects, including long-term effects, are not included. Molecular/cellular effects include those reported in in vitro systems. For reference, residue levels after seed treatments in field conditions range from 0.9 to 23 µg/Kg pollen and 1.9 to 16 µg/Kg honey, based on the subsequently cited references. A review of neonicotinoid residues in treated crops is provided in Blacquière et al. (2012) (average 0.9 -3.1 µg/Kg pollen, with levels in honey generally lower). Godfray et al. (2014) refer to average maximum residues in nectar following seed treatment of a range of crops as 1.9 µg/Kg (6.1 µg/Kg in pollen), but a recent study by Rundlöf et al. (2015) showed a mean of 10.3 ± 1.3 (range 6.7–16) µg/Kg nectar and mean 13.9 ± 1.8 (range 6.6–23) µg/Kg pollen following seed treatment of spring oilseed rape. Soil drench and foliar treatments may result in high residues depending on both application rate and pre-flowering application interval. Data from Cucurbitae has shown mean residues up to 11 µg/Kg nectar and 80 µg/Kg pollen following soil treatment and 8 µg/Kg nectar and 95 µg/Kg pollen following foliar treatment (Stoner and Eitzer, 2012; Dively and Kamel, 2012). Honey bee foragers consume 100% nectar; in-hive bees 84% nectar and 16% pollen (European Food Safety Authority (EFSA) 2013, United States Environmental Protection Agency (EPA, 2014).



12. Figure 2.3.6 Analysis of the numbers of reported sublethal endpoints at different levels of organisation reported for the neonicotinoid insecticides
(imidacloprid, clothiandin and thiamethoxam) conducted on *Apis*, *Bombus* and other bee species and the relative abundance of data on specific endpoints (excluding mortality) in honey bee individuals and colonies (as reported in Fryday et al., 2015).

2.3.1.5 Evidence of effects of pesticide mixtures

Pollinators may be exposed to mixtures of pesticides through a number of routes, including collection of nectar and pollen from multiple sources, storage of these in colonies of eusocial bees, tank mixes, and overspray of crops in flower where systemic residues are present in nectar and pollen. In addition, honey bees may also be exposed to beekeeper-applied treatments such as antibiotics and varroacides (Chauzat et al., 2009; Mullin et al., 2010) There is evidence of multiple residues of pesticides detected in bees, honey, pollen and wax within honey bee colonies (e.g. Thompson, 2012) but these data are complex in terms of the number, scale and variability of pesticide residues. Data are very limited or absent for other pollinators and for the effects of complex pesticide mixtures.

There is strong evidence that when combinations of pesticides have been screened in a range of aquatic invertebrates (Verbruggen and van den Brink, 2010; Cedergreen, 2014), synergistic interactions (resulting in greater than 2-fold increase in toxicity when compared with concentration addition) were rare (7%) and 95% of these could be predicted based on their mode of action, e.g. ergosterol biosynthesis inhibitor (EBI) fungicides and pyrethroids (Cedergreen, 2014). For the remainder the effects were at worst additive with many combinations showing no significantly increased toxicity or even antagonistic effects. This limited evidence of synergistic interactions, other than those through deliberately applied insecticide synergists (such as piperonyl butoxide or mixtures of insecticides (Andersch et al., 2010), is also evident in honey bees, and the vast majority of the literature relates to synergistic interactions resulting from EBI fungicide exposure (Thompson, 2012; Glavan and Bozic, 2013). The first evidence of unintended synergistic interactions in honey bees with increases in toxicity (decrease in LD_{50}) of up to 1,000-fold was that between EBI fungicides and pyrethroids (Colin and Belzunces, 1992; Pilling, 1992; Pilling et al., 1995; Johnson et al., 2006) and was identified as due to the inhibition of the P450s, responsible for pyrethroid metabolism. More limited evidence has identified the potential for synergism between the EBI fungicides and neonicotinoid insecticides (Schmuck et al., 2003; Iwasa et al., 2004) through the same mechanism, with reported increases in toxicity up to 500-fold. However, there is also evidence that the scale of synergism observed is dose-related with a low or no increase in toxicity at field-realistic dose levels (Thompson et al., 2014). There is some evidence that effects in honey bees at the nerve synapse receptor level between organophosphorus and neonicotinoid insecticides are additive (Palmer et al., 2013), and effects of lambda-cyhalothrin and imidacloprid on colony performance are additive in the bumble bee *B. terrestris* (Gill et al., 2012), as may be expected from the differing mode of action of these compounds. There is also limited evidence of the interactions between veterinary medicines used in honey bee colonies, such as varroacides (Johnson et al., 2013) with some evidence that other classes of pharmaceuticals, such as antibiotics, interacting with multi-drug resistance membrane-bound transporter proteins may result in significantly increased toxicity of varroacides (Hawthorne and Diveley, 2011).

2.3.1.6 Evidence of honey bee colony losses due to pesticide use from national monitoring programmes

National monitoring approaches have been undertaken to address directly the role of pesticides in over-wintering honey bee colony losses. To date, these have concluded that colony loss is a multifactorial issue with the predominance and combination of different drivers varying in space and time (section 2.7). There is no clear evidence that pesticides, particularly the neonicotinoid insecticides, have directly contributed to these longer-term colony losses in the EU or US (Chauzat et al., 2006a; Chauzat et al., 2006b; Chauzat and Faucon, 2007; Chauzat et al., 2009; Nguyen et al., 2009; vanEngelsdorp et al., 2009b; Chauzat et al., 2010; Genersch et al., 2010; Chauzat et al., 2011; Rundlöf et al., 2015), with the most recent statistically robust field study by Rundlöf et al. 2015 supporting this conclusion. In some cases (Orantes-Bermejo et al., 2010; vanEngelsdorp et al., 2010)), however, the residues of the most frequently suspected pesticides (e.g. neonicotinoids) were not analysed using methodology with sufficiently low limits of detection (LOD) and limits of quantification (LOQ). In addition, some studies have highlighted fungicides as a factor affecting honey bee health adversely, although their role in colony losses have not yet been demonstrated (vanEngelsdorp et al., 2009a; Simon-Delso et al., 2014). The mode of action underlying this observation is currently unclear. There is some evidence that fungicide exposure may result in decreased nutritional contribution of bee bread (processed pollen) by reducing the diversity of fungal spores returned to the hive and by affecting the diversity and growth of fungi present in bee bread and thus its fermentation (Yoder et al., 2013).

Box 2.3.5: Assessing the possible contribution of neonicotinoids to pollinator declines: What do we still need to know?

To date the role of neonicotinoids in pollinator declines has been a particularly polarised debate. There are both qualitative and quantitative aspects, so what evidence do we need to inform the debate?

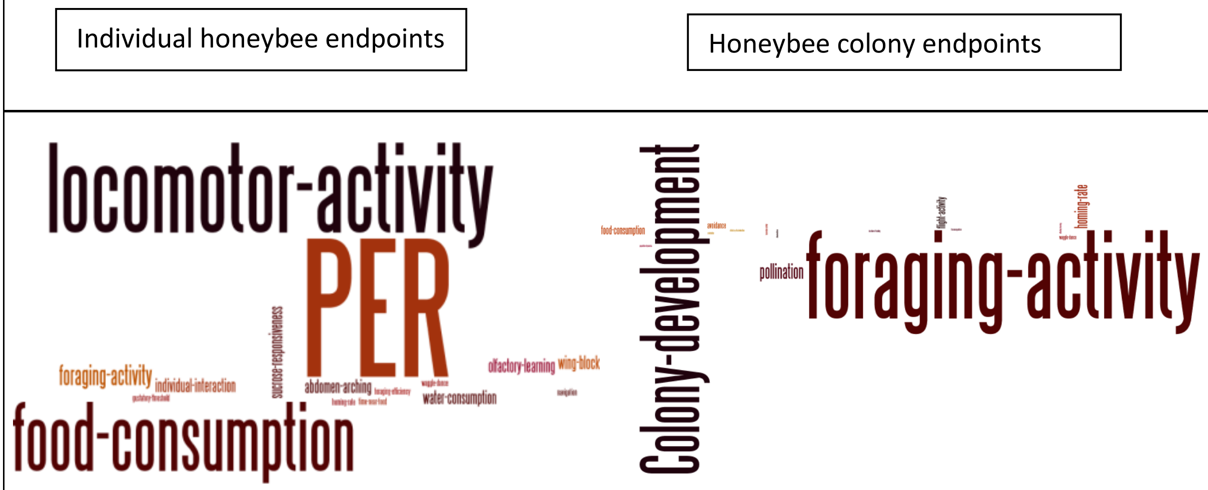
Where declines in species and possible drivers have been identified but not prioritised, we need to weigh the evidence carefully, and identify which are the key gaps (e.g. (Van der Sluijs et al., 2013; Godfray et al., 2014; Lundin et al., 2015). Where the evidence is still scant, Hill's epidemiological criteria can be used to identify whether the logic criteria (coherence, plausibility, gradient) coincide with the circumstantial epidemiological evidence, e.g. for honey bee declines (Cresswell et al., 2012a; Staveley et al., 2014). Such an analysis both identifies knowledge gaps, but also helps to differentiate between the differing drivers of declines. For example declines of bumble bees in the 1950s were certainly not initiated by neonicotinoids, but probably due to loss of flower-rich habitat with agricultural intensification (Ollerton et al., 2014).

Apart from dust generated during drilling of treated seed or off-label applications, national incident monitoring schemes suggest approved neonicotinoid use has not been associated with honey bee mortality. However, vigilance is needed to ensure that approved uses include mitigations to protect pollinators and the environment (e.g. buffer zones to off-crop areas, not applying to bee-attractive crops in flower or crops containing flowering weeds) and that use instructions are clear, understood and respected. Concerns have arisen primarily from acute or chronic sub-lethal exposures that might interfere with foraging, orientation and learning abilities and other behavioural characteristics of pollinators, as well as with the immune system at the individual and colony level.

There remain some key gaps in our knowledge:

1. Toxicity. There are large differences in the toxicity of neonicotinoids in honey bees, e.g. thiacloprid and acetamiprid vs. imidacloprid, clothianidin and thiamethoxam as well as their metabolites (Blacquière et al., 2012). Although, with appropriate assessment factors, acute (lethal) toxicity data for honey bees can be used as a surrogate for other species (Hardstone and Scott, 2010; Arena and Sgolastra, 2014), large differences in species sensitivity may occur (as for other invertebrates, e.g. *Cloen* (Mayfly) compared to *Daphnia* (Roessink et al., 2013)). The ability of bees to detoxify and excrete ingested neonicotinoid residues contributes to species differences in their chronic sensitivity (Cresswell et al., 2012b; Laycock et al., 2012; Cresswell et al., 2014). Therefore further data are required especially for wild pollinator species, to confirm that extrapolation between species is appropriate for neonicotinoids and their metabolites (Lundin et al., 2015).

Even less is known about sub-lethal toxicity, e.g. at which doses are no effects found, which effects are important for which species (see Figures 2.3.5 - 2.3.7 (Lundin et al., 2015))? For example, there is a plausible potential for interactions between sub-lethal exposure to neonicotinoids and foraging efficiency, resulting in effects at the colony level for species with low numbers of foragers (Rundlöf et al., 2015). The Rundlöf et al. (2015) study showed that, whilst there were no effects on honey bee colonies, exposure to flowering spring-sown oilseed rape grown from seed treated with the highest approved application rate of clothianidin in Sweden affected bumble bee colony development, *Osmia* nest establishment and the abundance of wild bees observed foraging on the crop. The residue levels in pollen and nectar were higher than previously reported in oilseed rape (Blacquière et al., 2012; Cutler et al., 2014a; Godfray et al., 2014) and highlight the need for understanding of the variability of pesticide residue levels in crops. For example, in Europe, varieties of oilseed rape sown in the autumn/winter are far more prevalent than spring-sown varieties. Autumn/winter sown varieties are often treated with lower levels of neonicotinoid and the time from sowing to flowering is about 7-8 months, rather than 3-4 months for spring varieties. However, these results are of considerable importance, because they show for the first time the effects under field conditions of a neonicotinoid insecticide on wild bees in the absence of an effect on honey bees. In order to quantify the possible contribution of these sublethal effects to the observed declines we need not only to test at levels that result in these effects under laboratory conditions, (Figure 2.3.5) but also at field-realistic exposure levels and profiles (Lundin et al., 2015). Such an approach may use designs similar to that of Rundlöf et al. (2015) to evaluate the effects on managed and wild bee populations of the most widely used insecticides, applied according to their approved use, in the most widely grown pollinator attractive crops.



13. **Figure 2.3.7 Relative abundance of data on specific memory, behavioural, morphological, physiological and molecular effect endpoints (excluding mortality) in honey bee individuals and colonies**
(as reported in EFSA, 2015)

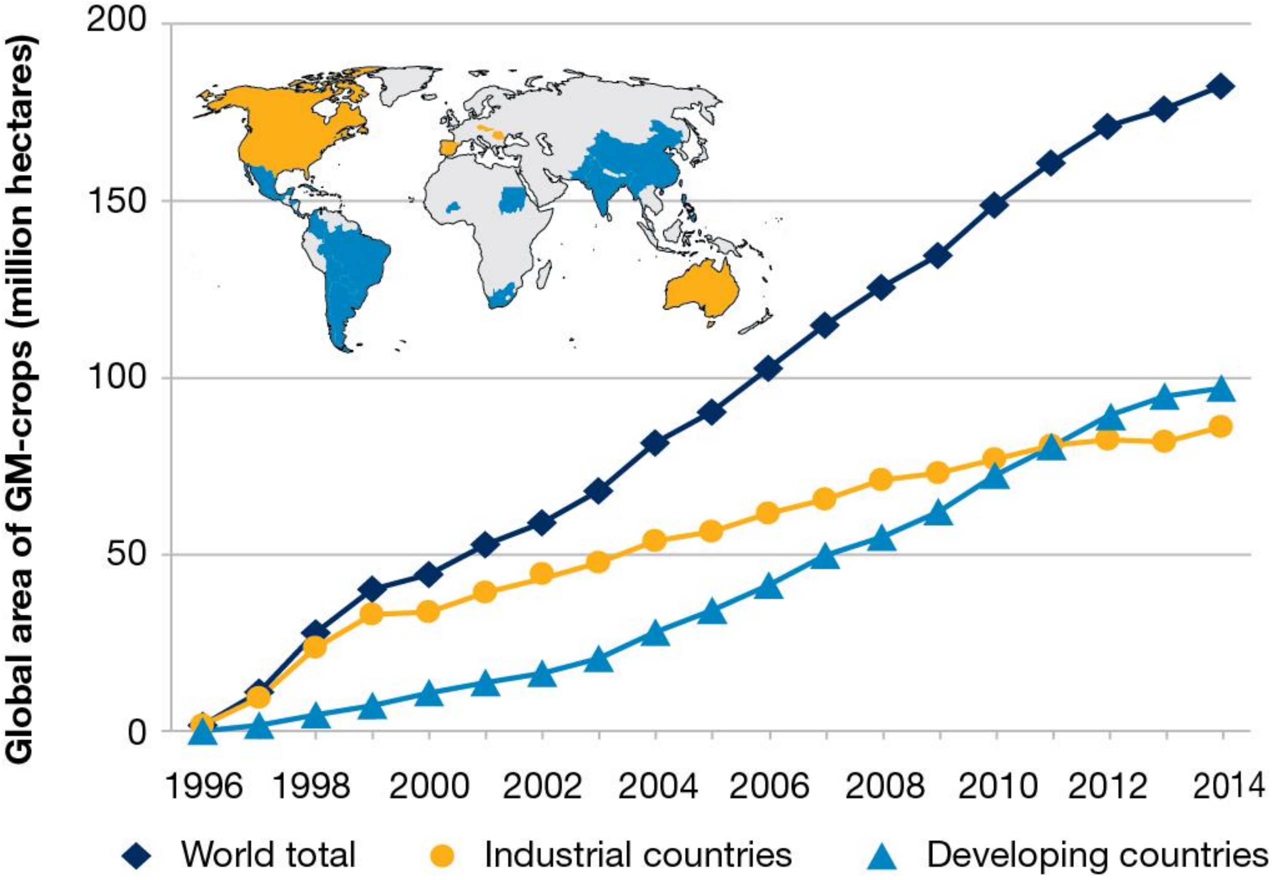
2. Exposure. To quantify field-realistic exposure levels we need to estimate both the potential total exposure to residues (parent and relevant metabolites), e.g., via pollen and nectar, and understand the relative consumption of these versus consumption of uncontaminated sources, because contaminated food will often form only part of the total available food resources within the landscape (Lundin et al., 2015). It is important to know what the impact is of the chemicals as applied in the field or in residential or amenity use at the colony or population level. What are the residue levels in different compartments of the plant, after real field applications and in subsequent crops grown on the treated fields, how do these translate into levels in pollen and nectar, and what are the consequences for the exposure of adult bees and larvae of different bee species, e.g. species that feed their larvae raw pollen versus processed brood food?

3. Interactions & Synergisms. What are the consequences of the sublethal effects of neonicotinoids with additional additive or synergistic stressors? A key area of challenge is the need to study the effects of realistic combinations and scales of stressors, some of which are not readily manipulated, e.g. pesticides and disease within the honey bee colony (Goulson et al., 2015; Lundin et al., 2015). Modelling (Bryden et al., 2013; Becher et al., 2014) may provide an opportunity to study both the potential interactions of such sublethal effects with each other and the effects of other factors, e.g. landscape, climate, as drivers of pollinator decline (Kielmanowicz et al., 2015).

2.3.2 GMO cultivation

2.3.2.1 Introduction

Genetically modified (GM) organisms (GMOs) are organisms that have been modified in a way that does not occur naturally by mating and/or natural recombination (FAO/WHO, 2001). One of the most common methods to do this is by bioengineering transgene(s) into the new organism. The most common plant transgenes confer herbicide tolerance (HT), or toxicity towards herbivores (insect resistance, IR), although other characteristics have been also engineered (e.g., drought resistance in wheat, nutritional values in sorghum; James, 2014). As of 2014, several GM crops were commercialized and grown in 28 countries, representing around 12% of the world’s arable land, an equivalent of 181.5 million ha (Figure 2.3.8, James, 2014; Li et al., 2014a). The most widely commercialized GM crops are maize, cotton, canola (oilseed rape) and soybean, which currently have varieties that can display one or both IR and HT. Other less widespread crops are already available and cultivated, such as sugar beets, papaya, alfalfa or rice (James, 2014).



14. Figure 2.3.8. Distribution and uptake of GM-crop production from 1996 to 2014
Yellow: total hectares; blue: hectares in industrial countries; red: hectares in developing countries. Green in map caption: lands growing GM-crops. Modified from James (2014).

The most common HT crops confer resistance to the herbicide glyphosate (Schwember, 2008), engineered through the introduction of an *Agrobacterium* (bacterial) enzyme gene (Funke et al., 2006).

All currently grown IR-crops express insecticidal proteins engineered from the bacterium *Bacillus thuringiensis* (Bt-toxins; mainly Crystalin –Cry– and Vegetative Insecticidal Proteins –Vip) (Gatehouse et al., 2011). The toxicity of these proteins is relatively taxon-specific, generally against Lepidoptera or Coleoptera. Non-Bt insecticidal proteins have been bioengineered from other non-bacterial organisms (e.g., alpha-amylase inhibitors, lectins, biotin-binding, fusion proteins; Malone et al., 2008; Vandenborre et al., 2011) and allow expanding the breadth of IR, as well as dealing with Bt-toxin resistance. Because these latter crops are not currently commercialized, their impact on pollinators will not be presented here.

In the framework of GMO production, pollinators are considered non-target organisms. Prior to commercialization, all GM-crop varieties are assessed for environmental risk. Effects of GM-crops of non-target organisms are generally tested on surrogates, species considered representative of the ecological function in question. In the case of pollinators, these species have been the honey bee (*A. mellifera*), *Osmia bicornis* and *B. terrestris*, and ladybird beetles (*Coleomegilla maculata*, *Adalia bipunctata* and *Coccinella* *septempunctata*) (Li et al., 2014b). When evaluating the potential effect of GMOs on pollinators, one should consider two types of effects: direct and indirect.

2.3.2.2 Direct effects

Exposure to the transgenic trait in IR crops has the potential to affect insect pollinators directly (Malone and Burgess, 2009). Thus, risk assessment procedures related to GMO release, cultivation and production have involved studies that assessed the toxicity of the transgenic proteins or transgenic tissue to insect pollinators (Andow and Zwahlen, 2006; Li et al., 2014b).

Pollinators consume pollen and/or nectar, and because the transgenes are expressed in both (Abrol, 2012; Malone and Burgess, 2009; Paula et al., 2014), their ingestion could potentially lead to reduced survival or behavioral/physiological disturbances. To test this, laboratory (reviewed in Li et al., 2014b; Paula et al., 2014), greenhouse (e.g., Arpaia et al., 2011; Hendriksma et al., 2013; see Malone and Burgess, 2009 for a review) and field (e.g., Hendriksma et al., 2013; Martins et al., 2008; reviewed in Malone and Burgess, 2009) studies were performed by either feeding pollinator larvae and/or adults with diets supplemented with the purified toxin or a quantified amount of GM-pollen or nectar, or by allowing the pollinators to harvest and consume GM-plant products from natural or semi-natural environments. These studies were carried out on a diverse array of pollinator taxa, such as Hymenoptera, Lepidoptera and Coleoptera. Toxicity against Diptera pollinators has never been tested and this remains an important knowledge gap.

Results from these studies vary based on the target group and the toxin concentration. Bt-toxins are non-lethal to Hymenoptera and their colonies (Abrol, 2012; Babendreier et al., 2008; Devos et al., 2012; Duan et al., 2008; Hendriksma et al., 2013; Li et al., 2014b; Malone and Burgess, 2009; Mommaerts et al., 2010). However, sub-lethal effects (see section 2.3.1.4) were reported in one study. In particular, ingestion of high concentrations of Bt-toxins (close to those found in some transgenic varieties, such as NaturGard KnockOut, Fearing et al., 1997) affected the behavior (however, see Arpaia et al., 2011 for a study where no behavioral difference was observed in bumble bees) and learning in honey bees, although there was no effect at lower toxin concentrations (such as those found in other transgenic varieties; Ramirez-Romero et al., 2008). As expected, toxins were shown to lead to reduced larval survival and body mass, and increased developmental time in Lepidoptera (Lang and Otto, 2010; Paula et al., 2014) (Table 2.3.4). In an environmental risk framework, European studies modelling the potential exposure and consumption of Bt-pollen by wild butterflies provided ambiguous results (e.g., Holst et al., 2013; Perry et al., 2013 and references therein), and more experimental research may be needed to resolve this issue. Finally, Cry1C and Cry2A Bt-toxins did not affect the larval development and survival of several pollen-feeding ladybirds (Li et al., 2015; Li et al., 2014b).

7. Table 2.3.4 – Summary of results for tested negative effects of insecticidal proteins on different insect pollinators.

"No" indicates no negative effects identified; "Yes" indicates negative effects identified; "Yes/No" indicates that the effects were identified on some species or particular developmental stages only; "NT": not tested. Cry: Crystalin proteins, Vip: Vegetative insecticidal proteins, E: empirical, R: review, MA: Meta-analysis.

Pollinator group	**Insecticidal protein**		**Publication**	**Details**
	Cry	**Vip**		
Hymenoptera (bees & wasps)	no	NT	Babendreier et al., 2008 (E)	*Bombus terrestris* microcolonies fed with purified Cry1Ab and SBTI
	no	no	Malone and Burgess, 2009 (R); Romeis et al., 2009 (R)	Several Hymenoptera groups fed with different Cry, Vip and non-Bt proteins
	no	NT	Konrad et al. (2008) (E)	Larvae of *Osmia bicornis* fed with Cry1Ab toxins
	yes/no	NT	Ramirez-Romero et al., 2008 (E); Devos et al., 2012 (E); Hendriksma et al., 2013 (E)	Sublethal effects of purified Cry1Ab on *Apis mellifera* adults; *A. mellifera* larvae fed with purified Cry3Bb1 proteins; composition of gut bacterial community of *A. mellifera*
	no	NT	Mommaerts et al., 2010 (E); Arpaia et al., 2011 (E)	Lethal and sublethal effect of Bt-formulations on *B. terrestris* microcolonies; foraging behavior on Bt-plants
	no	NT	Li et al., 2014b (R); Duan et al., 2008 (MA)	Several Hymenoptera stages treated with different purified Cry proteins
Lepidoptera (butterflies & moths)	yes/no	NT	Lang and Otto, 2010 (R)	Different Lepidoptera groups fed with Bt-pollen or purified Cry proteins
	yes	NT	Paula et al., 2014 (E)	Transgenerational effect in Cry1Ac-fed *Chlosyne lacinia*
	no	no	Romeis et al., 2009 (R)	Several Lepidoptera groups fed with different Cry, Vip and non-Bt proteins
Coleoptera (beetles)	no	NT	Li et al., 2014b (R), Li et al., 2015 (E)	Larvae of different ladybird species reared with Bt-pollen and purified Cry1C and Cry2A proteins

2.3.2.3 Indirect effects

These effects include those affecting pollinators, either through indirect contact with the GM-crop or parts of it, or through changes in the agroecosystem and/or agricultural practices (see section 2.2.2) associated with the GM-crop production. These latter changes can potentially lead to alterations in ecological communities, associated with changes in food or interaction webs, or population and follow-on effects if transgene flow from the GM-crop into non-GM- or wild (ancestor) species occurs.

2.3.2.3.1 Effects of GM-crops on the use of agrochemicals

One of the arguments supporting GM-crop production is its potential to reduce the use of agrochemicals (Brookes and Barfoot, 2013; Naranjo, 2009), especially insecticides. Indeed, because insecticides are produced by the plant itself, one would expect a reduced need to make further applications on the field. Although there is overall significant global reduction in insecticide applications (41.67% less insecticide applied in IR-crops compared to conventional; Klumper and Qaim, 2014; Brookes and Barfoot, 2013), the pattern varies depending on the crop species and the geographic region of the world, and is not affected by insecticide seed treatments. For instance, whereas global insecticide use was reduced by 45.2% for GM-maize, this reduction appears to be larger in the USA (42%) than in Argentina (stated as “very small”, based on the low background insecticide consumption of 1-2/ha in that country; Brookes and Barfoot, 2013). This can be explained by the fact that some crops can be affected by a large array of pests, some of which are not sensitive to the transgenic toxin (e.g., stink bugs in GM-cotton in the USA, Naranjo, 2009; mirid bugs in GM-cotton in China, Lu et al., 2010), or by the fact that the pests targeted by transgenics do not represent an important threat in particular regions (e.g., stem borer in GM-maize in Argentina; Brookes and Barfoot, 2013).

In a case of a reduction in insecticide applications, an increase of insect biodiversity in GM crops is expected (see section 2.3.1). This has been investigated under field conditions, and results demonstrate that insect communities on Bt-crops are overall more diverse than those on insecticide-treated non-Bt-crops (but not necessarily less than untreated non-Bt-crops; Marvier et al., 2007), and this situation holds for several types of Cry- and Vip-expressing crops (e.g., maize, cotton, potato), and at the global scale (Marvier et al., 2007; Naranjo, 2009; Whitehouse et al., 2014; Whitehouse et al., 2007). Because pollinators are included in these insect communities, IR-crops could be beneficial to pollinators.

HT-crops management is based on the idea that regular applications of herbicides will very likely be done in the field. As in any herbicide application, this will lead to weed reduction and potential toxicity towards pollinators (see section 2.2.2.1). Although weed eradication is of high agronomic interest, many generalist pollinators, including crop pollinators, exploit these weeds as pollen and nectar sources (see section 2.2.2.1.4). The limited evidence obtained from the few studies investigating this indicates that HT-crops can lead sometimes to a general reduction of pollinators in the fields, such as shown for beets and oilseed rape (e.g., Abrol, 2012; Bohan et al., 2005; Haughton et al., 2003), or as suspected for monarch butterflies (see Box 2.3.6). Such a reduction could lead to increased pollination deficits and yield reduction in crops benefiting from pollinators (e.g., oilseed rape, cotton). The evidence for this is very limited, due to a lack of studies, and to our knowledge only one study has investigated and confirmed this expectation (for oilseed rape, Morandin and Winston, 2005).

2.3.2.3.2 Transgene flow

Concern has been raised on the possibility of transgene escape and persistence in non-GM-crops and wild plants through hybridization and/or introgression (Kwit et al., 2011; Stewart et al., 2003). Indeed, all the engineered plants have wild ancestors or closely related species with which they can, and most of the times do, hybridize (Letourneau et al., 2004). While the risk of transgene flow is minimal when these wild species are not present in the area where the crops are being cultivated, this is not necessarily always the case. Although introgression events of these genes have been very rarely observed, they have been shown to be theoretically possible (e.g., Meirmans et al., 2009) and recent molecular investigations have identified the presence of transgenes in wild ancestors (e.g., in canola, wild cotton and maize; Pineyro-Nelson et al., 2009; Warwick et al., 2008; Wegier et al., 2011), sometimes far from the known contact zone (e.g., wild cotton in Mexico; Wegier et al., 2011).

Besides the potential of herbicide-resistant weed formation (Mallory-Smith and Zapiola, 2008), introgression and transgene expression in wild relatives can shift the previously mentioned direct effects of GMOs (see section 2.3.2.2) into the wild, potentially disturbing insect and pollinator communities in non-agricultural environments, affecting survival of other non-target species, and altering ecological networks (see above). It was shown that Lepidoptera herbivore survival is reduced after introgression of insecticidal transgenes into the wild relatives of sunflowers, and that this leads to higher seed set in the introgressed plants, which favors their spread (Snow et al., 2003). Although there is a lack of evidence on the real extent and consequences of such gene introgressions and spread in the wild, the ecological and evolutionary consequences of such an event for wild pollinators and pollination can be non-negligible (e.g., diminished adult/larval survival for leaf- or pollen feeding pollinators, reduced pollination). From that perspective, this is an important knowledge gap.

2.3.2.4 Effects on pollination

If the GM crop is animal-pollinated and has a negative effect on pollinators, this may potentially affect its yield. Studies demonstrated that some of the main GM-crops are partially animal pollinator-dependent, i.e., display increased yield in the presence of pollinators (e.g., soybean, cotton; Malone and Burgess, 2009), which agrees with general patterns demonstrated for other non-GM crops (e.g., Klein et al., 2007). Thus, there can be a risk of yield loss if pollinators are less abundant in some GM-fields (see 4.3.1), although this effect could be out-weighed by the benefits obtained through the loss of herbivore and weed pressure associated with the transgene expression (Christou and Capell, 2009).

Box 2.3.6: GM-crops in the US Midwest and monarch butterflies

The topic of the effect of GM-production on pollinators is complex, mainly because of the many direct and indirect variables that it involves. The case of the monarch butterfly in North America represents a good example of this complexity.

The monarch butterfly *Danaus plexippus* has a strong cultural value in North America. Much admiration surrounds this species, particularly because of its massive annual migrations between the USA, Canada and Mexico. After overwintering in Mexico, the Eastern monarch population moves mainly to the US Midwest, where it reproduces. For reproduction, monarchs depend on milkweeds (*Asclepias* spp.), on which larvae specifically feed and develop.

Because of the Lepidoptera-specific toxicity displayed by Bt-maize, the arrival of this crop to the US Midwest worried naturalists and the general public. *Asclepias* grow close to or within crop fields, so GM-pollen deposition on milkweed leaves could represent a risk for the monarch larvae. Thus, researchers evaluated whether monarch larvae could be affected by the ingestion of field-relevant amounts of GM-pollen (Hansen Jesse and Obrycki, 2000; Losey et al., 1999). Their laboratory results raised much concern, because the treatment reduced larval growth rates and increased mortality. However, when the tests were done in natural conditions, it was shown that although the pollen is toxic for the monarch larvae, it is very unlikely that they contact it, because larval development and maize flowering are not simultaneous (Sears et al., 2001; reviewed in Oberhauser and Rivers, 2003).

The discussion on the effect of GM-crops on monarchs had more or less ended until recently. Indeed, monarch populations arriving from the USA to Mexico have been particularly reduced in recent years (Rendón-Salinas and Tavera-Alonso, 2014). Because most of the monarchs arriving to Mexico migrate from the US Midwest, GM-crop production was again suspected to be associated with that population reduction, but in a more indirect manner. Indeed, the Midwest has seen increased glyphosate use associated with the expansion of HT-maize and soybean. Glyphosate applications could lead to a reduction of the milkweed population, and thus to smaller monarch populations. To test this, and to investigate further the non-significant results obtained by Davis (2012) in two Eastern US populations, Pleasants and Oberhauser (2013) combined historical land use (i.e., yearly area occupied by milkweed habitats from 1999 to 2010) and biological (i.e., number of monarch eggs per milkweed plant and density of milkweeds in different land cover types) data to estimate the number of monarch eggs laid per year. Their study identified a significant correlation between such estimates and overwintering population sizes, suggesting that both the widespread use of glyphosate and the strong GM-cropland expansion in the US Midwest could explain the changes in butterfly population sizes. Along with this, a recent study (Flockhart et al., 2015) used modelling approaches to identify the factor most strongly affecting the monarch population size. Their results indicated that habitat loss (see section 2.2.1) associated with the expansion of GM-crops in the USA is the strongest predictor of demographic changes in monarch butterflies.

2.3.3 Veterinary medicines

The use of veterinary medicines to control pests and diseases in pollinators is primarily restricted to honey bees. Non-chemical alternatives based on traditional knowledge are also being investigated (Singh, 2014; Simenel, 2015). There is strong evidence that chemical use is widespread in beehive management to control pests such as *Varroa destructor*, e.g., fluvalinate, coumaphos, and amitraz, and for diseases such as European and American foulbrood, e.g., oxytetracycline (Bogdanov, 2006; Reybroeck et al., 2012). In some continents the use of antibiotics is highly regulated, but there is also evidence that significant levels of various beekeeper-applied pesticides and antibiotics are present in hive matrices such as bee bread, honey and wax (Chauzat et al., 2009; Bernal et al., 2010; Mullin et al., 2010; Orantes-Bermejo et al., 2010; Reybroeck et al., 2012; Wei et al., 2013). Highly lipophilic chemicals may also accumulate in wax within the colony (Bogdanov, 2006; Bonzini et al., 2011). There is more limited evidence that beekeepers may use unauthorised (illegal) products for pest and disease control (many also have agricultural uses as insecticides or acaricides) with inherent risks of resistance and food safety (Bogdanov, 2006; Chauzat et al., 2011; Riscu and Bura, 2013). There is increasing evidence of the adverse effects of the chemical varroacides on honey bees, e.g. reductions in queen fecundity and sperm quality (Pettis et al., 2004; Johnson et al., 2013) and that antibiotics used to combat foulbrood are toxic to adults and larval honey bees (Thompson et al., 2006; Hawthorne and Diveley, 2011). There also is evidence of interactions between organophosphorus and pyrethroid varroacides (Johnson et al., 2009) and there is the potential for mixtures of veterinary medicines with pesticides to result in increased toxicity, e.g., pyrethroids such as fluvalinate with EBI fungicides highlighted above (Thompson, 2012) but there is currently no evidence of interactions occurring within hives between pesticides and veterinary medicine residues. Such adverse effects of veterinary medicines have potential consequences for the contribution of honey bees as pollinators and should be balanced against the beneficial consequences of pest and disease control, but currently there is limited evidence on which to base this.

2.3.4 The effect of pollution on pollinators

Pollution is a problem of wide concern. Industrial pollutants, like heavy metals, toxic chemicals such as arsenic or selenium washed out by irrigation, or other non-chemical pollutants are clearly affecting pollinators and their influence is increasing due to industrialization, agriculture and urbanization. During the last century, for example, production of heavy metals increased ten-fold and also levels of their emission (Nriagu, 1996). Currently efforts are focused globally on gradual reduction and prevention of pollution (for detailed information and reports see: AMAP, 2002; EEA, 2009; HELCOM, 2013; OSPAR, 2009). However, the rapidly industrializing countries in Asia are nowadays facing the problem of quickly growing pollution (Indian National Science Academy, 2011).

2.3.4.1 Heavy metals, arsenic and selenium

Heavy metals, both non-essential (e.g. lead and cadmium – toxic in all amounts) and essential ones (like e.g. zinc – toxic in excess) can weaken an organism by changing the conformation or causing the denaturation of enzymes (Deplegde et al., 1997). Impact of heavy metal pollution on insect pollinators has not been widely investigated, although it is well studied in other groups of invertebrates (for a review see Tyler et al., 1989). Pollinators are exposed to heavy metal contamination by various ways: air, soil and water pollution, but also through pollution of floral rewards due to hyper-accumulation of heavy metals in plants. Studies show a large variety of strategies used to cope with the effects of heavy metal pollution, and also various levels of susceptibility to contamination, making it difficult to foresee how a species – previously unstudied – may react to heavy metals in its environment (for a review see: Tyler et al., 1989). It has been demonstrated that metals, such as cadmium, copper, iron, manganese, zinc may play a direct role in the widespread decline, e.g., the butterfly Parnassius apollo in Finland (Nieminen et al., 2001). Oddly, there are no detailed studies concerning the effects of heavy metal pollution on honey bee physiology. Yet, there are numerous papers using honey bees (Van Der Steen et al., 2012) and their hive products (Conti and Botrè, 2001) as good indicators of environmental pollution levels, implying that honey bees are directly exposed to pollutants. Evidence is also scarce on how heavy metal pollution affects other bee species, but a recent study on bumble bees suggests that some soil pollutants (e.g. aluminium or nickel) could cascade to affect bees negatively in contaminated areas (Meindl and Ashman, 2013). In 2012, Moroń et al. detected a steady decrease in the number, diversity and abundance of solitary wild bees along heavy metal gradients in Poland and the UK. While in 2013 Moroń et al. also found a direct negative impact of zinc contamination on the survival of the solitary bee *Osmia bicornis* along this pollution gradient. Bees had fewer offspring with a higher mortality rate with increasing pollution level and also the ratio of emerging males and females in offspring was changed, due to probably higher mortality of males, with increasing contamination. Whereas Szentgyörgyi et al. (2011) did not find a significant correlation between heavy metal pollution level of the environment (with cadmium, lead and zinc) and the diversity of bumble bee species caught on Polish and Russian heavy metal gradients. Despite the small number of available studies, in a questionnaire undertaken by Kosior et al. (2007), specialists considered heavy metal pollution to be one of the more important factors associated with bumble bee decline in Europe (ranked 6th of 16 stressors surveyed).

Besides heavy metal pollution, there is a growing concern about non-metal pollutants, e.g., arsenic or selenium. Arsenic occurs as by-product of coal and other ore mining, including copper production. Air pollution by arsenic was shown to destroy honey bee colonies near an arsenic discharging electrical plant (for review see Lillie, 1972). Selenium, on the other hand, is an essential trace element, but as with most trace elements it is toxic in high concentrations. Due to mining and other industrial activities, as well as through drainage water from irrigation of seleniferous soils, some areas are highly contaminated. In the environment selenium bioaccumulates and therefore bees may be at risk through the biotransfer of selenium from plant products such as nectar and pollen (Quinn et al., 2011). Recent studies showed that selenium increased mortality in honey bee foragers (Hladun et al., 2012) and negatively affected larval development (Hladun et al., 2013).

Bee larvae feed mainly on pollen (Michener, 2000); thus, in polluted sites, they may consume food that is contaminated with heavy metals or other pollutants. The main source of pollution of pollen is probably soil dust deposited on flowers or on the pollen during transport to and placement in the bee's nest (Szczęsna, 2007), and probably hyperaccumulation of pollutants by plants in floral rewards (Hladun et al., 2011; 2015). This suggests that both soil type and flower type can affect the deposition of pollutants, such as heavy metals on pollen (Szczęsna, 2007). For bee species nesting in the ground, the impact of pollution may be larger because besides pollen, larvae can also come into contact with contaminated soil during their development. Sociality may also affect susceptibility to pollution: a hierarchy in the nest protects reproducing individuals (queens) from pollution, therefore allowing the colony to reproduce (Maavara et al., 2007). This phenomenon was already described in ant colonies, in which individuals had lower levels of pollutants in their bodies' concomitant with higher positions in the nest hierarchy (Maavara et al., 2007). This might explain why honey bees can be used as good indicators of environmental pollution for even relatively high levels of pollution (Rashed et al., 2009). In solitary species such protection of reproducing females is simply lacking and therefore they might be more susceptible to pollution, as shown by the contrasting result of Moroń et al., in 2012 on bee diversity and Szentgyörgyi et al., in 2011 on bumble bee diversity on similar gradients of heavy metal pollution.

2.3.4.2 Nitrogen deposition

Besides the aforementioned heavy metals and non-metals, another driver that has also received relatively little attention to date is atmospheric nitrogen deposition (Burkle and Irwin, 2009; Burkle and Irwin, 2010; Hoover et al., 2012), which can reduce the diversity and cover of flowering plants that provide pollinator foods (e.g., Burkle and Irwin, 2010; Stevens et al., 2011). The individual impact of nitrogen deposition on pollinators, networks and pollination may be relatively weak (Burkle and Irwin, 2009; Burkle and Irwin, 2010). Nonetheless, nitrogen in combination with climate warming and elevated CO^2 produced subtle effects on bumble bee nectar consumption and reduced bee longevity (Hoover et al., 2012). Nitrogen deposition was shown to have another, indirect effect – nitrogen deposition near freeways in California favoured growth of grasses eliminating butterfly hostplants of an endangered species. If grazing is used to reduce the grass, this effect of N deposition can be reversed (Weiss, 1999). Further work is required to elucidate the potential of nitrogen deposition as part of a suite of pressures affecting pollinators.

2.3.4.3 Light pollution

Light pollution, a driver clearly affecting nocturnal species and growing in importance due to urbanization has to be mentioned. Its effect is still scarcely studied, though artificial night light is known to alter the perception of photoperiod (Hölker et al., 2010, Lyytimäki, 2013) and even at low levels can affect the organism (Gaston et al., 2013). Artificial night light was shown to influence moth physiology and behaviour, e.g., inhibit the release of sex pheromones by females (Sower et al., 1970), suppress their oviposition (Nemec, 1969), negatively affect the development of nocturnal larvae of Lepidopteran species (van Geffen et al., 2014), or act as ecological traps for some vulnerable species, drawing them to suboptimal habitats like urban areas (Bates et al., 2014). Moths are known pollinators of some plants, especially plants whose flowers open at night (MacGregor et al., 2015), however their role as pollinators is still not evaluated in depth (MacGregor et al., 2015). Studies suggest that the effects of artificial night lighting may cause not only declines in moth populations – due to their negative influence on reproduction and development – but might, as a result, also cause potential changes in the composition of moth assemblages and possibly in the ecosystem functions they provide (MacGregor et al., 2015). Further studies are needed to evaluate the extent of light pollution effects on nocturnal pollinators.

2.3.5 Conclusions

It is clear that pollinators may be exposed to a wide range of pesticides in both agricultural and urban environments. The risk posed by pesticides is driven by a combination of the toxicity (hazard) and the level of exposure; the latter being highly variable and affected by factors including crop type, the timing, chemical type, rate and method of pesticide applications, as well as the ecological traits of managed and wild pollinators. Insecticides are toxic to insect pollinators and their exposure, and thus the risk posed, is increased if, for example, labels do not provide use information to minimise pollinator exposure or the label is not complied with by the pesticide applicator. In addition, there is good evidence from laboratory and in-hive dosing studies that insecticides have the potential (depending on exposure level) to cause a wide range of sublethal effects on individual pollinator behaviour and physiology, and on colony function in social bees, that could affect the pollination they provide. However, significant gaps in our knowledge remain as most sublethal testing has been limited in the range of pesticides, exposure levels and species, making extrapolation to managed and wild pollinator populations challenging. For example, there is considerable uncertainty about how the level, time course and combination of sublethal effects recorded on individual insects in the laboratory might affect the populations of wild pollinators over the long term. The interaction between pesticides and other key pressures on pollinators in realistic combinations and scales of stressors (land-use intensification and fragmentation, climate change, alien species, pests and pathogens) is little understood.

The GMOs (Genetically Modified Organisms) most used in agriculture carry traits of IR (Insect Resistance), HT (Herbicide Tolerance) or both. Though pollinators are considered non-target organisms of GMOs, they can be subject to direct and indirect effects. Direct effects of insect pollinators' exposure to IR-crops show that Bt-toxins are non-lethal to Hymenoptera and Coleoptera, and can be lethal to

Lepidoptera pollinatoros. Sub-lethal effects on the behavior and learning in honey bees have been reported in one study. IR-crops result in a global reduction of insecticide use, which in turn impact positively the diversity of insects. Because of the use of herbicides, HT-fields harbor reduced number of the weeds attractive to pollinators, what can lead to a reduction of pollinators in GM-fields. Introgression of transgenes in wild relatives (e.g. canola, cotton and maize) and non-GM crops has been shown, but there is a lack of evidence on the effect of these events on pollinators, pointing to the need for more studies on this topic.

Pollutants pose a potential threat to pollinators. There are numerous papers using honey bees and their hive products as good indicators of environmental pollution levels, indicating that honey bees can be directly exposed to pollutants. Yet, detailed studies are still lacking concerning the effects of various forms of pollution on bee biology. Invertebrate models suggest that susceptibility of various species of insects to industrial pollutants, like heavy metals, can vary greatly due to various strategies used to cope with such contamination. Some pollutants can bioaccumulate, especially through plants and their products, like nectar or pollen, and affect the level of exposure depending on the pollinator species' ecology. Large, between-species differences in susceptibility and various plant-pollinator dependences make it difficult to foresee the effect of a given pollutant to the environment without direct field studies.

2.4 Pollinator diseases and pollinator management

2.4.1 Pollinator diseases

Bee diseases by definition have some negative impacts at the individual bee, colony or population level. As such, they can be pointed to as potential drivers of pollinator decline (Potts et al., 2010; Cameron et al., 2011a; Cornman et al., 2012). Parasites and pathogens can be widespread in nature but may only become problematic when bees are domesticated and crowded (Morse and Nowogordzki, 1990; Ahn et al., 2012). Additionally, stressors such as pesticides or poor nutrition can interact to cause disease levels to increase (Vanbergen and the Insect Pollinators Initiative, 2013). Disease spread can be a consequence of bee management (detailed in section 2.4.2) and has been most studied in honey bees, somewhat in bumble bees and much less in other bees. Bee diseases can spillover or move from one bee species to another (e.g., Deformed Wing Virus DWV between honey bees and bumble bees) and even within a genus the movement of managed bees to new areas can spread disease to indigenous species (e.g. *Apis* and *Varroa*, Morse et al., 1990; and *Bombus* and *Nosema*, Colla et al., 2006). In addition to parasites and pathogens in bees, bats, birds and other pollinators can suffer from disease and thus impact pollination (Buchmann and Nabhan, 1997). Diseases can directly impact pollinator health but can also interact with other factors, such as poor nutrition, pesticides, etc., which cause stress and thus together contribute to pollinator declines (vanEngelsdorp et al., 2010; Vanbergen and the Insect Pollinators Initiative, 2013). Table 2.4.1 gives an overview of bee parasites and pathogens.

8. Table 2.4.1: Bee parasites and pathogens.

	Host	**Remarks**	**References**
Viruses			
Acute bee paralysis virus (ABPV)	*Apis mellifera; A. ceranae*, *Bombus* spp.	*Varroa* mites can “activate” release virus in *Apis*. In *Bombus*, experimental infection.	1
Black queen cell virus (BQCV)	*Apis mellifera, A. ceranae, A. florea, A. dorsata, Bombus* spp., *Megachile rotundata, Nomia melanderi* (only in adults)	Mainly affects developing queen larvae and pupae in the capped-cell stage. Associated with *Nosema apis*. Found in different *Bombus* species.	2, 43
Chronic bee paralysis virus (CBPV)	*Apis mellifera, A. ceranae*	Causes the same symptoms of trembling and the iability to fly in infected bees that ABPV.	3
Deformed wing virus (DWV)	*Apis mellifera, A. ceranae, A. florea, A. dorsata, Bombus* spp., *Nomia melanderi*	Causes well-defined disease symptoms (crumpled wings, shrunken, decreased body size, and discoloration), activated by *Varroa*.	4, 5, 42
Israeli Acute Paralysis Virus (IAPV)	*Apis mellifera, A. ceranae, Megachile rotundata, Nomia melanderi*	A widespread RNA virus of honey bees that has been linked with colony losses, activated by *Varroa*. It disrupts the diapause of Megachile rotundata, though does not affect larval survival and development.	6, 7, 43
Kashmir bee virus (KBV)	*Apis mellifera, A. ceranae*, *Bombus* spp.	‘Covert’ infections. Multiplies quickly and kills host within 3 days when injected.	8
Sacbrood virus（SBV）Thailand sacbrood virus (TSBV) Chinese sacbrood virus (CSBV)	*Apis mellifera, A. ceranae, Nomia melanderi* (SBV only in adults)	Highly infective in *Apis cerana*. Causes the delince of *A. ceranae*.	9, 10, 11, 42, 43
Lake Sinai Virus (LSV)	*Apis mellifera*	Common and very abundant at peak incidence.	12
Tobacco Ringspot Virus (TRSV)	*Apis mellifera*	Host-jumping virus from plant to honeybee.	13
Black-head virus	*Nomia melanderi*	Dead pupae has a black head. Little is know about this virus and its effects.	42
Mahagony virus	*Nomia melanderi*	Dead pupae are uniform mahagony color. Little is know about this virus and its effect.	42
Protozoa			
Crithidia mellificae, Leptomonas apis	*Apis mellifera*	Common. No harmful effects known.	14, 15
Crithidia bombi	*Bombus* spp., subgenus *Psithyrus*	Highly infective, In *Psithyrus*: known from males only.	16, 17
Crithidia expoeki	*Bombus* spp.		17
Amoeba (*Malpighamoeba mellificae*)	*Apis mellifera*	Associated with Bee Virus and *Nosema apis*, few effects.	18
Apicystis (=Mattesia) bombi	*Bombus* spp. *Psithyrus*	Also found in queens.	19
Bacteria			

	Host	**Remarks**	**References**
Melissococcus (Streptococcus) plutonius	*Apis mellifera*	Causes European Foulbrood. More benign than American foulbrood.	20
Paenibacillus (Bacillus) larvae	*Apis mellifera, Osmia bicornis*	Causes American Foulbrood. Kills larvae after cocoon is spun. Pathogenicity is unconfirmed in mason bees that may only serve as an intermediate host, vector or habitat for these bacteria, which are virulent to honey bees.	20, 44
Aerobacter cloaca	*Apis mellifera, Bombus* spp.	In ovaries of queens. Causes B-meleanosis.	21
Bacillus alvei, B. laterosporus	*Apis mellifera*	Some are secondary invaders with *P. larvae* after years of endemic foulbrood.	21
Bacillus pulvifaciens	*Apis mellifera*	Causes “powdery scale” of larvae. Perhaps a saprophyte that occasionally infects larvae.	21
Bacillus thuringiensis	*Osmia bicornis*	Pathogenicity is unconfirmed and mason bees may only serve as an intermediate host, vector or habitat for these bacteria, which are virulent to honey bees.	44
Bacterium eurydice	*Apis mellifera*	Secondary invader with *M. Pluton*.	21
Hafnia alvei	*Apis mellifera*	Associated with infection by *Varroa* mites. Causes septicemia and death when in hemolymph.	21
Nonidentified bacterium (gram-positive)	*Bombus melanopygus*	Dead larvae characteristically hard.	21
Pseudomonas aeruginosa, P. apiseptica	*Apis mellifera*	In hemolymph of moribund bees near hives, also in soil.	21
Spiroplasma apis, S. melliferum	*Apis mellifera, Bombus* spp.*, Osmia bicornis, Osmia cornifrons*	In *Bombus* in hemolyph. Found on flowers, also in solitary bees. There is no information whether these are real pathogens in mason bees.	21, 44, 45
Fungi			
Nosema apis	*Apis mellifera, A. cerana*	Association with BQCV virus, and with Malpighamoeba. Queens are replaced in the colony or become sterile. Colony growth reduced, lower honey yield.	21, 22, 23
Nosema ceranae	*Apis mellifera, A. cerana, A. dorsata, A. koschevnikovi, Bombus* spp.	The most widespread adult bee disease. This invading pathogen is now common and seems to rapidly replace *N. apis* as the dominant microsporidian infection in many geographic locations.	24, 25, 26, 27, 28, 29, 30 31, 54, 55, 56, 57
Nosema bombi	*Bombus* spp.	Can cross-infect among *Bombus* species. Workers die quickly. Colonies develop poorly.	31
Nosema thomsoni	*Bombus* spp.	Found in different *Bombus* species.	31
Ascosphaera alvei, A. apis, A. flavus, A. fumigatus	*Apis mellifera*	*A. apis* causes chalkbrood disease. *A. flavus, A. fumigatus* causes stonebrood of larvae.	21
Ascosphaera aggregata	*Megachile rotundata*	In the alfalfa leafcutter bee usually, infection	46, 47,

	Host	Remarks	References
		levels are not exceeding 5%, however, in extreme cases infection levels above 50% were also recorded and can cause serious losses. These fungi are rather species specific, but some cross-infectivity is possible.	49, 50
Ascosphaera torchioi	*Osmia lignaria*	So far it seems, that these fungi are rather species specific, but some cross-infectivity is possible	48, 49
Acrostalagmus sp.	*Bombus* spp.	Diseased queens with short hibernation.	21
Aspergillus candidus, A. niger	*Bombus* spp., *Apis mellifera, Megachile rotundata, Nomia melanderi*	*A.niger* probably opportunistic infetions. In Oregon *Aspergillus* spp. have destroyed up to 53% of *Nomia melanderi* cells.	21, 42, 51
Beauveria bassiana, B. tenella	*Bombus* spp., *Apis mellifera*	From worker pupae in *Apis*	21
Candida pulcherrima, Candida sp. Various yeasts	*Apis mellifera, Bombus* spp., *Megachile rotundata, Nomia melanderi*	Appears as a consequence of stress. Diseased queens with short hibernation. *Saccharomyces* sp. infenction causes larval bloating in *Nomia melanderi*.	21, 42, 51
Cephalosporium	*Apis mellifera, Bombus* spp.	Causes typical discolorations. Serious effects in *Bombus*.	21
Chactophoma sp., *Cladosporium cladosporioides*	*Apis mellifera*	Causes typical discolorations. Also in combs.	21
Hirsutella sp., *Metarhizium anisopliae*	*Bombus* spp.	Mycel extends beyond host body.	21
Paecilomyces farinosus	Apidae, *Bombus* spp.	Pathogenic in *Bombus*.	21
Penicillium funiculosum, P. cyclopium	*Apis mellifera*	From all stages, workers, drones.	21
Phoma sp., *Rhodotorula glutinis*	*Apis mellifera*	Causes typical discolorations. In drone larvae.	21
Torulopsis sp.	*Apis mellifera*	Pathogenic yeast. In sick bees.	21
Verticillium lecanii	*Bombus* spp.		21
Fusarium sp., *Mucor* sp.	*Megachile rotundata*		51
Parasitic mites			
Varroa destructor	*Apis mellifera*	The most serious threat to honey bee populations worldwide, and as a serious and deadly vector for transmitting viruses.	32
Honey bee tracheal mite (*Acarapis woodi*)	*Apis mellifera*	Cause bee to have disjointed wings and be unable to fly.	33
Bumblebee tracheal mite (*Locustacarus buchneri*)	*Bombus* spp.	Puncture trachea and suck hemolymph.	34
Tropilaelaps clareae and *Troplilaelaps* spp.	*Apis cerana, A. dorsata, A. laboriosa, A. florea, A. mellifera*	The most serious threat to honey bee in Asia.	35
Pests			
Wax Moths	*Apis mellifera*	The most serious pest of honeycombs.	36
Small Hive Beetle	*Apis mellifera*	Can cause colonies to abscond and can	36, 52,

	Host	**Remarks**	**References**
		damage brood and honey when larva reproduce. Reported to also infest *Bombus* and stingless bee nest.	53
Bee-louse	*Apis mellifera*	No detrimental effect on adult bees, larvae can damage the appearance of comb honey.	36
Bee eaters (*Merops* sp.)	*Apis mellifera*	Problematic locally when queens are being reared.	37
Chlacid wasps (*Monodontomerus* sp., *Melittobia* sp.)	*Osmia* sp., *Megachile* sp.	Parasiting solitary bee nests, destroying/eating the developing larvae. *Melittobia acasta* caused significant losses in *Osmia coerulescens* populations.	38, 39, 58, 59
Checkered beetles (*Trichodes apiaries, Trichodes ornatus*)	*Osmia* sp., *Megachile* sp.	Commonly found in nests. Can cause losses up to 89%, but on average around 30% in managed colonies.	38, 39, 41
Flies (*Cacoxenus indagator*, *Anthrax anthrax*)	*Osmia* sp., *Megachile* sp.	*Anthrax* flies and most probably also other flies are of less concern due to low infestation rates of nests.	38, 58
Mites (*Chaetodactylus osmiae*)	*Osmia* sp., *Megachile* sp.		38, 58, 59

	Host	Remarks	References
Viruses			
Acute bee paralysis virus (ABPV)	*Apis mellifera; A. ceranae, Bombus* spp.	*Varroa* mites can “activate” release virus in *Apis*. In *Bombus*, experimental infection.	1
Black queen cell virus (BQCV)	*Apis mellifera, A. ceranae, A. florea, A. dorsata, Bombus* spp., *Megachile rotundata, Nomia melanderi* (only in adults)	Mainly affects developing queen larvae and pupae in the capped-cell stage. Associated with *Nosema apis*. Found in different *Bombus* species.	2, 43
Chronic bee paralysis virus (CBPV)	*Apis mellifera, A. ceranae*	Causes the same symptoms of trembling and the iability to fly in infected bees that ABPV.	3
Deformed wing virus (DWV)	*Apis mellifera, A. ceranae, A. florea, A. dorsata, Bombus* spp., *Nomia melanderi*	Causes well-defined disease symptoms (crumpled wings, shrunken, decreased body size, and discoloration), activated by *Varroa*.	4, 5, 42
Israeli Acute Paralysis Virus (IAPV)	*Apis mellifera, A. ceranae, Megachile rotundata, Nomia melanderi*	A widespread RNA virus of honey bees that has been linked with colony losses, activated by *Varroa*. It disrupts the diapause of Megachile rotundata, though does not affect larval survival and development.	6, 7, 43
Kashmir bee virus (KBV)	*Apis mellifera, A. ceranae, Bombus* spp.	‘Covert’ infections. Multiplies quickly and kills host within 3 days when injected.	8
Sacbrood virus（SBV） Thailand sacbrood virus (TSBV) Chinese sacbrood virus (CSBV)	*Apis mellifera, A. ceranae, Nomia melanderi* (SBV only in adults)	Highly infective in *Apis cerana*. Causes the delince of *A. ceranae*.	9, 10, 11, 42, 43
Lake Sinai Virus (LSV)	*Apis mellifera*	Common and very abundant at peak incidence.	12
Tobacco Ringspot Virus (TRSV)	*Apis mellifera*	Host-jumping virus from plant to honeybee.	13
Black-head virus	*Nomia melanderi*	Dead pupae has a black head. Little is know about this virus and its effects.	42
Mahagony virus	*Nomia melanderi*	Dead pupae are uniform mahagony color. Little is know about this virus and its effect.	42
Protozoa			
Crithidia mellificae, Leptomonas apis	*Apis mellifera*	Common. No harmful effects known.	14, 15
Crithidia bombi	*Bombus* spp., subgenus *Psithyrus*	Highly infective, In *Psithyrus*: known from males only.	16, 17

	Host	Remarks	References
Crithidia expoeki	*Bombus* spp.		17
Amoeba (*Malpighamoeba mellificae*)	*Apis mellifera*	Associated with Bee Virus and *Nosema apis*, few effects.	18
Apicystis (=Mattesia) bombi	*Bombus* spp. *Psithyrus*	Also found in queens.	19
Bacteria			
Melissococcus (Streptococcus) plutonius	*Apis mellifera*	Causes European Foulbrood. More benign than American foulbrood.	20
Paenibacillus (Bacillus) larvae	*Apis mellifera, Osmia bicornis*	Causes American Foulbrood. Kills larvae after cocoon is spun. Pathogenicity is unconfirmed in mason bees that may only serve as an intermediate host, vector or habitat for these bacteria, which are virulent to honey bees.	20, 44
Aerobacter cloaca	*Apis mellifera, Bombus* spp.	In ovaries of queens. Causes B-meleanosis.	21
Bacillus alvei, B. laterosporus	*Apis mellifera*	Some are secondary invaders with *P. larvae* after years of endemic foulbrood.	21
Bacillus pulvifaciens	*Apis mellifera*	Causes "powdery scale" of larvae. Perhaps a saprophyte that occasionally infects larvae.	21
Bacillus thuringiensis	*Osmia bicornis*	Pathogenicity is unconfirmed and mason bees may only serve as an intermediate host, vector or habitat for these bacteria, which are virulent to honey bees.	44
Bacterium eurydice	*Apis mellifera*	Secondary invader with *M. Pluton*.	21
Hafnia alvei	*Apis mellifera*	Associated with infection by *Varroa* mites. Causes septicemia and death when in hemolymph.	21
Nonidentified bacterium (gram-positive)	*Bombus melanopygus*	Dead larvae characteristically hard.	21
Pseudomonas aeruginosa, P. apiseptica	*Apis mellifera*	In hemolymph of moribund bees near hives, also in soil.	21
Spiroplasma apis, S. melliferum	*Apis mellifera, Bombus* spp., *Osmia bicornis, Osmia cornifrons*	In *Bombus* in hemolyph. Found on flowers, also in solitary bees. There is no information whether these are real pathogens in mason bees.	21, 44, 45
Fungi			
Nosema apis	*Apis mellifera, A. cerana*	Association with BQCV virus, and with Malpighamoeba. Queens are replaced in the colony or become sterile. Colony growth reduced, lower honey yield.	21, 22, 23
Nosema ceranae	*Apis mellifera, A. cerana, A. dorsata, A. koschevnikovi, Bombus* spp.	The most widespread adult bee disease. This invading pathogen is now common and seems to rapidly replace *N. apis* as the dominant	24, 25, 26, 27, 28, 29, 30 31,

	Host	Remarks	References
		microsporidian infection in many geographic locations.	54, 55, 56, 57
Nosema bombi	*Bombus* spp.	Can cross-infect among *Bombus* species. Workers die quickly. Colonies develop poorly.	31
Nosema thomsoni	*Bombus* spp.	Found in different *Bombus* species.	31
Ascosphaera alvei, A. apis, A. flavus, A. fumigatus	*Apis mellifera*	*A. apis* causes chalkbrood disease. *A. flavus*, *A. fumigatus* causes stonebrood of larvae.	21
Ascosphaera aggregata	*Megachile rotundata*	In the alfalfa leafcutter bee usually, infection levels are not exceeding 5%, however, in extreme cases infection levels above 50% were also recorded and can cause serious losses. These fungi are rather species specific, but some cross-infectivity is possible.	46, 47, 49, 50
Ascosphaera torchioi	*Osmia lignaria*	So far it seems, that these fungi are rather species specific, but some cross-infectivity is possible	48, 49
Acrostalagmus sp.	*Bombus* spp.	Diseased queens with short hibernation.	21
Aspergillus candidus, A. niger	*Bombus* spp., *Apis mellifera, Megachile rotundata, Nomia melanderi*	*A.niger* probably opportunistic infetions. In Oregon *Aspergillus* spp. have destroyed up to 53% of *Nomia melanderi* cells.	21, 42, 51
Beauveria bassiana, B. tenella	*Bombus* spp., *Apis mellifera*	From worker pupae in *Apis*	21
Candida pulcherrima, Candida sp. Various yeasts	*Apis mellifera, Bombus* spp., *Megachile rotundata, Nomia melanderi*	Appears as a consequence of stress. Diseased queens with short hibernation. *Saccharomyces* sp. infenction causes larval bloating in *Nomia melanderi*.	21, 42, 51
Cephalosporium	*Apis mellifera, Bombus* spp.	Causes typical discolorations. Serious effects in *Bombus.*	21
Chactophoma sp., *Cladosporium cladosporioides*	*Apis mellifera*	Causes typical discolorations. Also in combs.	21
Hirsutella sp., *Metarhizium anisopliae*	*Bombus* spp.	Mycel extends beyond host body.	21
Paecilomyces farinosus	Apidae, *Bombus* spp.	Pathogenic in *Bombus*.	21
Penicillium funiculosum, P. cyclopium	*Apis mellifera*	From all stages, workers, drones.	21
Phoma sp., *Rhodotorula glutinis*	*Apis mellifera*	Causes typical discolorations. In drone larvae.	21
Torulopsis sp.	*Apis mellifera*	Pathogenic yeast. In sick bees.	21
Verticillium lecanii	*Bombus* spp.		21
Fusarium sp., *Mucor* sp.	*Megachile rotundata*		51

	Host	Remarks	References
Parasitic mites			
Varroa destructor	*Apis mellifera*	The most serious threat to honey bee populations worldwide, and as a serious and deadly vector for transmitting viruses.	32
Honey bee tracheal mite (*Acarapis woodi*)	*Apis mellifera*	Cause bee to have disjointed wings and be unable to fly.	33
Bumblebee tracheal mite (*Locustacarus buchneri*)	*Bombus* spp.	Puncture trachea and suck hemolymph.	34
Tropilaelaps clareae and *Troplilaelaps* spp.	*Apis cerana*, *A. dorsata*, *A. laboriosa*, *A. florea*, *A. mellifera*	The most serious threat to honey bee in Asia.	35
Pests			
Wax Moths	*Apis mellifera*	The most serious pest of honeycombs.	36
Small Hive Beetle	*Apis mellifera*	Can cause colonies to abscond and can damage brood and honey when larva reproduce. Reported to also infest *Bombus* and stingless bee nest.	36, 52, 53
Bee-louse	*Apis mellifera*	No detrimental effect on adult bees, larvae can damage the appearance of comb honey.	36
Bee eaters (*Merops* sp.)	*Apis mellifera*	Problematic locally when queens are being reared.	37
Chlacid wasps (*Monodontomerus* sp., *Melittobia* sp.)	*Osmia* sp., *Megachile* sp.	Parasiting solitary bee nests, destroying/eating the developing larvae. *Melittobia acasta* caused significant losses in *Osmia coerulescens* populations.	38, 39, 58, 59
Checkered beetles (*Trichodes apiaries, Trichodes ornatus*)	*Osmia* sp., *Megachile* sp.	Commonly found in nests. Can cause losses up to 89%, but on average around 30% in managed colonies.	38, 39, 41
Flies (*Cacoxenus indagator*, *Anthrax anthrax*)	*Osmia* sp., *Megachile* sp.	*Anthrax* flies and most probably also other flies are of less concern due to low infestation rates of nests.	38, 58
Mites (*Chaetodactylus osmiae*)	*Osmia* sp., *Megachile* sp.		38, 58, 59

References

[1] Benjeddou et al., 2001; [2] Benjeddou et al., 2002; [3] Allen and Ball, 1996; [4] de Miranda and Genersch, 2010; [5] Li et al., 2011; [6] Di Prisco et al., 2011; [7] Cox-Foster et al., 2007; [8] Stoltz et al., 1995; [9] Ma et al., 2010; [10] Abrol and Bhat, 1990; [11] Bailey and Fernando, 1972; [12] Ravoet et al., 2015; [13] Li et al., 2014; [14] Langridge and Mcghee, 1967; [15] Schwarz et al., 2015; [16] Schmid-Hempel and Reber, 2004; [17] Schmid-Hempel and Tognazzo, 2010; [18] Butler, 1945; [19] Plischuk et al., 2011; [20] Bailey and Ball, 1991; [21] Schmid-Hempel, 1998; [22] Li et al., 2012b; [23] Wang and Moller, 1970; [24] Fries et al., 2006; [25] Higes et al., 2006; [26] Huang et al., 2007; [27] Paxton et al., 2007; [28] Chen et al., 2008; [29] Chen et al., 2009; [30] Fries, 2010; [31] Li et al., 2012; [32] Tentcheva et al., 2004; [33] Ball, 1997; [34] Macfarlane et al., 1995; [35] Wongsiri et al., 1989; [36] Shimanuki and Knox, 2000; [37] Ball and Bailey, 1997; [38] Krunić et al., 2005; [39] Eves et al., 1980; [40] Purves et al., 1998; [41] Fairey et al., 1984; [42] Johansen, 1976; [43] Singh, 2011; [44] Keller et al., 2013; [45] Whitcomb, 2012; [46] Stephen, 1959; [47] Stephen, 1978; [48] Abrol, 2012; [49] Youssef et al., 1985; [50] James, 2008; [51] Inglis et al., 1993; [52] Hoffmann et al., 2008; [53] Greco et al., 2010; [54] Fries et al., 1996; [55] Cox-Foster et al., 2007; [56] Klee et al., 2007; [57] Plischuk et al., 2009; [58] Bosch, 1992; [59] Bosch and Kemp, 2001b.

2.4.1.1 Honey bee parasites and pathogens

A honey bee colony may harbor a wide variety of disease-causing agents, bacteria, fungi, viruses, parasitic mites and even other insects that try to take advantage of the rich resources contained within bee colonies (Morse et al., 1990; Evans and Spivak, 2010). Experiments to determine cause and effect often use a single pathogen but multiple pathogens, including viruses, may be contributing to colony decline (Johnson, 2009; vanEngelsdorp et al., 2010, Cornman et al., 2012). Interactions have been documented between Black Queen Cell Virus (BQCV) and *Nosema* (Doublet et al., 2015) and pesticide exposure and *Nosema* (Alaux et al., 2010b; Vidau et al., 2011, Pettis et al., 2012) but these same effects have not been seen at the colony level (Retsching et al., 2015). What is widely accepted is that bee diseases vary in time and space (Highfield et al., 2009; Martin et al., 2012) and are often associated with bee colonies that are not in ideal nutritional state or under some other form of stress (Staveley et al., 2014), such as transportation (Ahn et al., 2012), pesticide exposure (Pettis et al., 2013) or crowding (Morse et al., 1990).

2.4.1.1.1 Viruses of honey bees

Viral diseases are numerous in bees, with over 18 identified; the major ones studied being Acute bee paralysis virus (ABPV); BQCV; Chronic bee paralysis virus (CBPV); DWV; Israel Acute Paralysis Virus (IAPV); Kashmir bee virus (KBV); and Sacbrood virus (SBV). Viral infections in honey bee colonies have often been reported to be involved in the collapse of bee colonies infested with *Varroa destructor* (de Miranda et al., 2011). The combination of *Varroa* and many viruses are known to impact colony survival (Neuman and Carreck 2010; Nazzi et al., 2012). Specifically, the association of *Varroa* mite infestation with Deformed wing virus (DWV) has been reported to be responsible for colony losses (Martin et al., 2012; Dainat et al., 2012; Ryabov et al., 2014).

2.4.1.1.2 Bacteria of honey bees

Bacterial diseases including American Foulbrood (AFB) and European Foulbrood are caused by *Paenibacillus larvae* ssp. *larvae* and by *Melissococcus plutonius*, respectively. Both foulbrood diseases are "notifiable" (must be reported to appropriate authorities) in most parts of the world (OIE, 1996) as they are contagious and can cause damage to equipment by contamination and death to colonies that become heavily infected (Morse et al., 1990).

2.4.1.1.3 Fungi of honey bees

Fungal agents include *Nosema* (or nosemosis), which is probably the most widespread adult honey bee pathogen and includes two species, *Nosema apis* and *Nosema ceranae*, both of which are microsporidia that infect the gut of adult bees, but infection may or may not affect hive productivity (Fries, 2010). *Nosema ceranae* is a parasite that was first described to infect *A. cerana* (Fries et al., 1996) and has become widespread in *A. mellifera* throughout the world (Fries et al., 2006; Higes et al., 2006; Cox-Foster et al., 2007; Klee et al., 2007). *Nosema ceranae* also has wide host range, for example in *Apis* species (i.e. *A. florea, A. dorsata,* and *A. korchevnikovi*) and bumble bees (Plischuk et al., 2009; Li et al., 2012). The wide host range of this parasite is of significant epidemiological concern. Other fungal diseases of bees include 'Chalkbrood' (*Ascosphaera apis*) and 'Stonebrood', caused by *Aspergillus fumigatus, Aspergillus flavus* and *Aspergillus niger,* both of which can result in larval death.

2.4.1.1.4 Parasitic mites of honey bees

The major parasitic mites include two external mites, *Varroa spp*. and *Tropilaelaps spp*., with *Varroa* being widespread while the *Tropilaelaps* mites only attack honey bees in Asian countries, including but not limited to South Korea, China and Thailand (Oldroyd and Wongsiri, 2009). A small internal parasitic mite with worldwide distribution is the tracheal mite *Acarapis woodi* that infests the airways of adult honey bees.

2.4.1.1.5 Pests of honey bees

Honey bee pests are numerous and include many invertebrates and some vertebrates (Morse et al., 1990). Birds can be problematic; "bee eaters" (*Merops* sp.) are pests in managed apiaries in the Old World (Fry, 2001; Kastberger and Sharma, 2000). Several hornets are major pests around the world (Oldroyd and Wongsiri, 2009), and *Vespa velutina* has recently spread to Europe from SE Asia (Villemant et al., 2011). Another pest that has recently expanded its host range is the small hive beetle, *Aethina tumida*, moving from Africa to the US, Australia, Portugal and Italy in the past 20 years (Hood, 2004; Neumann and Elzen, 2004; Mutinelli, 2014). The small hive beetle has the potential to damage bees beyond the genus *Apis* and may threaten bumble bees (Hoffmann et al., 2008) as well as stingless bees (Greco et al., 2010). Of the known pest, the parasitic mites are most problematic, as they switch host and spread worldwide (Morse et al., 1990, Oldroyd and Wongsiri, 2009).

2.4.1.2 Bumble bee parasites and pathogens

The relative importance of the several factors involved in the decline of bumble bee populations is controversial, in particular because considerably less effort has been given by scientific research to these bees than to honey bees. The spread of pathogens during management of bumble bee colonies for pollination (see section 2.5) is highly suspected to be one of the main factors in their decline in North and South America (Williams and Osborne, 2009; Cameron et al., 2011a; Arbetman et al., 2013; Manley et al., 2015). In the frame of the red-listing of bumble bee species worldwide, a collective expertise conducted by the IUCN and a panel of experts (Cameron et al., 2011b) identified four patterns by which pathogens are a major cause of decline in bumble bees (Manley et al., 2015). As reviewed in section 2.5 and Box 2.4.1, the use of infected commercially-reared bumble bees for crop pollination has been shown to result in local spread of pathogens, or “spillover”, to wild bumble bees (Colla et al., 2006; Goka et al., 2006; Schmid-Hempel et al., 2014). In theory, such spillover can result in disease epidemics in wild populations, leading to local bumble bee declines (Otterstatter and Thomson, 2008). Studies show that commercial bumble bee colonies commonly harbor parasites and pathogens harmful to wild bees, such as microsporidia and viruses (Singh et al., 2010). This results in pathogen spillover from greenhouse raised to wild bumble bees. For example, in Canada Colla et al., (2006) showed a significantly higher prevalence of *Crithidia bombi*, a bumble bee pathogen, in the vicinity of greenhouses. Otterstatter and Thomson (2008) theoretically and experimentally demonstrated that during the first three months of spillover, transmission from commercial hives infected up to 20% of wild bumble bees within two km of greenhouses. Consistent with these data, (Murray et al., 2013) found the greatest pathogen prevalence in a radius of two km from greenhouses, decreasing at distances higher than ten km.

2.4.1.2.1 Viruses of bumble bees

Viruses are cofactors in the decline of pollinators, and in some cases, of bumble bees. DWV, one of the most common viruses in honey bees, was demonstrated to cause wing deformities in bumble bees (Li et al., 2011; Fürst et al., 2014). ABPV, BQCV (Peng et al., 2011), and KBV were found to be equally capable of infecting different species of bumble bees (Anderson, 1991). Fast-evolving RNA viruses, known to cause severe colony losses in managed honey bee populations, deserve particular attention for their propensity to jump between host species, in particular when transmitted by pollen foraged from flowers (Singh et al., 2010). Viruses thus threaten ecologically and economically important wild pollinator communities (Manley et al., 2015). Impacts of these pathogens on bumble bees are currently unknown, but potentially could lead to severe consequences in terms of colony survival and population dynamics, as has been observed in honey bees. Immediate research efforts are needed to understand the disease dynamics and potential health impacts of multi-host parasites on bumble bees and to develop risk mitigation strategies for rational use of pollen in bee rearing, considering the possible role of pollen in the transmission of viruses.

Finally, potential exists for inter-generic pathogen transmission among Hymenoptera, as suggested by spatial analysis (Fürst et al., 2014). In general, the transportation of honey bee colonies, honey bee products, and other managed pollinators could potentially lead to emergence of new diseases in bumble bees as well as introduction of more virulent strains of naturally occurring diseases via intergeneric transmission of pathogens and parasites. Reports are increasing of bumble bees infected with RNA viruses (DWV, ABPV, BQCV, KBV, SBV, and IAPV) that were originally isolated from honey bees (Meeus et al., 2011; Singh et al., 2010; McMahon et al., 2015).

2.4.1.2.2 Protozoa of bumble bees

The trypanosome *Crithidia bombi* (Kinetoplastida: Trypanosomatidae) has been the focus of considerable study. It infects the gut of bumble bees and has been found throughout Europe, Canada and China. Recently a second species of this genus, *Crithidia expoeki*, has been discovered to occur globally (Schmid-Hempel and Tognazzo, 2010). Infection occurs via ingestion of parasite cells, and infected hosts later release hundreds of thousands of parasite cells in their faeces. *C. bombi* infects the fat bodies of bumble bees, and does not seem to occur more commonly in commercial than wild bees (Otterstatter et al., 2005). Infection may have different effects, from the reduction in the colony founding success, colony growth and reproduction (Brown et al., 2003), to the increase in mortality rates in food-stressed bees (Brown et al., 2000). A different protozoan, *Apicystis bombi*, can also be highly virulent, and is suspected to be a main factor of decline in South American bumble bees (Arbetman et al., 2013).

2.4.1.2.3 Fungi of bumble bees

Parasites in bumble bees are numerous and widespread (Schmid-Hempel, 2001) and their effect can be quite devastating (Rutrecht and Brown, 2009; Otti and Schmid-Hempel, 2007). *Nosema bombi* (Microsporidia: Nosematidae) has been suspected to be the driving factor for declines of certain North American bumble bee species as well as in China (Li et al., 2011).

2.4.1.2.4 Parasitic mites of bumble bees

The tracheal mite (*Locustacarus buchneri*) occurs in wild bumble bees (Otterstatter, 2004) and is associated with lethargy in infected workers (Husband and Sinha, 1970), but evidence that it can reduce colony survival and reproduction is lacking.

2.4.1.2.5 Pests of bumble bees

The entomopathogenic nematode, *Sphaerularia bombi*, is a well known pest of bumble bees that only attacks queens, a strategy that restricts it to a very small proportion of the host population, but can have a strong impact, considering that the queen is the single egg-laying female of the colony.

2.4.1.3 Stingless bee parasites, pathogens and predators

Scant information is available on diseases that affect meliponiculture (stingless bee management) across different regions of the world. In nature, stingless bee colonies live inside tree trunks, branches, roots, buildings or ground cavities (Nogueira-Neto, 1997; Roubik, 2006), which are often invaded by parasites, pathogens, pests and predators. Nests of stingless bees are attractive habitat and food source for various pathogens and predators, which can destroy these colonies (Wattanachaiyingcharoen and Jongjitvimo, 2007; Roubik, 1989). The presence of natural enemies may impose cost and reduce the number of forager bees.

2.4.1.3.1 Viruses of stingless bees

Stingless bee pathogens are less known and investigated. The first virus detected in stingless bees was the acute bee paralysis virus (ABPV) in *Melipona scutellaris* in Brazil in 2015, however. It is unknown whether it is pathogenic to these bees (Ueira-Vieira et al., 2015). Native stingless bee colonies of various species kept together with managed honey bees infected with DWV, IAPV, SBV, and KBV were found to be free of these viruses (Freiberg, 2012).

2.4.1.3.2 Protozoa of stingless bees

So far, no information is available on Prokaryotes accompanying stingless bees.

2.4.1.3.3 Bacteria of stingless bees

Two bacterial diseases, the para-foul brood (*Bacillus para alvei*) and the American foul brood (*Bacillus larvae*) have been diagnosed in *M. quadrifasciata* so far. Most developed colonies of stingless bees are well protected inside the nest (Chinh et al., 2005; Roubik 2006), as the sticky resin they store as part of their defence mechanism (Klumpp, 2007; Dollin, 2010) is known to have antibacterial properties (Lokvam and Braddock, 1999).

2.4.1.3.4 Fungi of stingless bees

The fungus *Geotrichum* was found in *M. puncticollis* colonies in South America (Nogueira-Neto, 1997).

2.4.1.3.5 Pest and predators of stingless bees

Wild and domesticated stingless bees have wide range of pests and predators including flies (Phoridae), ants, anteaters, birds, lizards, spiders, reduviid bugs, assassin bugs, termites and pillage bees (Klumpp, 2007; Wattanachaiyingcharoen and Jongjitvimo, 2007), hive beetles (Lea, 1910, 1912; Halcrof et al., 2011), wasps *Braconid spp*, phorid flies (Klumpp, 2007), reptiles, birds, amphibians, frogs and toads, sun bears, rodents, squirrels, and wasps (*Vespa spp.*) (Jalil and Shuib, 2014). Adults and larvae of many species are parasitoids or specialist predators of the bees (Feener and Brown, 1997; Morrison, 1999). Phorid flies (Diptera, Phoridae) are the most devastating pests of stingless bee colonies (Disney and Bartareau, 1995; Nogueira-Neto, 1997; Van Veen *et al*., 1990). The flies are attracted by the odors emitted by stored pollen, enter colonies and lay hundreds of eggs, which after becoming larvae deplete the colony's food stores, causing a considerable damage and often the total collapse of the colony (Maia-Silva et al., 2012). However healthy stingless bees have capability to defend themselves and their nests against pests and diseases and acquire a variety of defensive strategies by protective building behaviour and defensive reactions (Greco et al., 2010; Halcroft et al., 2011; Kerr and Lello, 1962; Lehmberg et al., 2008; Pasteels et al., 1983; Roubik, 2006).

2.4.1.4 Solitary bee parasites and pathogens

The most important managed solitary bee species belong to three families: Megachilidae (mainly *Megachile* and *Osmia* species), Halictidae (*Nomia melanderi* Cockerell and *Rhophitoides canus* (Eversmann)) and Apidae (mainly *Anthophora* and *Peponapis* species). Their growing importance as managed agricultural crop pollinators facilitated studies of their natural pathogens and parasites. The best-studied species are the alfalfa leafcutter bee (*Megachile rotundata* Fab.), the alkali bee (*N. melanderi*), the blue orchard bee (*Osmia lignaria* Say) the red mason bee (*O. bicornis* L.), the hornfaced bee (*O. cornifrons* Radoszkowski) and the horned bee (*O. cornuta* Latr.).

2.4.1.4.1 Viruses of solitary bees

Alkali bees are known hosts to viruses appearing also in honey bees, like the deformed wing virus (DWV), sacbrood virus (SBV), and also the black-head and mahagony viruses. (Johansen, 1976). Similarly, managed leafcutter bees are also known hosts of honey bee viruses, like the black queen cell (BQCV) or DWV (Vega and Kaya, 2012). A recent study (Singh et al., 2010) also described a number of RNA viruses with a broad host range among various Hymenopterans. Their findings suggest that at least RNA viruses can freely circulate in the pollinator community and can have important implications on export/import or movement of managed pollinators, including solitary bees. Nevertheless, information on viral diseases in solitary bees is still scarce and they require further studies.

2.4.1.4.2 Protozoa of solitary bees

Solitary bees and their nests are accompanied by a wide variety of Prokaryotes (Inglis et al., 1993). Most of these microorganisms are usually either beneficial or harmless, living in the midgut of bees, found in faeces, or in provisions (Inglis et al., 1993; Goerzen, 1991). Some of them may be part of the resident microflora, others simple commensals found in the midgut and reported to have significant importance in food uptake and host survival (Keller et al., 2013).

2.4.1.4.3 Bacteria of solitary bees

Only a few bacteria are raising concerns such as *Bacillus thuringiensis, Paenibacillus larvae* or *Spiroplasma melliferum. Bacillus* and *Paenibacillus* were found to be well represented in *Osmia* nests (Keller et al., 2013). However, their pathogenicity is speculative and mason bees may only serve as an intermediate host, vector or habitat for these bacteria, which are virulent to honey bees (Keller et al., 2013). Similarly, *S. melliferum*, a *Spiroplasmataceae* found in *O. cornifrons* (Whitcomb, 2012), is known to be lethal for honey bees (Clark et al., 1985), however, there is no information whether it is a real pathogen in mason bees. Nevertheless, co-appearance of these bacteria in both honey bees and some solitary bees suggest that pathogen spill-over from managed populations into wild ones cannot be excluded and further studies are needed to clarify microbiota interaction in solitary bees.

2.4.1.4.4 Fungi of solitary bees

Chalkbrood, caused by various species of the genus *Ascosphaera*, is one of the most widely studied fungal disease found in solitary bee species as well as in honey bees (Evison, 2012; James, 2008; Stephen, 1978; Wynns et al., 2013). The most heavily infected species with chalkbrood is the alfalfa leafcutter bee, in which the disease is commonly found in North America. Usually, infection levels do not exceed 5%, however, in extreme cases infection levels above 50% have been recorded (Stephen, 1959) in spite of various control/disinfection methods, causing serious losses (James 2008). The species infecting leafcutter bees, *A. aggregate*, was identified in 1973 (Stephen et al., 1981). *A. torchioi* was identified in *O. lignaria* by Youssef and McManus in 2001. So far it seems that these fungi are rather species specific (Stephen et al., 1981), but some cross-infectivity is possible (Youssef et al., 1985). Besides chalkbrood disease solitary bees were found to also harbour large numbers of other fungi, like *Aspergillus*, *Candida*, *Fusarium, Mucor* or even *Saccharomyces* species; however, the role of most of these species is uncertain (Inglis, 1993).

2.4.1.4.5 Pests of solitary bees

Solitary bees also host a large variety of parasites, starting from numerous phoretic mites and ending on parasitic wasps feeding on bee larvae. Due to their economic importance mostly parasites of the intensively managed species are described in literature together with possible methods of protection against them. Most of these parasites are not strictly species specific, and are found in various solitary bee species (Krunić et al., 2005). The most widespread are various chalcid wasps, like *Monodontomerus* and *Melittobia* sp., beetles (*Trichodes* sp.), flies (*Cacoxenus indagator*, *Anthrax anthrax*), mites (*Chaetodactylus* sp.), etc. (Bosch and Kemp, 2001; Krunić et al., 2005). Chalcid wasps are widespread parasitizing *Megachile* (Eves et al., 1980) and *Osmia* (Bosch and Kemp, 2001; Krunić et al., 2005). Using artificial nesting material or insecticide strips (Hill et al., 1984) the level of these parasitic wasps was found to be controllable (Krunić et al., 2005). *Melittobia* sp. wasps have high reproductive potential, short life cycle, and are often found in managed bee nests (Bosch and Kemp, 2001; Krunić et al., 2005) causing significant losses in *O. coerulescens* populations (Purves et al., 1998). Other species like *Sapyga pumila* or *S. quinquepunctata* also attack the nests of solitary bees, however in their case some effective control methods are already available (Torchio, 1979). Cleptoparasitic *Chaetodactylus* mites were also found to cause losses in managed *Osmia* sp. populations (Bosch, 1992; Bosch and Kemp 2002; Yamada 1990) and thermal shock treatment is used to control these pests (Yamada, 1990). The checkered beetle (*Trichodes apiarius*) is commonly found in Europe and North Africa parasitizing both *Megachile* and *Osmia* species (Krunić et al., 2005), while *T. ornatus* is common in North America (Fairey et al., 1984; Bosch and Kemp, 2001). According to Eves at al. (1980) this beetle can cause losses up to 89%, but on average around 30% in managed colonies. Methods of control are usually mechanical, like sorting the cocoons (Fairey et al., 1984) or eliminating the beetles using aromatic attractant bait traps (Wu and Smart, 2014). *Anthrax* flies and most probably also other flies are of less concern due to low infestation rates (3% of *Anthrax* sp. in Washington, USA in alfalfa leafcutter bee colony) (Eves et al., 1980).

2.4.2 Pollinator management

2.4.2.1 Honey bee management

The management of honey bees has facilitated the movement of different bee species to areas of the world where they are not native. This movement, while beneficial in some cases for honey production and pollination, has also had negative impacts through disease spread and replacement of local pollinators (Goulson, 2003). However, when using native bees, beekeeping can be viewed as a conservation tool and enhance local fauna and food production (Jaffé et al., 2010). The number of colonies managed in any given area can be linked to supply and demand for pollination and or the price of honey (vanEngelsdorp and Meixner, 2010). Thus, the actual number of colonies managed and the need for those colonies are driven by external factors beyond the control of the beekeeper (Morse and Calderone, 2000; vanEngelsdorp and Meixner, 2010). Lastly, the demand for pollination is growing faster than the supply of managed pollinators in developing areas of the world (Aizen and Harder, 2009).

The name honey bee refers to all bees in the genus *Apis* with two major species managed around the world; the western honey bee *Apis mellifera* and the eastern honey bees *Apis cerana* and *Apis indica*. Both cavity-nesting bees can be managed in human-made containers and moved to follow honey flows or for pollination (Crane, 1983). Modern beekeeping started with the invention of the movable frame hive in 1853 (Langstroth, 1853), allowing beekeepers to harvest honey without destructively cutting out combs, inspect for disease, and to remove frames to start new colonies (see Chapters 1, 3 and 5 for more on historical bee management). One example of disease spread and reduction in pollination availability comes from the use of non-movable comb hives in South Korea where a viral disease, Thai sacbrood, wiped out 90% of *A. cerana* hives resulting in the need for hand pollination of pears and other fruit trees (Yoo *et al.*, 2012).

Growing demands for pollination and searching for better honey production areas have driven beekeepers to become migratory in many areas of the world. This migratory trend has increased recently but bees have been moved since humans began to manage them (e.g., on the Nile in ancient Egypt, Crane, 1983). Because honey production depends on the availability of flowers in the immediate area, beekeepers quickly learned that by moving hives to areas of better forage (nectar flows as they are called) they could produce more honey. The need to move hives for honey production, and more recently pollination, has made migratory beekeeping standard practice in many parts of the world (Pettis *et al.*, 2014). Bee colonies are most often moved at night over short distances but if longer distances are required then bees may be closed with screens or nets and placed on large trucks for transport. During a move some bees are lost or left behind, and this can spread diseases and pests to new areas. The most extreme migratory beekeeping for pollination occurs in the U.S. each year, when 1.5 million or more colonies are moved from across the U.S. to California to pollinate almond trees in February and March (Sumner and Boris, 2006). Migratory beekeeping is advantageous to the beekeeper in moving to paid pollination contracts or to maximize honey production. However, migratory beekeeping does have impacts on local honey bee and native bee populations as it facilitates the rapid spread of bee diseases and pests and can cause pathogen spillover to native bee populations (Goulson, 2003; Moritz et al., 2005; Furst et al., 2014; Smith et al., 2014). Lastly, the worldwide trade in bee products (wax, honey, pollen and propolis) is another avenue for the spread of diseases and pests to new areas (Ritter, 2014). Diseases and pests can survive on traded bee products and be a source of spread to new areas if used in beekeeping or rearing of bumble bees (e.g., pollen – Graystock et al., 2013, or e.g., royal jelly can harbor diseases that then spread globally if used in queen rearing upon importation into disease-free areas).

Movement of bee species to new areas or continents can cause unanticipated additional risks beyond pests and disease spread and may include; changes in local bee fauna, competition for resources and changes in beekeeping practices with newly introduced species (Roubik and Wolda, 2001; Goulson, 2003; Moritz *et al.*, 2005; Howlett and Donovan, 2010). One example is the importation of Caucasian bee stocks into the Cevennes National Park in France to replace the local native bees *Apis mellifera mellifera*; this introduced *Varroa* mites that then largely wiped out the native bees in that area (Elie, 2015). Other examples of introgression of new genetic stock into local populations are known (De la Rua et al., 2009). Two additional examples are the movement of *A. mellifera* to the Americas and into SE Asia; both of these moves have some positive aspects in pollination and honey production. However, in SE Asia this has led to host shifts of a parasitic mite (*Varroa destructor*, Anderson and Trumann, 2000) and a gut parasite of adult bees (*Nosema ceranae*; Fries, 1996), both of which adversely affect honey bees worldwide and can spill over to other bee species (see disease section). Additionally, the use of *A. mellifera,* while good for honey production, has caused a decline in the keeping of other bees native to these areas, for example, stingless bees (Quezada-Euan et al., 2001, Cortopassi-Laurino et al., 2006, Dohzono et al., 2008, Jalil, 2014) and *A. cerana* in SE Asia (see Chapt. 5; Oldroyd and Wongsiri, 2006). Competition for resources with the introduction of exotic bee species has been studied but the results are mixed (Roubik et al., 1986, Steffan-Dewenter and Tscharntke, 2000, Roubik and Wolda, 2001, Hansen et al., 2002, Paini, 2004). When Africanized bees moved into South and Central America, the native bees were able to shift to other host plants and thus behaviorally compensate in the diverse plant habitat of the Americas (Roubik, 2009). Competition between Apis and Bombus has been documented (Thomson, 2004, 2006). There is no question that if resources are limited then competition between introduced species like honey bees and native bees, birds or other nectar feeders can occur (Roubik and Wolda, 2001, Hudewenz and Klein, 2013; Elbgami et al., 2014). To date there is only limited evidence that competition is sufficient to lead to major declines of local bees or other pollinators.

Despite the negative aspects of disease spread, modern agriculture in many parts of the world relies on a mobile pollinator that can be moved to a crop during bloom. This is most important in large-scale agricultural production systems such as almonds, apples, melons and other cucurbits where large fields provide limited edges where wild pollinators may nest (Kremen, 2005). In many areas of the world with less intensive and large-scale agriculture beekeepers primarily move for honey production and the pollination they provide is free. Solutions to the issue of large field sizes can include more plant diversity in the agricultural landscape and the use of smaller fields or orchards (Winfree et al., 2007). It has been shown that wild bees provide a great deal of pollination (Garibaldi et al., 2013) and thus managed bees may be considered supplemental in some but not all cases. Efforts to maximize the proper distribution of managed pollinators can increase efficiency and reduce costs (www.beeswax.me.uk http://almopol.com). This research suggests that if agriculture can adopt smaller plot sizes and more diverse flora, then managed pollinators can serve as pollination "insurance" and be used to augment the wild pollinators in a given area (Winfree et al., 2007; Breeze et al., 2014). Currently, with limited agricultural land available and a need to maximize production, managed pollinators like honey bees will remain in demand for crop pollination.

2.4.2.2 Bumble bee management

In the past few decades, bumble bees (the genus *Bombus*) have been increasingly subject to commercial trade for use as pollinators (see Chapter 1). Five species of bumble bees are currently used for crop pollination, the major ones being *Bombus terrestris* from Europe and *Bombus impatiens* from North America (Velthuis and van Doorn, 2006). The massive introduction of colonies, within or outside the natural range of these species, was identified as one of the main threats to native bumble bees and other bee species (Cameron et al., 2011b), due to several types of risks (Table 2.4.2). These introductions create two main kinds of risks: the competition for resources (including nesting sites and the transmission of diseases and pathogens. A third, less–explored, kind of risk is the reproductive interference due to interspecific mating between introduced and native bumble bee species (Kanbe et al., 2008).

The initial risk occurs when non-native commercial bumble bees escape to the wild, potentially becoming invasive, competing with native bumble bees. Non-native bumble bees include exotic species, but also subspecies or even different ecotypes or genotypes. Two well-described cases are the importation and subsequent naturalization of *B. terrestris* to Northern Japan (Hokkaido) in the 1990s (Inoue et al., 2007) and the introduction and establishment of several *Bombus* species in New Zealand and Australia (Macfarlane and Griffin, 1990). A recent case has been the rapid extension of *B. ruderatus* and *B.*

terrestris in South America (see text in Box 2.4.1). There is then a risk of competition for nesting sites and for floral resources between introduced species and native non-bumble bee species, but few studies have addressed this aspect.

The greatest risk related to bumble bee management is probably the spread of diseases at local, national, and international levels (Goka et al., 2006) (see also 2.4.1.2). A recent study (Graystock et al., 2013) referred to managed colonies as "Trojan hives", after showing that 77% of commercially produced bumble bee colonies from three main producers imported to the UK on the basis of being parasite-free were shown to carry eight different parasites. This publication actually contributed to establish new restrictions for bumble bee use in the UK. Spread of such parasites is unavoidable considering the permeability of cropping systems to commercial bumble bees. This was demonstrated in Ireland when bumble bees kept in greenhouses from which they were supposedly unable to exit were shown to collect 31% to 97% of their pollen from outside the greenhouses (Murray et al., 2013). This presents a risk to native bumble bees in the regions to which they are introduced, so that the prevalence of bumble bee pathogens shows considerable variation among sites (Gillespie, 2010) and among species (Koch et al., 2012). Available data show that commercially produced bumble bee colonies can pose a significant risk to native pollinators (e.g. Szabo et al., 2012), not only due to introduction of parasites in populations that may have a low prevalence of pathogens, but also because the movement of commercial colonies may disrupt spatial patterns in local adaptation between hosts and parasites (Meeus et al., 2011). This risk could even be higher when bumble bees are used for open field pollination; this is a noted limitation in all of the mentioned studies that used greenhouses as a focal point for the spillover hypothesis. Another factor that increases the risk is that commercial bumble bees have been noted to have a higher prevalence of several diseases than their wild counterparts.

Box 2.4.1: Case study: the invasion of European bumble bees introduced for crop pollination in southern South America

The southern tip of South America (Argentina and Chile) is inhabited by a single native bumble bee species, *Bombus dahlbomii*, whose key role in plant-pollinator webs and in the pollination of native plant species has been recognized. This region has been invaded by the European bumble bee *B. ruderatus* in 1993 (Roig-Alsina and Aizen, 1996) and *B. terrestris* in 2006 (Torretta et al., 2006), following their introduction for crop pollination into Chile in 1982 and 1997, respectively.

Three independent studies have shown that both introduced bumble bee species have spread widely in the region, invading new habitats (Montalva et al., 2011; Morales et al., 2013; Schmid-Hempel et al., 2014). More specifically, a recent large-scale survey of bumble bee fauna across the eastern slopes of the southern Andes in Argentina revealed that *B. terrestris* was by far the most widespread and abundant species, one order of magnitude more abundant than *B. dahlbomii* and *B. ruderatus.* Meanwhile, *B. dahlbomii* had disappeared from a large part of its historical range (Morales et al., 2013).

B. dahlbomii closely interacts with the native endemic plant "*amancay*" (*Alstroemeria aurea*), related to a variety of commercial hybrid lilies. A 20-year survey of pollinators of *amancay* in an old growth forest whose understory is dominated by this flowering plant revealed that first *B. ruderatus*, and later *B. terrestris*, replaced *B. dahlbomii*, formerly the most abundant pollinator (Morales et al., 2013).

What are the mechanisms underlying displacement of native bumble bees by invasive ones? In the case of *B. ruderatus*, mechanisms behind its initial, partial displacement of *B. dahlbomii* on the local level remain unknown, and the hypothesis of competition for resources has received little support (Aizen et al., 2011). In the case of *B. terrestris,* its wide range and long-lasting displacement of *B. dahlbomii* has been hypothesized to be the result of an interplay between competition for resources and pathogen spillover. *B. terrestris* is a highly generalist species, foraging on many types of flowers – even those classified as anemophilous or ornithophilous (see Chapter 1). Furthermore, its colonies are larger and they begin their activity earlier in the spring than do colonies of *B. dahlbomii*; this likely provides it with a competitive advantage.

Recent studies provide evidence that populations of *B. terrestris* in southern South America carry *Apicystis bombi*, a highly pathogenic parasite new to this region (Plischuk and Lange, 2009) that seems to have been introduced along with it and transmitted *in situ* to *B. dahlbomii* and *B. ruderatus* (Arbetman et al., 2013). This pathogen also infects honey bees (*Apis mellifera*). Moreover, the fact that infected honey bees have been detected in a region of southern Argentina invaded with *B. terrestris* but not in regions free of this invasive bumble bee (Plischuk et al., 2011), and that infected *B. terrestris, B. ruderatus* and *A. mellifera* from this region share the same *Apicystis* haplotypes (Maharramov et al., 2013), supports the theory of a common origin of this pathogen in all three species, and suggests a probable spillover from *B. terrestris* to these species, though this remains to be confirmed.

The impacts of these invasions on plant pollinator interactions and plant pollination range from disruption of local plant-pollinator webs (Aizen et al., 2011) to reduced weight and quality of raspberries along a gradient of increasing *B. terrestris* invasion (Sáez et al., 2014) due to their overabundance.

This case study illustrates how the issues of bumble bee management for crop pollination, invasive pollinators (see section 2.5.4), and bumble bee diseases (see section 2.4.1.2) are closely linked and therefore should be addressed in an integrated manner. In addition, this evidence provides sound arguments for discouraging introduction of non-native pollinator species.

Movement of managed bumble bees may also pose risks to other bee species, because diseases are spread by transfer of pathogens between bumble bees and other bees through shared flowers. Following importation, commercially produced bumble bees interact with native bumble bees and other pollinators during shared flower use (Whittington and Winston, 2004), creating a risk for the community of pollinators as a whole (Durrer and Schmid-Hempel, 1994).

Finally, other significant risks are the possibility of hybridization of native and non-native bumble bees, which thus far has been shown to occur only at the intraspecific level, or the risk of reproductive failure consecutive to interspecific mating. In Poland, (Kraus et al., 2010) have demonstrated 33% to 47% introgression of the commercial subspecies *B. terrestris dalmatinus* and *B. t. sassaricus* to the local *B. terrestris*, indicating a potential risk of loss of genetic diversity, even when moving colonies of the same species. This suggests that for commercial species, the colonies should be moved only to areas where local bees are genetically close.

9. Table 2.4.2: Bumble bee management and its effects on crop and wild plant pollination and other native wild pollinators.
For a list of crops pollinated, see Klein et al. (2007).

Species (managed first, year, when known)	Negative effects on wild pollinators
Bombus terrestris dalmatinus **(Europe 1997, Asia 1992, South America 1998)**	Displacement of native bumblebee due to a potential combination of competition for resources and pathogen spillover (Arbetman et al. 2013, Morales et al. 2013, Schmid-Hempel et al. 2014,) **STRONG EVIDENCE** Genetic pollution of local population by managed individuals of distant populations or subspecies (Kraus et al. 2010) **MEDIUM EVIDENCE** Hybridization of native and non-native bumblebees (Tsuchida et al. 2010) **MEDIUM EVIDENCE**

	Introduction of non-native species causing disturbance in native bee diversity and competing with native species (Inoue et al. 2007) **MEDIUM EVIDENCE**
B. t. audax **(introduced from UK to New Zealand in approx. 1900)**	May compete with native species for nectar and pollen from a range of plant species (Howlett & Donovan 2010) **WEAK EVIDENCE**
B. impatiens **(North America 1990)**	Greenhouse escapees infect local populations with parasites/pathogens, raising the natural local level of pathogens (Colla & Packer 2008) **STRONG EVIDENCE**
B. ignitus **(Japan 1999, China 2000)**	This will result in introduction of exotic pathogens/parasites (Goka et al. 2006) **STRONG EVIDENCE**
B. t. terrestris **(Norway)** ***B. t. canariensis*** **(Canary Islands 1994)** ***B. t. saccaricus*** **(Sardinia)** ***B. occidentalis*** **(North Amercia 1990)**	No studies

2.4.2.3 Stingless bee management

Stingless bees (Meliponini) are a traditional honey, propolis and wax source in South and Central America (Cortopassi-Laurino et al., 2006, Nates-Para, 2001; 2004), Australia (Heard and Dollin, 2000), Africa (Kwapong et al., 2010), and Asia (Cortopassi-Laurino et al., 2006), but recently their role as possible managed pollinators of agricultural crops is also raising interest (Slaa et al., 2006, Giannini et al., 2014). Stingless bees are an important asset to fulfill the growing agricultural demand for pollination, because they could compensate for the local declines in honey bee populations (Brown and Paxton, 2009, Jaffé et al., 2010, van Engelsdorp and Meixner, 2010) by assuring enough pollinators (Aizen and Harder, 2009) and by pollinating crops more effectively (Garibaldi et al., 2013). Across developing countries, stingless beekeeping (also known as meliponiculture), remains essentially informal, technical knowledge is scarce, and management practices lack standardization. Commercialized bee products, including honey, colonies, and in a few cases crop pollination, are generally unregulated, and demand often exceeds supply. Meliponiculture thus remains a largely under-exploited business (Jaffé et al., 2015).

In most African countries stingless bees are hunted for their honey instead of being managed, which can lead to the destruction of wild colonies however, meliponiculture does exist in Tanzania and Angola (Cortopassi-Laurino et al., 2006, Jaffé et al., 2015). While in e.g. Ghana (Kwapong et al., 2010) and Kenya (Macharia et al., 2007) an interest to develop stingless bee management has been identified. In Australia management practices were developed to provide pollination with stingless bees for agricultural crops (Heard and Dollin, 2000). Stingless bees were found to be as often managed for pollination purposes as for honey production, already at the end of the last century according to the survey conducted by Heard and Dollin (2000). They found that the most common species kept in Australia are *Trigona carbonaria* (69%) and *T. hockingsi* (20%). Stingless bees in Australia are used and promoted mostly for macadamia nut, orchards (Heard and Dollin, 2000), mango and watermelon pollination (Dollin, 2014). In Central and South America stingless bees are usually used for honey, propolis and wax production used for medicinal and ritual purposes, however, their role in crop pollination is being more often investigated (Cortopassi-Laurino et al., 2006). Meliponiculture in these countries can take various forms and use different traditional and modern techniques or types of hives depending on the target bee species (Cortopassi-Laurino et al., 2006). Stingless bee honey producers can be well organized, e.g. in Brazil a private virtual initiative was created to connect stingless beekeepers to exchange experiences, buy and sell products and acquire know-how. In Mexico, some species are actively managed in rural areas (Sommeijer, 1999; Quezada-Euán et al., 2001; González-Acereto et al., 2006), while a number of species are still traditionally hunted for their honey (Reyes-González et al., 2014). There is also active promotion of such beekeeping in Mexico and studies show, that the stingless bee species *Nannotrigona perilampoides* is a cost-effective pollinator for some locally-grown crops (González-Acereto et al., 2006).

Management of stingless bees for crop pollination purposes, as mentioned earlier, is less popular, but efforts are underway to promote them as crop pollinators in Brazil (Imperatriz-Fonseca et al. 2006). *Melipona fasciculata* was identified as a potential eggplant pollinator (Nunes-Silva et al., 2013), and *N. punctata* and *M. scutellaris* have been identified as potential pollinators of guava, greenhouse strawberries (Castro, 2002), and apples (Vianna et al., 2014). In Mexico, the stingless bee N. perilampoides was tested for tomato pollination (Cauich et al., 2004). Similar trends are observed in southern Asia (in India) and in South-East Asia (Indonesia, Malaysia, Thailand and the Philippines), where besides traditional stingless bee honey production (Kahono, 2011; Kumar et al., 2012), management for pollination is beginning to take root (Cortopassi-Laurino et al., 2006).

Important efforts have been directed to train beekeepers and standardize management practices (Nogueira-Neto, 1997; Villas-Bôas, 2012), quantify investment costs and profit perspectives (Lobato and Venturieri, 2010), assess honey properties, quality and commercialization routes (Vit et al., 2013), rear queens artificially (Menezes et al., 2013), and diagnose the overall situation of the sector in different regions (Halcroft et al., 2013; González-Acereto et al., 2006). More recently, quantitative efforts have been directed to the optimization of stingless beekeeping. Relying on Brazil-wide surveys, Jaffé et al. (2015) assessed the impact of particular management practices on productivity and economic revenues from the commercialization of stingless bee products. Another recent contribution analyzed the long-term impact of management and climate on honey production and colony survival in a commercial stingless bee from North-eastern Brazil (Koffler et al., 2015).

Stingless beekeeping should be regarded as a prime tool to achieve sustainable development. Keeping bees can help low-income communities earn additional revenues from selling bee products, thus reducing the need to exploit other natural resources and creating incentives to protect natural habitats as food sources and nesting sites for the bees. Moreover, beekeeping contributes to the provision of pollination, assuring crop yields and helping maintain plant biodiversity in natural ecosystems. Stingless beekeeping could thus help protect the bees, safeguard their pollination, and contribute to the development of many rural communities. However, more efforts are needed to optimize this activity. Achieving such optimization is difficult, given the huge diversity of management practices (tightly linked to cultural heritage), as well as the striking biological differences among species (Vit et al., 2013; Roubik, 2006). Recent interest in the production of more stingless bee honey, as described above, has already generated some new practices, like the developing trade of colonies of these bees, e.g., in Australia, or attempts to introduce species out of their natural range, like in Japan (Amano, 2004). This poses new potential risks – as seen mainly in honey bees and bumble bees (see diseases section for details), like the introduction of pathogens and the loss of genetic diversity. Therefore, optimization of stingless bee managment should be done with care and within the borders of their native range.

2.4.2.4 Solitary bee management

Solitary bees have been used for agricultural crop pollination for almost a century. The longest-managed and described species are undoubtedly the alfalfa leafcutter bee (*Megachile rotunda*) (Pitts-Singer and Cane, 2011; Ruz, 2002), introduced to North and South America and Australia, the alkali bee (*Nomia melanderi*) (Cane, 2008), the blue orchard bee (*Osmia lignaria*) (Bosch and Kemp 2001), both used in North America, the hornfaced bee (*O. cornifrons*) in Japan (Maeta, 1990), the horned bee (*Osmia cornuta*) and the red mason bee (*Osmia bicornis*) in Europe. All these species require relatively simple handling including the use of standardized nesting boxes for their nesting aggregates and simple cocoon collection and cleaning procedures for further breeding (Bosch and Kemp 2002; Sedivy and Dorn, 2013). In return, they significantly increase crop yield and often provide better crop quality compared to crops pollinated mostly by honey bees (for details see Table 2.4.2.). Due to their effectiveness as crop pollinators and their simple handling, solitary bees are often introduced to new locations as managed pollinators. They are mostly used in open field pollination, but they also do well in greenhouse conditions (Bosch and Kemp, 2000, Wilkaniec and Radajewska, 1997). Recently, solitary bees have also been supported by introducing of artificial nesting sites, so-called "bee hotels", to promote not only wild bee conservation but also pollination of both crops and wild plants on a small scale (Gaston et al., 2005). However, the effectiveness of these artificial nesting sites was questioned by MacIvor and Packer (2015), who showed that bee hotels might promote introduced species more than native ones, and also may act as population sinks for bees through facilitating the increase of parasites and predators.

10. Table 2.4.3: Managed solitary bees and the opportunities they offer and – respectively - risks they pose to their environment.

Species	**Originating (or.) and managed in (since); if known**	**Crops pollinated**	**Effects on**	
			Crop pollination	Wild pollinators
Anthophora pilipes **shaggy fuzzyfoot bee**	Japan (or.), USA (introduced in 1988), Germany (1990), Japan (1990)	Blueberry orchards	**POSITIVE** Superior pollinator of blueberries in Japan..[1, 2] European subspecies of this bee has been managed to increase the pollination of fruit trees and orchards.[3] **HIGH CONFIDENCE**	**NONE** described but being used without noticeable side effects for decades in its original location. **MEDIUM CONFIDENCE**
Megachile pugnata **sunflower leafcutter bee**	North America (or.) (1990s)	Sunflower	**POSITIVE** Increased sunflower pollination. Active earlier during the days, than honeybees or bumblebees. [4] **HIGH CONFIDENCE**	**NONE** described, but being used without noticeable side effects for decades in its original location. **MEDIUM CONFIDENCE**
M. rotundata **alfalfa leafcutter bee**	Europe (or.), USA (1930), Western Canada (1962), New Zealand (introduced in 1971), Australia (introduced in 1987)	Alfalfa, lowbush blueberry, carrots, vegetables, canola, melon, sweet clover, cranberry	**POSITIVE** in USA tripled alfalfa seed production. In New Zealand, bees have been observed foraging on 10 different introduced plant species from the families Asteraceae, Brassicaceae, Crassulaceae and Fabaceae. In Canada [1] leafcutter bees saved the alfalfa industry. [5, 6, 7] **HIGH CONFIDENCE**	**NONE** described but being used without noticeable side effects for decades in its original location. **MEDIUM CONFIDENCE** **NONE** described in New Zealand and Australia. Although competition for nesting sites may occur with the native *Hylaeus* spp. low abundance, restricted distribution and preferences for introduced plants suggest that these managed bees are unlikely to pose a competitive threat to native pollinators.[6] **LOW CONFIDENCE**

Species	Originating (or.) and managed in (since); if known	Crops pollinated	Effects on	
			Crop pollination	Wild pollinators
Nomia melanderi **alkali bee**	USA (or.) (1940) New Zealand (introduced in 1964)	Red clover, alfalfa	**POSITIVE** Greater seed production in lucerne. Both males and females are superior to honeybees in pollinating alfalfa. [5] **HIGH CONFIDENCE**	**NONE**. Alkali bees have specific nesting requirements restricting their spread, no competition for nesting sites with native bees was noted in New Zealand. [6] **LOW CONFIDENCE**
Osmia cornifrons **hornfaced bee**	Japan (or.) (1960), USA (introduced beginning of 1980s), Korea, China (1990s)	Orchards, especially apple, mustard	**POSITIVE** The hornfaced bee is 80 times more effective than honeybees for pollinating apples. In Japan, where hornfaced bees pollinate up to 70 percent of the country's apple crop. [8] **HIGH CONFIDENCE**	**NONE** described, but being used without noticeable side effects for decades in its original location and where introduced **MEDIUM CONFIDENCE**
O. cornuta **horned bee**	Europe (or.), Spain, France and Yugoslavia	Orchards, oilseed rape, blackberry	**POSITIVE** Generally increases crop pollination [9] and especially apple. *Osmia* pollinated orchards produce enhanced yields in favourable years. Also safeguard a yield in years that would otherwise be devoid of any yield. [10] **HIGH CONFIDENCE**	**NONE** described, but being used without noticeable side effects for decades in its original location and in the US. **MEDIUM CONFIDENCE**
O. lignaria **blue orchard bee**	North America (or.) (1970)	Orchards	**POSITIVE** Orchard pollination. They are particularly efficient pollinators of fruit trees, promote cross-pollination and increase yield in cultivars that require cross-pollination. [11, 12] **HIGH CONFIDENCE**	**NONE** described, but being used without noticeable side effects for decades in its original location. **MEDIUM CONFIDENCE**
O. bicornis **red mason bee**	Europe (or.), Germany (2010), Poland (2012)	Oilseed rape, blackcurrant, strawberries, orchards	**POSITIVE** They are efficient pollinators of blackcurrant [13] and strawberries also in tunnels. [14, 15] **HIGH CONFIDENCE**	**NONE** described, but being used without noticeable side effects for decades in its original location. **MEDIUM CONFIDENCE**

References

[1] Batra, 1994; [2] Stubbs and Drummond, 1999; [3] Thalann and Dorn, 1990; [4] Parker and Frohlich, 1985; [5] Cane, 2002; [6] Howlett and Donovan, 2010; [7] Cane, 2008; [8] Maeta, 1990; [9] Krunic and Stanisavljevic, 2006; [10] Bosch and Kemp, 2002; [11] Bosch et al., 2006; [12] Torchio, 1985; [13] Fliszkiewicz et al., 2011; [14] Wilkaniec and Radajewska, 1997; [15] Schindler and Peters, 2011.

Managed solitary bees, in contrast to honey bees and bumble bees, are less studied concerning the risk they pose to their environment (for details see Table 2.4.2.). Managed solitary bees, which are transported or just simply introduced into new localities (Bartomeus et al., 2013), can impact native pollinator species and the pollination they provide. However, the only well-documented case of invasiveness of an introduced pollinator is the giant resin bee (*M. sculpturalis*), a legume pollinator from Central Asia. Giant resin bees were accidentally introduced to the USA, where they started to outcompete the native carpenter bee (*Xylocopa virginica*) at its nesting sites (Laport and Minckley, 2012). Disease spread by managed solitary bees requires further studies, especially studies on procedures for controlling pathogens and internal parasites, and the impact of management on native bees. Lack of appropriate disease control, together with large aggregation sizes, may facilitate disease spread and therefore impact native pollinators and their pollination. (For further details see section 2.4.1.4.)

2.4.3 Conclusions

Bee management is a global and complex driver of pollinator loss. Spreading of diseases by managed honey bees and bumble bees into wild bee species has been shown to present a threat to some wild species and populations. Preservation of some of the economically important (for their pollination in crop production) bee species that otherwise could decline is also important from a conservation point of view. In some cases, like honey bees or bumble bees, both pros and cons of their large-scale management for pollination are well known. These managed bees provide convenient pollination, because they can be moved in large numbers to large-scale pollinator-dependent monoculture plantings that have high pollination requirements at specific time points. However, these managed bees can also transmit diseases to local populations of wild pollinators, further diminishing naturally-occurring pollination, which already tend to be low in large, monoculture croplands that supply few natural nesting habitats or floral resources across time for wild bees (see section 2.2.2). The logical conclusion is to create pollinator-friendly habitats to promote pollinator abundance and diversity instead of migratory bee management, when possible. However, if pollinator-friendly habitats cannot be created, it is advisable to manage native or in some cases naturalized populations rather than non-native bee species, because the greatest risk by bee management occurs when species are moved out of their native range. In case of solitary and stingless bee management the picture is less clear because empirical studies on the impact they have on their environment are still scarce. Yet, to foresee and avoid possible pitfalls of managing solitary and stingless bees it is important to keep in mind the negative impacts observed from honey and bumble bee management thus far.

2.5 Invasive alien species

2.5.1 Introduction

Especially since the 1950s, the growth in global economic wealth, trade and commerce and transport efficiency has facilitated the ongoing worldwide human-mediated dispersal of organisms into novel environments (Hulme, 2009; Mack et al., 2000). This represents a key component of global environmental change as once introduced beyond their natural range, and given evolutionary and ecological constraints or opportunities, these alien plant and animal species can become invasive, altering the biological and physical nature and processes of the recipient ecosystem (Jones and Gomulkiewicz, 2012; Mack et al., 2000). 'Alien species' are defined as a (non-native, non-indigenous, foreign, exotic) species, subspecies, or lower taxon occurring outside of its natural range (past or present) and dispersal potential (i.e. outside the range it occupies naturally or could occupy without direct or indirect introduction or care by humans) and includes any part, gametes or propagule of such species that might survive and subsequently reproduce (IUCN, 2000). 'Alien invasive species' are alien species that become established in natural or semi-natural ecosystems, and are an agent of change, threatening native biological diversity (IUCN, 2000). In this section we assess the evidence for impacts by alien invasive species on native pollinators, plant-pollinator interactions and pollinator community networks. We assess impacts from different invasive alien groups accidentally or deliberately introduced beyond their natural range, namely: flowering plants (2.5.2); herbivores that consume pollinator food plants (2.5.3); predators (2.5.4); and competitors (other pollinators) (2.5.5) (Traveset and Richardson, 2006). The effects of invasive alien pests and pathogens of pollinators are dealt with separately in the preceding section (2.4) on pollinator diseases and management.

The main sources (meta-analyses, reviews) and scope of evidence used in the assessment of the impact of invasive alien plants, pollinators, herbivores and predators on native pollinator species, networks and pollination are summarized in Table 2.5.1.

11. Table 2.5.1: Main sources (meta-analyses, reviews) and scope of evidence used in assessment of the impact of invasive alien plants, pollinators, herbivores and predators on native pollinator species, networks and pollination.

Citation	Study type	UN Geographical Region (numeric code)	Topic	Effect of invasive
Aizen et al. (2008). PLoS Biology 6, e31.	Meta-analysis	South America-Argentina (005-032), Eastern Africa-Mauritius (014-480), Southern Europe-Portugal (Azores) (039-620)	Impact of invasive alien plants or pollinators on networks	-
Albrecht et. al. (2014) Proceedings of the Royal Society-B: 281.	Meta-analysis	Northern Europe-UK (154-826), Western Europe-Germany (155-276), Southern Europe–Spain (039-724), Eastern Africa-Mauritius (014-480), South America-Argentina (005-032)	Impact of invasive alien plants on pollinator networks	+/=
Montero-Castano & Vilà (2012) Journal of Ecology 100, 884-893.	Meta-analysis	World (001)	Impact of alien species invasions on pollinators	Invasive animals: - Invasive plants: +/-/=
Morales & Traveset (2009) Ecology Letters 12, 716-728.	Meta-analysis	World (001)	Effects of alien invasive plants on pollinator visitation to and reproduction of native plants	Invasive plants: -
Carvalheiro et al. (2014) Ecology Letters, 17, 1389-1399.	Meta-analysis	World (001)	Effect of the abundance, relatedness and geographic origin of co-flowering plants on insect pollination	Invasive alien plants generally = (but if invasive floral traits match natives or invasive floral abundance is high then impact can be: +/-)
Bjerknes et. al. (2007) Biological Conservation 138, 1-12.	Review	Northern America-USA-Canada (021-840-124), Western Europe-Germany (155-276), Southern Europe–Spain (039-724), Northern Europe-Norway (154-578)	Effects of alien plant invasions on native plant pollination via competition for, or facilitation of, pollinator visits	+/-/=
Desurmont et al. (2014) Plant, cell & environment 37, 1854-1865.	Review	Northern America-USA (021-840)	Disruption of chemical signaling between plants and pollinators by invasive insect herbivores	Hypothesised only: -
Kenis et al. (2009) Biological Invasions, 11, 21-45.	Review	World (001)	Ecological effects of invasive alien insects	Invasive pollinators: =/-
Stout & Morales (2009). Apidologie 40, 388-409	Review	Northern America (021), Northern Europe (154), Western Europe (155), Eastern Europe	Impact of alien invasive species (plants &	Invasive plants: +/-/=; Invasive pollinators: =/-

		(151), Southern Europe (039), Asia (142)	pollinators) on pollinators and pollination	
Traveset & Richardson (2006) Trends in Ecology & Evolution 21, 208-216.	Review	World (001)	Impact of alien invasive insect & plant species on pollinators and pollination	Invasive pollinators: +/-/= Invasive herbivores: - Invasive predators: - Invasive plants: +/-
Traveset & Richardson (2014) Annual Review of Ecology, Evolution and Systematics 45, 89–113.	Review	World (001)	Effect of invasive alien species on mutualisms, including pollinators and pollination	Invasive plants: +/-
Bezemer et al. (2014) Annual Review of Entomology 59, 119-141.	Review	World (001)	Response of native insect pollinators and plants to invasive alien plants.	Invasive plants +/-/=
Dafni et al. (2010) Applied Entomology and Zoology 45, 101-113.	Review	World (001)	Impact of commercial *Bombus terrestris* introductions on native bumble bees	-
Dohzono & Yokoyama (2010) Applied Entomology and Zoology 45, 37-47.	Review	South America-Argentina-Brazil (005-032-076), Eastern Africa-Mauritius (014-480), Australia (036), Southern Europe–Spain (039-724), Asia-Japan (142-392)	Impacts of alien honey bee (*Apis mellifera*) and bumble bees on native plant-pollinator relationships	*Apis mellifera*: +/- *Bombus* -
Monceau et al. (2014) Journal of Pest Science 87, 1-16.	Review	Europe (150)	Potential impacts of invasive Asian hornet (*Vespa velutina*) on European honey bees	- (Predicted)
Morales & Traveset (2008) Critical Reviews in Plant Sciences 27, 221-238.	Review	World (001)	Impact of invasive alien plants for native plant reproduction	=/-
Paini (2004) Austral Ecology 29, 399-407.	Review	World (001)	Impact of the introduced honey bee (*Apis mellifera*) on native bees	Some - interactions, but impacts on fitness or population size either equivocal or =
Moritz et al. (2005) Ecoscience 12, 289-301.	Review	World (001)	Global invasions of the western honey bee (*Apis mellifera*) and the consequences for pollinators and pollination	Mostly =, but a few examples of - impacts

2.5.2 Invasive alien plants

Alien plant dispersal has increased worldwide, both accidentally (e.g., contamination of agricultural cargo) and deliberately (e.g., for horticulture) (Hulme, 2009). Introduced alien plants may establish and prosper because they: i) escaped biotic constraints; ii) occupy a vacant ecological niche - either pre-existing or due to ecosystem disturbance; iii) possess novel weapons or phenotypic plasticity conferring ecological advantage; and iv) evolved increased competitive ability following colonisation outside of their range (Bossdorf et al., 2005; Cappuccino and Arnason, 2006; Catford et al., 2012; Mack et al., 2000; Uesugi and Kessler, 2013).

When involved in mutualistic interactions (such as pollination), the interaction strength (extent of mutual dependence between interacting species shaped by the probability of encounter and their separate phylogenetic histories) may be important for the persistence of invasive plant species. Introduced mutualists may either fail or succeed in establishing within a novel ecological community according to the strength of interaction with the native species, for instance, if an introduced pollinator fails to obtain sufficient resources from the resident plant species then establishment is unlikely (Jones and Gomulkiewicz, 2012). Moreover, genetic diversity in introduced and resident species may, contingent on interaction strength, lead to rapid evolutionary selection for integration of the invader into the recipient community (Bossdorf et al., 2005; Jones and Gomulkiewicz, 2012; Vandepitte et al., 2014).

Insect-pollinated plant species often dominate lists of invasive alien plants, but at least in the early stages of colonization the ability of these plants to self-pollinate enables establishment and spread (Chrobock et al., 2013; Pysek et al., 2011; Traveset and Richardson, 2014). Over time, other plant traits (e.g., flower morphology, copious nectar or pollen rewards, large floral or long duration displays) lure and co-opt pollinator species whose phenotypes are pre-adapted to the floral resources the invasive alien plant offers (Chrobock et al., 2013; Kleijn and Raemakers, 2008; Morales and Traveset, 2009; Naug and Arathi, 2007; Pysek et al., 2011; Stout and Morales, 2009). Invasive alien plant species can thus become integrated into the ecosystem and dominate plant-pollinator interactions (Pysek et al., 2011; Traveset et al., 2013; Traveset and Richardson, 2006; Vilà et al., 2009). For example, pollen loads carried by insects may become dominated by alien pollen and hence potentially reduce conspecific pollen transfer among native plant species (e.g., Kleijn and Raemakers, 2008; Lopezaraiza-Mikel et al., 2007). Invasive flowering plants can also affect pollinators’ nutrition. Indeed, nutritional requirements differ among bee species and honey bee worker castes, and the growth and survival of social and solitary bee species is sensitive to the composition of the pollen diet (Paoli et al., 2014; Praz et al., 2008; Sedivy et al., 2011; Tasei and Aupinel, 2008). Therefore, while alien pollen and nectar may provide an additional food source for pollinators adapted to exploit them, there may also be a potential risk to pollinator health if invasive alien plant pollen is nutritionally poor compared to that from native plants (Stout and Morales, 2009).

Invasive plants are expected to affect pollinators adversely if they either ill-adapted to exploit the alien food resource or dependent on native plants outcompeted by the invader (Bjerknes et al., 2007; Palladini and Maron, 2014; Stout and Morales, 2009). There is, however, little evidence from meta-analyses or reviews (Bjerknes et al., 2007; Montero-Castaño and Vilà, 2012; Stout and Morales, 2009), and only very few individual examples (Lopezaraiza-Mikel et al., 2007; Moroń et al., 2009; Nienhuis et al., 2009) of alien plant invasions consistently lowering overall pollinator diversity or abundance.

There is more evidence, however, that alien plant invasions can influence the assembly of pollinator communities. Plant-pollinator community networks are permeable to plant invaders (Traveset et al., 2013; Traveset and Richardson, 2014), which according to the species involved can rewire plant-pollinator interactions (e.g., Bartomeus et al., 2008). Network architecture can often be relatively unaltered by alien plant invasions, for instance, a Pan-European analysis showed network nestedness, a property thought to confer a degree of stability on the community, was unaffected by the integration of alien plants into the network (Vilà et al., 2009). Although recent global meta-analyses have demonstrated changes in network properties following integration of alien invasive plants, the attractiveness of these invasive plants to native pollinators altered their behaviour, which led to changes in network properties (e.g. modularity, interaction strength) that are thought to enhance community stability (Aizen et al., 2008; Albrecht et al., 2014). For example, invasive plant species increased connectivity between network modules (subsets of tightly connected species) (Albrecht et al., 2014), which potentially increased the network's robustness to species losses arising from future environmental changes. Furthermore, highly invaded networks are, on average, characterised by weaker mutualism strength (i.e. weak or asymmetric mutual dependences between interacting species), a property that may reduce the probability of secondary extinctions should a partner species in the network be lost (Aizen et al., 2008). It should be noted, however, that many of these predictions around network robustness are derived from simulation models of empirical network data (frequency of pairwise species interactions at the community level). The challenges of collecting such data typically preclude greater biological realism (temporal network dynamics, species competition) being built into these simulations. Therefore, due caution is required in interpreting these insights from simulation models for community stability. Nonetheless, while invasive plant species do not generally alter diversity or abundance (Montero-Castaño and Vilà, 2012) through usurpation and domination of pollinator interaction networks, they often hold a key role in community organisation (Aizen et al., 2008; Albrecht et al., 2014).

This key role of invasive alien plants (and invasive alien pollinators – see section 2.5.3), once integrated into pollinator networks, has potential ramifications for individual native plant species. If the native plant becomes overly reliant on the invader for facilitation of pollination, then there is a potential risk to the native species should those connections become eroded or lost due to further environmental changes (Aizen et al., 2008).

Invasive alien plants may alter interactions between native plants and their pollinators either through competition for pollinator visitation (Bjerknes et al., 2007; Dietzsch et al., 2011) or by elevation of pollinator activity to the level where co-flowering native plant pollination is facilitated (Bjerknes et al., 2007; Cawoy et al., 2012; McKinney and Goodell, 2011). Primary and meta-analyses suggest that pollinator visitation rates to native plant species tend to decrease with plant invasion, suggesting that competition for pollinators may be the prevailing process (Brown et al., 2002; Montero-Castaño and Vilà, 2012; Morales and Traveset, 2009). Whether this translates into reduced reproductive output of native plant species is less certain, potentially because of plant compensatory mechanisms (i.e., self-reproduction, recruitment of alternative pollinators) (Bjerknes et al., 2007; Dietzsch et al., 2011; Morales and Traveset, 2009; Traveset and Richardson, 2014), but see examples where negative effects are reported (Brown et al., 2002; Chittka and Schurkens, 2001; Thijs et al., 2012). Furthermore, the level of impact on flower visitation may be contingent on the composition of the pollinator community because of differential responses of pollinator groups (e.g., flies versus bees) to the invasive plant (Carvalheiro et al., 2014; Montero-Castaño and Vilà, 2012). The negative impact that alien plant invasions can have on native plant pollination and reproductive success is increased at high relative densities of alien flowers and/or when alien and native plants are related or have similar floral traits (i.e., flower anatomy, color, phenology large floral displays) (Bjerknes et al., 2007; Brown et al., 2002; Carvalheiro et al., 2014; Morales and Traveset, 2009; Pysek et al., 2011). Only if some or all of these conditions are met will the extent of pollinator sharing between the native and the invasive plant species rise to the point where there is an impact, positive or negative, on the native plant (e.g. Thijs et al., 2012). There have been no studies (to our knowledge at the time of writing) that have examined the impact of invasive alien wild plants on food crops, which represents a significant knowledge gap. The pollinator-mediated impacts of native co-flowering plant species on flowering invasive plants are not clear and have been less studied (Carvallo et al., 2013).

2.5.3 Invasive alien plant pathogens

Alien, potentially invasive, plant pathogens may be directly introduced through human trade movements (e.g., crops) or associated with high levels of anthropogenic environmental impact, including the human-mediated spread of invasive plant species, in the recipient ecosystem (Santini et al., 2013). There is some understanding of how plant pathogens are spread by insect vectors, including pollinators (Shykoff and Bucheli, 1995), and how plant pathogens can influence pollinator visitation to affect pollination in co-flowering yet uninfected neighboring plant species (Roy, 1994). Thus there is potential for invasive alien plant pathogens, perhaps introduced along with invasive alien plants, to affect plant physiology or flowering, native plant-pollinator interactions and plant reproduction, however, this has been little studied to date.

2.5.4 Invasive alien herbivores and predators

Mammalian herbivores, such as ungulates (e.g., cattle, goats, deer), through consumption of floral or vegetative plant tissues or by direct trampling, have the potential to affect the floral or nesting resources available to pollinators (Traveset and Richardson, 2014). For instance, cattle introduced to Patagonian forests represent an invasive alien herbivore, which through trampling the vegetation indirectly altered pollinator network structure, visitation and the reproductive success of certain plant species (Vazquez and Simberloff, 2003, 2004).

Similarly, invasive insect herbivores, by attacking plant roots or shoots, can reduce floral resources to impact potentially an array of pollinator species (Louda et al., 1997; Traveset and Richardson, 2006). Insect herbivory can alter the emissions of constitutive or induced volatile organic compounds from the plant (Desurmont et al., 2014). Pollinators use such volatiles as olfactory cues to locate floral resources (Stokl et al., 2011; Theis et al., 2007), and insect herbivory can disrupt these signals to affect pollinator visitation and pollination (Barber et al., 2012; Kessler et al., 2011; Steets and Ashman, 2004). A recent review considered it likely this disruption of native plant-pollinator signals and pollination may arise as a result of herbivory by invasive insects, yet there has been little study to date of this aspect of invasion ecology (Desurmont et al., 2014).

Predators can also have strong indirect effects on pollination and plant fitness via consumption of pollinators or altering pollinator behaviours (Dukas, 2001, 2005; Knight et al., 2006). Invasive predators such as the cats, stoats and rats introduced to oceanic islands have reduced the population sizes of vertebrate pollinators (birds, lizards), with associated impacts on their mutualistic interactions with plants

(Traveset and Richardson, 2006). For example, in the Ogasawara archipelago of Japan, invading predatory lizards consumed endemic bee species to the point that the honey bee (*A. mellifera*), invasive in these islands, became the dominant pollinator (Abe et al., 2011). The preference of the honey bee for visiting flowers of invasive alien plants over flowers of endemic plants meant that the invasive predator transformed the ecological system by eliciting a shift from the native to an invasive-dominated pollination system (Abe et al., 2011). Insect predators also have the potential to disrupt pollinator communities and pollination. For instance, the yellow-legged hornet (*Vespa velutina*), a recent (2004) accidental introduction into Europe from Asia, is a predator of insects including bees and is thought to represent a direct or indirect threat to already stressed European honey bee populations, via altered ecological dynamics (Monceau et al., 2014). Invasive ant species in South Africa and Mauritius, which are more aggressive or competitive than native ants, can deter pollinator visitation, disrupt pollination and seed dispersal, thereby leading to reductions in plant fitness (Lach, 2007; Hansen and Müller, 2009). Alien parasitoids have been deliberately introduced worldwide for biocontrol of exotic agricultural pests. In many cases, these parasitoids have also reduced populations of indigenous non-target insects, including butterflies, moths and flies that are potential pollinators of native plant species (Louda et al., 2003). In New Zealand beech (*Nothofagus solandri* var. *solandri*) forests, invasive alien wasps (*Vespula vulgaris, V. germanica)* compete for energy-rich food, in the form of honeydew secretions produced by native scale insects, with alien honey bees (*A. mellifera*) and also native vertebrate (birds) pollinators. The wasps significantly appropriate and reduce this food resource thereby representing a threat to the native bird pollinators (Markwell et al., 1993; Moller et al., 1991). In Hawaii, the experimental removal of the invasive predatory wasp (*V. pensylvanica*) increased visitation rates to flowers of a native tree (*Metrosideros polymorpha*) by both native bees (*Hylaeus* spp) and the invasive alien honey bee *A. mellifera*. This change to species interactions resulted in greater fruit production of this tree species (Hanna et al., 2013). Removal of *V. pensylvanica* led to *A. mellifera* becoming the most effective pollinator in this system, likely replacing a niche previously fulfilled by extinct or declining bird pollinators, highlighting the complex nature of species interactions among predators, pollinators and plants, and the potential role invasive species may have in supporting pollination in human-modified ecosystems (Hanna et al., 2013).

2.5.5 Invasive alien pollinators

Certain bee species – introduced accidentally or intentionally to provide apicultural or pollination services to agriculture – can also disrupt native pollinator communities either by directly outcompeting indigenous insects for floral or nesting resources or by spreading pests and pathogens to which other pollinators are susceptible. Transmission of pest and pathogens through movements and use of managed bees is dealt with elsewhere (see section 2.4 on pollinator diseases and management). Here we assess the ecological impacts of human-mediated invasion of natural communities by alien bees.

The western honey bee (*A. mellifera*) has over centuries been transplanted worldwide for apiculture (production of honey, beeswax, etc.) and crop pollination and is often managed in large densities. The intentional and accidental movement of different honey bee (*Apis*) species continues (e.g., *A. mellifera* globally, *A. florea* into Israel, Sudan and Iran) (Goulson, 2003; Moritz et al., 2010; Moritz et al., 2005). Migratory beekeeping practice (see section 2.4) in South Africa trans-located the honey bee sub-species *A. mellifera capensis* into the range of *A. m. scutellata*, where it behaved as a social parasite, leading to substantial colony losses of *A. m. scutellata* (Moritz et al., 2005). This is a stark example of a negative interaction between *Apis* sub-species. However, there is little evidence that the human-assisted movement of the principal managed pollinator, the European sub-species (*A. m. mellifera*) into the regions (Europe, Africa) where other sub-species of *A. mellifera* are endemic has had a significant impact on these conspecifics (Moritz et al., 2005). Moreover, while *A. mellifera* introductions into ecosystems that lack other subspecies but contain other congeneric *Apis* species (i.e., East Asia) can lead to interspecific competition for floral and nesting resources, overall there is little sign that the net effect is the domination or replacement of the indigenous *Apis* species (Moritz et al., 2005). However, in China, the distribution and population size of *A. cerana* in China has reduced by over 75% and 80%, respectively, following the introduction of *A. m. ligustica* in 1896. Coupled with overall losses of food and nesting resources, direct competition with *A. m. ligustica* and inter-species transfer of pathogens (e.g. Sacbrood viruses) to *A. cerana* have been implicated in this decline (Ji et al., 2002; Yang, 2005). In the Americas, a region lacking indigenous congeneric *Apis* species, various sub-species of *A. mellifera* were introduced for apiculture and became naturalized in North America ~ 250 years ago. More recently, feral descendants of the introduced African honey bee sub-species *A. m. scutellata* (again introduced for apiculture) have spread across tropical America and into the southern USA (Goulson, 2003; Moritz et al., 2005). It is possible that the lack of native *Apis* spp. in the Americas means the introduced honey bee has occupied a vacant ecological niche, although they do have the same proboscis length as workers of the North American short-tongued bumble bees (Inouye, 1977). The consequences of this invasion for non-*Apis* pollinators are not clear, either because it had little effect or the historical impacts went unrecorded (Moritz et al., 2005).

Overall, alien honey bee populations have become readily integrated into pollinator communities and direct competition for food has sometimes altered native wild bee behaviour and reproductive success in a locale, although these species interactions are highly dynamic (Dohzono and Yokoyama, 2010; Roubik, 1980; Roubik and Wolda, 2001; Thomson, 2004; Traveset and Richardson, 2006). There have been very few reports of invasive alien honey bees reducing the survival or densities of native wild bees through competition (Kenis et al., 2009; Paini, 2004; Roubik and Wolda, 2001; Yang, 2005) and to date no extinctions have been recorded (Goulson, 2003; Moritz et al., 2005; Paini, 2004; Traveset and Richardson, 2006). However, it is possible that alien honey bee invasions may have contributed to historic declines of native pollinators in places like oceanic islands (Kato and Kawakita, 2004; Magnacca, 2007). Behavioural interactions between alien honey bees and native pollinators (bees and birds) have been documented both reducing and enhancing pollination of native plants and crops (Brittain et al., 2013; Dohzono and Yokoyama, 2010; Greenleaf and Kremen, 2006; Traveset and Richardson, 2006). The ecological traits of the honey bee (e.g., high dispersal, generalised feeding habit, forager recruitment), have also maintained pollination function once they have been introduced to areas where indigenous pollinator species losses have been incurred by anthropogenic disruption of natural ecosystems (Aizen and Feinsinger, 1994; Dick, 2001; Hanna et al., 2013). There is potential, however, for micro-evolutionary effects on wild plant-pollinator networks arising from *A. mellifera* introductions. Mathematical models have predicted that the widespread introduction of this super-generalist honey bee may promote convergence in flower traits across many wild species, which may alter the functioning and structure of wild plant-pollinator communities (Guimaraes et al., 2011).

The introduction (see section 2.4) and subsequent establishment of feral populations of bumble bee (*Bombus*) species has led to some disruption of indigenous pollination systems involving native congeners (Dohzono and Yokoyama, 2010; Kenis et al., 2009; Morales et al., 2013). Many native plant taxa in the temperate, alpine and arctic zones of the world have evolved to become closely associated with different bumble bee species. A few bumble bee species are managed for crop pollination (see section 2.4), but in particular the life-history traits of *B. terrestris* have predisposed its commercial rearing and translocation around the world for crop pollination (Dohzono and Yokoyama, 2010). Many of these traits (e.g., nesting flexibility, generalist feeding habit) have also enabled this species once introduced to establish successfully in novel temperate environments such as in Australasia, Japan, Israel and South America (Dafni et al., 2010; Morales et al., 2013). Introduced alien bumble bee species can transmit novel pathogens into native bee populations (see section 2.4) and often compete with native congeners for nesting and floral resources (Dafni et al., 2010; Dohzono and Yokoyama, 2010; Ings et al., 2006). This competitive displacement of native pollinators by alien ones can reduce native plant species richness and abundance and promote processes leading to inbreeding depression (by enhancing selfing) or hybridization (by moving pollen across closely related alien and native plants) and ultimately lower fitness of plants (Dohzono and Yokoyama, 2010; Kenta et al., 2007; Morales et al., 2013; Morales and Traveset, 2008). Certain solitary bee species have been introduced, some possessing similar traits to invasive social bees, but relatively little is known about their impact on the ecology of native pollinators; representing a gap in understanding that could help to forecast impacts of future invasions (Goulson, 2003; Kenis et al., 2009).

2.5.6 Vulnerability of different pollinator habitats to invasions

As invasions are primarily a human-mediated process, anthropogenic and disturbed environments are likely to be prone to the immigration and establishment of alien species, for example where human activity creates or makes accessible new niches (Catford et al., 2012; Mack et al., 2000). A recent global meta-analysis suggested that the tendency for alien invasions to reduce pollinator diversity or abundance was both statistically non-significant and did not differ among forest, shrubland, and grassland ecosystems (Montero-Castaño and Vilà, 2012). While these broad ecosystem classifications were necessary for this meta-analysis due to data limitations, they were lacking important contextual information (e.g., level of disturbance or human activity, carrying capacity of recipient habitat, mainland vs. island), which may have affected the sensitivity of the analysis (Mack et al., 2000). Oceanic island ecosystems may be particularly vulnerable to disruption of pollination systems, at least where those ecosystems support a smaller and more specialised plant-pollinator fauna (Abe et al., 2011; Hansen and Müller, 2009; Mack et al., 2000; Traveset et al., 2013; Traveset and Richardson, 2006). Island pollination systems tend to be more robust when the native pollination system is generalised and thus the invasive alien species becomes integrated without significant disruption (Kaiser-Bunbury et al., 2011; Olesen et al., 2002). Although because of likely different demographic processes, populations of animal or plant species that are rare or restricted to oceanic islands or have undergone a strong recent genetic bottleneck related to habitat fragmentation are likely to have less genetic diversity than more common or less spatially restricted species (Darvill et al., 2006; Eckert et al., 2010; Frankham, 1997; Stuessy et al., 2014). The impoverished genetic diversity of such species may thus affect adaptive processes that could contribute to the success or failure of invasions, depending of the type of interaction they have with the invasive species. For instance, modeling approaches indicated that a higher ability to adapt (higher genetic diversity) in the invasive species generally leads to establishment, and further, higher genetic diversity in the resident species can lead to exclusion of the invasive in predator-prey interactions, and may allow adaptation to the invasive (and thus favor invasion) and survival of both species (mutualism or competition) (Jones and Gomulkiewicz, 2012).

2.5.7 Conclusions

The outcome of an invasion on pollinator populations, diversity, network structure or pollination processes is likely to be highly contingent on the ecological and evolutionary context. For example, invasive plant species are often readily incorporated into native pollination networks, especially where generalised plants and pollinators predominate. This can have major consequences for the function, structure and stability of pollinator networks, negative impacts on particular native pollinator species and, less commonly, reductions in overall pollinator abundance or diversity. The ramifications of such changes for native plant pollination can be positive or negative depending on the traits of the species involved. By altering the plant community, introduced mammal herbivores can have a profound effect on pollinator communities and pollination, but the effects of invasive insect herbivores are unknown. Invasive predators can directly kill pollinators or disrupt pollinator communities and associated pollination systems, whilst invasive pollinators can outcompete or transmit diseases to native pollinator species or simply be accommodated in the existing pollinator assemblage. The ecological complexity and context of different invasions precludes overall generalisation. Nonetheless, the trophic position (plant/herbivore/pollinator/predator) of an invasive species and the degree of specialisation in the invasive and the recipient pollination system are crucial to understanding the outcome of alien species invasions. There is also a risk that the impact of invasive alien species on pollinators and pollination may be further exacerbated when it occurs in combination with other threats (section 2.7) such as diseases, climate or land-use change (González-Varo et al., 2013; Schweiger et al., 2010; Vanbergen and the Insect Pollinators Initiative, 2013).

2.6 Climate change

2.6.1 Vulnerability of biodiversity and ecosystems to climate change

Climate change “refers to a change in the state of the climate that can be identified … by changes in the mean and/or the variability of its properties, and that persists for an extended period, typically decades or longer. Climate change may be due to natural internal processes or external forcings such as modulations of the solar cycles, volcanic eruptions and persistent anthropogenic changes in the composition of the atmosphere or in land use” (IPCC, 2013). Species respond to climate change by adaptation, by moving out of unfavorable into favorable climates, or by going first locally and later globally extinct (Dawson et al., 2011, Bellard et al., 2012). Climate change is regarded as one important factor contributing to the decline of pollinators (Potts et al., 2010) and changes and disruptions of plant–pollinator interactions (Memmott et al., 2007; Hegland et al., 2009). Vulnerability of biodiversity and ecosystems to climate change is defined as the combination of three things: a) the degree to which their climatic environment has or will change relative to conditions under which they evolved; b) the sensitivity of the ecosystem processes to the elements of climate which are changing; and c) the degree to which the system can maintain its structure, composition and function in the presence of such change, either by tolerating the change or adapting to it (Settele et al., 2014; see Oppenheimer et al., 2014 for a comprehensive discussion on vulnerability concepts).

2.6.2 Evidence of changes in ecosystems, pollinators and pollination

2.6.2.1 Phenology change and interaction mismatch

Monitoring of the phenology of biological events across a large number of sites worldwide has allowed the detection of an advance in spring events (breeding, bud burst, breaking hibernation, flowering, flight time, migration) for many plant and animal taxa in many regions, especially in the northern hemisphere (e.g., Europe, North America, Arctic) but also some in the southern hemisphere and in tropical areas (e.g., Africa, Australia, South America, Antarctica). Studies on plants include Cleland et al. (2007), Amano et. al. (2010) and Gordo and Sanz (2010), while plants and animal taxa in combination have been dealt with by Høye et al. (2007), Primack et al. (2009), and McKinney et al. (2012). Meta-analyses based on observation studies were conducted by Parmesan (2006, 2007), Cook et al. (2012b), Ma and Zhou (2012), and Wolkovich et al. (2012), while those of Cleland et al. (2012) and Wolkovich et al. (2012) were based on warming experiments.

Generally, there is great intra- and interspecific variability in phenological responses to changing climatic factors. Insect species with phenotypic plasticity in their life-cycle may increase in number of generations per year due to increase in temperatures and length of growing seasons (e.g. due to the contraction of the onset and cessation of winter frosts; Menzel et al., 2006; Robinet and Roques, 2010). Uncertainties and biases are introduced in research that (1) compares different taxonomic groups or geographic regions with incomplete or non-overlapping temporal and/or spatial time series and scales, or (2) fails to consider the effects of local climatic variability (e.g., wind speed, climatic conditions at stop-over places during migrations) or the mostly unknown pressures on winter ranges for migratory species (Hudson and Keatley, 2010). Further, if time series are too short, long-term trends in phenological changes cannot be detected, although responses to annual climate variability can often be characterized. Cross-taxa observations show high variation in species- and location-specific responses to increasing temperatures in both direction and magnitude (e.g. Parmesan, 2007; Primack et al., 2009).

Changes in interspecific interactions stemming from changes in phenological characteristics and breakdown in synchrony between species have been reported (Gordo and Sanz, 2005). Species unable to adjust their behavior, such as advancement of spring flowering in response to temperature, are likely to be negatively affected, if for example, their pollinators do not respond to the same signals. The degree, direction and strength of the asynchrony due to changing climatic variables depends on differences in the phenology of the interacting species (van Asch and Visser, 2007). Increasing temperatures may either increase or decrease synchrony between species, depending on their respective starting positions (Singer and Parmesan, 2010). Climate changes (e.g. warming, elevated CO_2) and its consequences (e.g. increased drought) may affect the synchrony between plants and pollinators by altering the chemical signals emitted by plants (floral volatiles) to attract pollinators (Farre-Armengol et al., 2013). For example, increased temperatures may elevate the overall rate of volatile emissions, and hence the strength of the signal to pollinators, but alter the chemical composition potentially affecting the ability of specialist pollinators that rely on species-specific floral bouquets to locate food-plants (Farre-Armengol et al., 2014). However, the consequences of individual and multiple climate-stressors on pollination are likely to be complex due to different impacts on various plant biochemical pathways and biotic interactions and much remains to be understood (Farre-Armengol et al., 2013).

Generally, changes in synchrony of interacting species are assumed to affect ecological community dynamics, such as trophic cascades, competitive hierarchies and species coexistence (Nakazawa and Doi, 2012). For example, fig plants are keystone species in tropical rain forests at the centre of an intricate web of specialist and generalist animals. Jevanandam et al. (2013) report that fig plants have a reciprocally obligate mutualism with tiny, short-lived (1-2 days) fig wasps (Agaonidae). Their results of experiments from equatorial Singapore suggest that the small size and short life of these pollinators make them more vulnerable to climate change than their larger and longer-lived hosts. An increase of 3°C or more above the current temperatures across much of the equatorial tropics would markedly decrease the active adult lifespan of all four species investigated. Unless fig wasps can acclimate or adapt to warmer temperatures in time, these responses may disrupt the mutualism, potentially affecting multiple trophic levels (Jevanandam et al., 2013).

Insects show a variety of phenological responses to changing temperatures (reviewed in Robinet and Roques, 2010). In a 2009 review, Hegland et al. (2009) find empirical evidence for linear relationships between phenological events and temperature in both plants (e.g., first flowering) and pollinators (e.g., first emergence date), however, they also emphasise that temporal mismatch may still occur due to the varying slopes of the linear relationships in the two mutualistic partners. Observations that show the phenological de-coupling of plants and their pollinators are also presented by Gordo and Sanz (2005) and McKinney et al. (2012). Parmesan (2007) found that butterflies showed spring advancement three times stronger than for herbs and grasses. Because most butterflies use herbs and grasses as host plants, this suggests an increasing asynchrony between these two interacting groups. Bartomeus et al. (2011) similarly reported that the phenology of 10 bee species from northeastern North America has advanced by about 10 days over about 130 years with most of this advance occurring since 1970; however, this rate of advance in bee phenology was not distinguishable from co-occurring advances in forage plant phenology. This suggested that the phenology of generalist bee species, such as those investigated by Bartomeus et al. (2011), will keep pace with shifts in forage-plant flowering. This view is supported by experimental evidence (Willmer, 2012), which also suggests that phenological effects on pollinator-plant synchrony may be of limited importance. However, an analysis of phenological observations in plants (Wolkovich et al., 2012) showed that experimental data on phenology may grossly underestimate phenological shifts.

Shifts in flowering phenology can reshape entire plant and pollinator communities (CaraDonna et al., 2014). Earlier snowmelts are reported to decrease floral resources and can hence affect survival of associated insects (Boggs and Inouye, 2012). In temperate, arctic and alpine habitats, snow cover is a more important factor than temperature per se. Interactions between temperature and precipitation determine snowmelt changes, which are reported to lead to earlier flowering and appearances of plants and arthropods in Greenland (Høye et al., 2007), earlier flowering in an alpine plant (Lambert et al., 2010), and an increase in frost damage to montane wildflowers (Inouye, 2008).

In a modelling study, Memmott et al. (2007) used a highly resolved empirical network of interactions among 1420 pollinator and 429 plant species to simulate the consequences of phenological shifts in plant-pollinator networks that can be expected with a doubling of atmospheric CO_2. They found that diet breadth (i.e., number of plant species visited) of the pollinators might decrease due to the reduced phenological overlap between plants and pollinators and that extinctions of plant, pollinators and their crucial interactions could be expected as consequences of these disruptions. While there are methodological shortcomings (e.g., sampling effects and rarity are both confounded with specialization; Blüthgen, 2010), and while the results of Benadi et al. (2014) suggest that many pollinator species are not threatened by phenological decoupling from specific flowering plants, a follow-up empirical study by Burkle et al. (2013) in which the highly resolved network analysed by Memmott et al. (2007) was resampled. Their empirical evidence suggests that climate change over the last 120 years may have resulted in phenological shifts that caused interaction mismatches between flowering plants and bee pollinators. As a consequence, many bee species were extirpated from this system, potentially as a result of climate-induced phenological shifts.

In a study on the influence of climatic seasonality on tropical pollinator groups Abrahamczyk et al. (2011) it was stated that the species richness of pollinating hummingbirds was most closely correlated to the continuous availability of food, that of bees and wasps to the number of food planst species and flowers, and that of butterflies to air temperature. In relation to climate change the authors state that all pollinators will likely be directly affected by an increase in climatic seasonality and indirectly by changes in the distribution and phenologies of food plants (see Potts et al., 2009), with the latter being especially likely for hummingbirds, bees, and wasps in their study system.

In summary, an increasing number of observational and experimental studies across many organisms provide strong evidence that climate change has contributed to the overall spring advancement observed especially in the Northern Hemisphere (Settele et al., 2014). Additionally, there is some evidence that daily activity patterns may change with climate change (e.g. Rader et al., 2013). However, the effects of these shifts in terms of interacting species are still not well understood and require further investigation (Burkle and Alarcon, 2011; Bartomeus et al., 2011). It can be assumed, that climate change-induced shifts in phenology may be a particularly pronounced problem for migratory pollinator species in temperate regions, with numerous moths (Macgregor et al., 2015) and other insect groups being well known for their extended migrations. However, how climate change influences most migratory pollinators has not been studied.

2.6.2.2 Observed changes in species range and abundance

Observed changes in species abundance are difficult to relate to climate change, because of the complex set of factors mediating population dynamics in non-managed (wild) populations. Some of the clearest examples of climate-related changes in species populations come from high-latitude ecosystems where non-climate drivers are of lesser importance (see Settele et al. 2014, Kiøhl et al., 2011, Hegland et al., 2009). There are also documented changes in effective population size in response to climate change since the last glacial maximum, demonstrating the potential plasticity in certain populations (López-Uribe et al., 2014; Groom et al., 2014).

Some examples of climate change impacts on pollinator abundance are reported from Indonesia. Wild honey bees *Apis dorsata* perform annual migration cycles, which are influenced by seasonal (Kahono et al., 1999; Kahono, 2002) and non-seasonal periods of natural flowering (Itioka et al., 2001). Wetter climate has led to a decline in food resources of *Apis dorsata* (flowering plants) and thus a decrease in the number of *Apis dorsata* colonies, largely as a result of a failure to produce new queens (Kahono, 2011). However, the increasing volume and frequency of rainfall in Indonesia and the consequent reduction in food resources have also led to a sharp decline in honey production by both *Apis dorsata* and the western honey bee *Apis mellifera* (Kahono, 2011). Kullu farmers, who practice traditional beekeeping with *Apis cerana* in the Himalayas, report that the potential effects of climate change on pollinators have included changes to swarming times (advanced by at least a month) and also changes in population sizes (Sharma 2004).

There is substantial evidence that terrestrial plant and animal species' ranges have moved in response to warming over the last several decades and this movement will accelerate over the coming decades (Settele et al., 2014). A synthesis of range shifts by Chen et al. (2011) indicates that terrestrial species (e.g. butterflies) have moved poleward about 17 km per decade (sites in Europe, North America and Chile) and 11 m per decade in altitude up mountains (sites in Europe, North America, Malaysia, and Marion Island), which corresponds to predicted range shifts due to warming. The "uphill and poleward" view of species range shifts in response to warming is a simplification of species response to changing climate, since response to climate change is also conditioned by changes in precipitation (including desertification), interactions with land use, and possibly many other factors (see section 2.7). These interactions can lead to responses that are not predictable from warming alone (Rowe et al., 2010; Crimmins et al., 2011; Hockey et al., 2011). For bumble bees, Kerr et al. (2015) report on consistent trends across continents in failures to track warming through time at species’ northern range limits (based on averaging observations across species), although range losses from southern range limits and shifts to higher elevations among southern species occur, leading to range contractions. Their study underscores the need to test for climate impacts at both leading and trailing latitudinal and thermal limits for species.

Detailed investigations of the mechanisms underlying observed range shifts show that there are many confounding factors (e.g., Crimmins et al., 2011; Hockey et al., 2011), but our ability to detect range shifts and attribute them to changes in climate has drastically improved (Parmesan and Yohe, 2003; Parmesan, 2006; Chen et al., 2011; Kerr et al., 2015). There is, however, tremendous variation in range shifts among species and species groups. Much of this variation can be explained by large differences in regional patterns of temperature trends over the last several decades (Burrows et al., 2011) and by differences in species dispersal capacity, life history and behavior (Lenoir et al., 2008; Devictor et al., 2012a, 2012b), like an increased susceptibility of rare or highly specialized pollinator species to changes in ranges and composition of plant communities (Goulson and Darvill, 2004; Williams, 2005).

On average, plants and animals appear to be tracking recent changes in climate (Chen et al., 2011), but there is also evidence that many species are lagging far behind (Lenoir et al., 2008; Devictor et al., 2012a). Species with short life cycles and high dispersal capacity – such as butterflies or herbaceous plants – are generally tracking climate more closely than longer-lived species or those with more limited dispersal such as birds (in terms of expanding their breeding ranges) and trees (Lenoir et al., 2008; Devictor et al., 2012a). While the rate of climate change in Europe was equal to a 249 km northward shift between 1990-2008, butterfly and bird community composition only made average shifts of 114 km and 37 km northward shifts, respectively; thus leading to an accumulated climatic debt of 212 km for birds and 135 km for butterflies (Devictor et al., 2012a).

2.6.2.3 Models and scenarios on changes in species range and abundance

Models indicate that range shifts for terrestrial species will accelerate over the coming century. Much of the contemporary work relies on species distribution models that predict future ranges based on current relationships between climate and species distribution (a.k.a., "niche" or "bioclimatic envelope" models; Peterson et al., 2011), applied to projected future climates. A variety of mechanistic species distribution models are also being developed and more widely used (e.g., Dawson et al., 2011; Cheaib et al., 2012). Recent model comparisons suggest that niche models often predict larger range shifts than more mechanistic models (Morin and Thuiller, 2009; Kearney et al., 2010; Cheaib et al., 2012). Most models do not realistically account for species migration rates, so they generally indicate changes in areas of favorable and unfavorable climate from which shifts in species distribution are inferred (but see Midgley et al., 2010 and Meier et al., 2012 for examples of models that include migration). Pacifici et al. (2015) present a review on the assessment of species vulnerabilities to climate change. They describe the three main approaches (correlative, mechanistic and trait-based), and their associated data requirements, spatial and temporal scales of application and modelling methods.

Major findings of niche modeling studies can be summarized as follows. In regions with weak climate gradients (e.g., little altitudinal relief), most species would need to migrate many 10s to 100s of km by the end of the century to keep pace with climate change (Leadley et al., 2010). Species that cannot migrate will see their favorable climate space diminish or disappear, but migration that keeps pace with climate change would allow some species to increase their range size (Thomas et al., 2006). Models that account for migration mechanisms indicate that many species will be unable to keep pace with future climate change due to dispersal and establishment limitations (Nathan et al., 2011; Meier et al., 2012; Renwick et al., 2012). Biotic interactions such as pollination or predator-prey networks can be disrupted due to decoupling of range overlaps or phenological mismatches, and this may cause much greater impacts on biodiversity than generally predicted (Memmott et al., 2007; Schweiger et al., 2008; Bellard et al., 2012; Nakazawa and Doi, 2012). Climate change mitigation would substantially reduce the distance that species would need to migrate to track favorable climates (Thuiller et al., 2005). Topography also reduces the distance, but as you go up mountains you have less area available.

Two studies have used envelope modelling techniques to investigate the effects of climate change on important pollinators of a particular crop, always however with underlying uncertainties due to limitations of the data (e.g. bias, unknown sampling effort, patchiness). Giannini et al. (2013) estimated present and future distributions of important passion fruit pollinators (four large carpenter bee *Xylocopa* species) and 33 plant species they rely on for nectar and pollen when passion fruit is not flowering, in mid-Western Brazil under a moderate climate change scenario. The study showed a substantial reduction and northward shift in the areas suitable for passion fruit pollinators by 2050. Polce et al. (2014) modelled the present and future distributions of orchard crops in the UK, and 30 species of bees and hoverflies known to visit fruit tree flowers under a medium emissions scenario. They showed that the present distribution of orchards in the UK largely overlaps with areas of high pollinator richness, but there could be a substantial geographical mismatch in the future (2050), as the area with climate most suitable for orchards moves substantially north and west. Future ranges also have been projected for some bee species using the approach in Europe (Roberts et al. 2011) and South Africa (Kuhlmann et al. 2012), and in particular for bumble bees and butterflies on a European continental scale (see Box 2.6.1). Giannini et al. (2012) modelled a decrease in bee habitats due to climate change in Brazil. However, the possibility that pollinators gradually change their target plant species is not taken into account in such approaches. There are indications for such shifts (Schweiger et al., 2008) which would mean that there is no necessity to move with the current plant species. Instead, this is a component of novel ecosystems evolving under climate change.

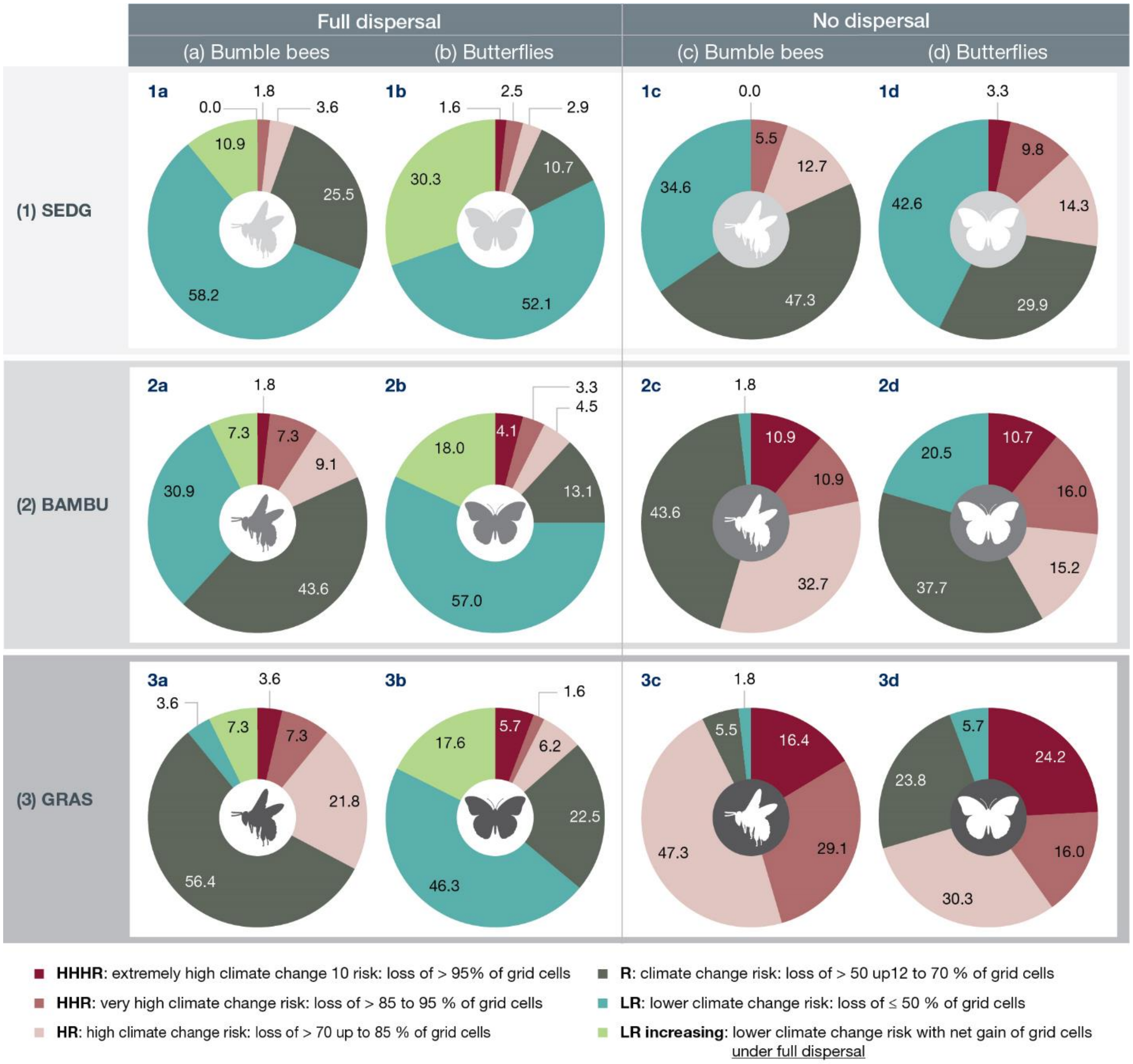
Box 2.6.1: ***Climatic risks for bumble bees and butterflies in Europe***

Due to the large number of species assessed in a very similar overall approach, we want to highlight some core results of two large-scale studies available on the potential future distribution on butterflies (Settele et al. 2008) and bumble bees (Rasmont et al. 2015a) in Europe.

Both studies used three scenarios that were based on storylines developed within the EU FP6 project ALARM (www.alarmproject.net; Settele et al., 2005; Spangenberg et al., 2012). The three scenarios were: **a) SEDG, S**ustainable **E**urope **D**evelopment **G**oal scenario – a storyline for moderate change which in the climate change component approximates the IPCC (Intergovernmental Panel on Climate Change) B1 scenario with a mean expected temperature increase in Europe until 2100 by 3.0°C; **b) BAMBU, B**usiness-**A**s-**M**ight-**B**e-**U**sual scenario – a storyline for intermediate change, which approximates the IPCC A2 scenario with an expected increase in temperature until 2100 of 4.7°C; and **c) GRAS, GR**owth **A**pplied **S**trategy scenario – a storyline for maximum change which approximates the IPCC A1FI climate change scenario. Here the mean expected increase in temperature until 2100 is 5.6°C.

Based on these scenarios, the future distributional ranges of bumble bees and butterflies have been projected for the years 2050 and 2080 (butterflies), and 2050 and 2100 (bumble bees), with climate envelopes derived from the distribution of the species from 1970 to 2000 (bumble bees) and 1980 to 2000 (butterflies), respectively. Changes have been quantified in numbers of 10’x10’ grid cells and led to the following climate risk categories used throughout both studies: HHHR extremely high climate change risk: loss of > 95% of grid cells; HHR very high climate change risk: loss of > 85 to 95% of grid cells; HR high climate change risk: loss of > 70 up to 85% of grid cells; R climate change risk: loss of > 50 up to 70% of grid cells; LR lower climate change risk: loss of ≤ 50% of grid cells; PR potential climate change risk: any change in number of grid cells, but modelling of present distribution had a low reliability.

Results of the different scenarios for the years 2080 (butterflies) and 2100 (bumble bees) are presented in Figure 2.6.1 for the 244 butterfly and the 56 bumble bee species that could be modelled reasonably well (modelling results were of low reliability for an additional 50 butterfly and 13 bumble bee species). A species’ ability to disperse and colonise new potentially suitable areas in the course of climate change is a key factor to predict species responses to climate shifts. However, as detailed dispersal distances are not available for most species two extreme assumptions/scenarios have been simulated: a) unlimited or full dispersal, such that the entire projected niche space denotes the actual future distribution, and b) no dispersal, in which the future distribution results solely from the overlap between current and future niche space.



15. Figure 2.6.1 Climate change risk categories of European bumble bees and butterflies
(assuming full dispersal and no dispersal; see text in Box 2.6.1 for definition of categories and scenarios).

Under the extreme, no dispersal GRAS scenario for bumble bees and butterflies (years 2100 and 2080, respectively) 16% (9 species) of the modelled European bumble bees and 24% (59 species) of the analysed butterflies are projected to be at an extremely high climatic risk, 29% (16 spp.) and 16% (39 spp.) will be at very high risk, 46% (26 spp.) and 30% (74 spp.) are at high risk, 5% (3 spp.) and 24% (58 spp.) are at risk, and only 2% (1 sp.) and 6% (14 spp.) are at low risk.

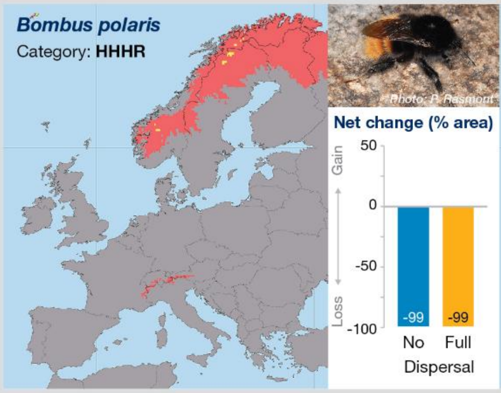
Because bumble bees are mainly better adapted to colder conditions, they show a higher degree of vulnerability to climate change than butterflies: only 7% of bumble bees compared to 30% of butterflies were categorized as R or LR. In addition to the projections of the modelled bumble bee and butterfly species, species that were not modelled due to data limitations are all very rare and localized in distribution and so their ranges are very likely to shrink considerably in any global change situations. Only a limited number of species are projected to benefit from climate change under a full dispersal assumption (and given there are adequate floral or larval resources) and can potentially enlarge their current distributions in Europe: among the bumble bees there are only 7% (4 species), including *Bombus zonatus* (see Figure 2.6.2), while 18% (43 species) of butterflies could potentially profit, including Apatura metis (see Figure 2.6.2).

Note: Contrary to the mostly cold-adapted bumble bees, many more solitary bees might benefit in a way similar to B. zonatus*, as they are more frequently adapted to drier and warmer climates and thus show higher diversity in, for example, the Mediterranean regions.*

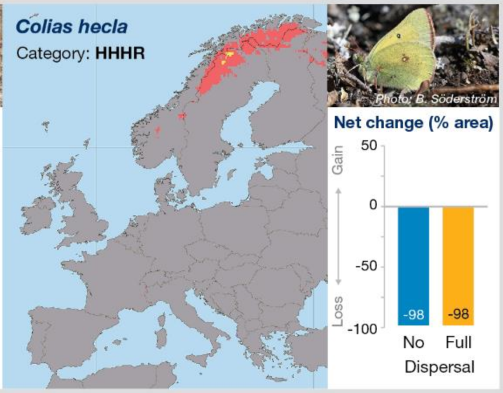
As could be expected, the three scenarios considered provide considerably different projections for 2080 and 2100, respectively. While under the moderate change scenario (SEDG) only 8 butterfly and no bumble bee species are projected to be at the verge of extinction, 26 butterflies and 6 bumble bees are at this particular high risk under the intermediate change scenario (BAMBU). Under the most severe change scenario (GRAS) 59 butterflies and 9 bumble bees are projected to lose almost all of their climatically suitable area.

Characteristics | **Bumblebee examples** | **Butterfly examples**

- Boreo-alpine species
- **Extremely high climate change risk** independent of dispersal as the climatic space vanishes

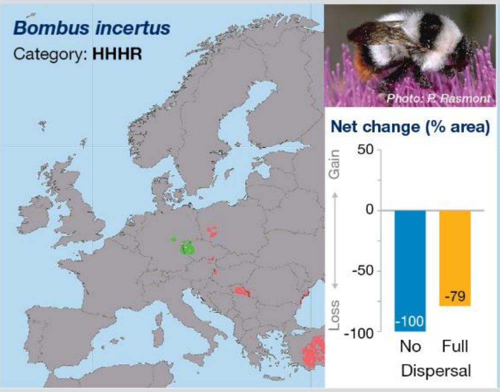


Bombus polaris
Category: **HHHR**

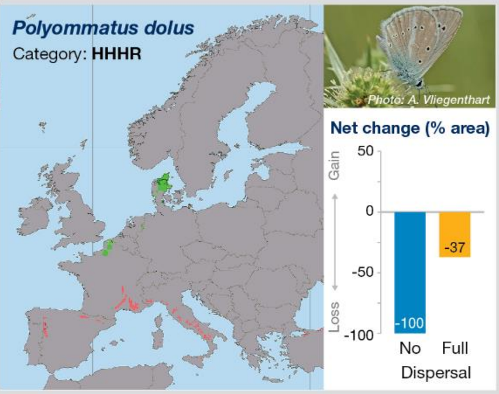


Colias hecla
Category: **HHHR**

- Patchy distribution pattern
- **Complete loss with "no dispersal"**
- Much lower losses under "full dispersal"

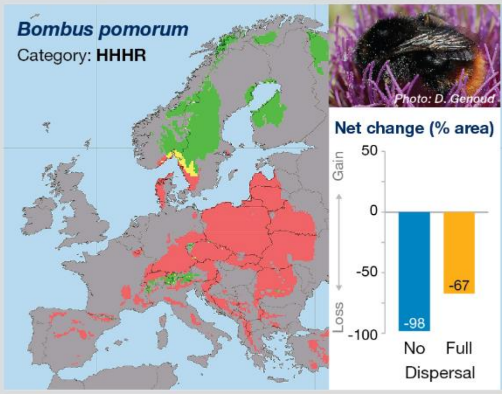


Bombus incertus
Category: **HHHR**

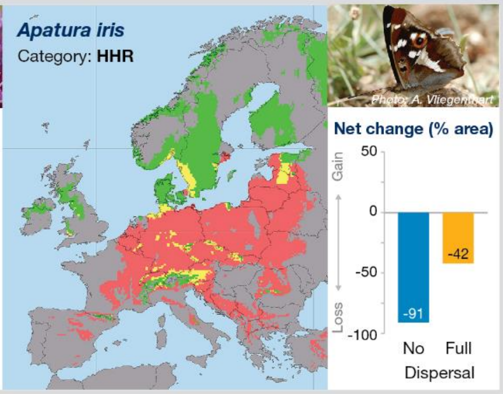


Polyommatus dolus
Category: **HHHR**

- More widespread species
- High losses under "no dispersal"
- **Moderate losses with "full dispersal"**

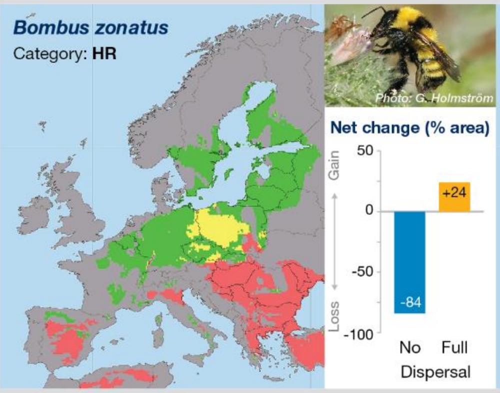


Bombus pomorum
Category: **HHHR**

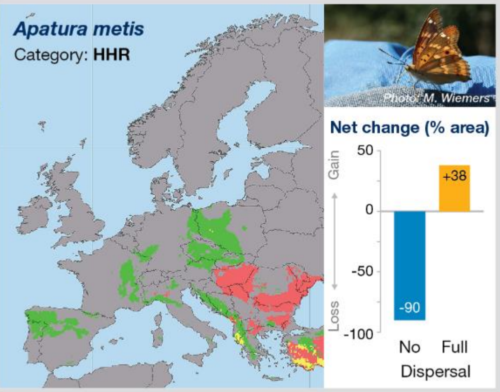


Apatura iris
Category: **HHR**

- More widespread species
- High losses under "no dispersal"
- **Gains under "full dispersal"**



Bombus zonatus
Category: **HR**



Apatura metis
Category: **HHR**

🟥 Lost distribution (becomes climatically unsuitable for species by 2080/2100)

🟨 Stable distribution (remaining climatically suitable area by 2080/2100)

🟩 Gained distribution under "full dispersal" assumption (become climatically suitable by 2080/2100)

16. Figure 2.6.2: Butterfly and bumblebee examples of climate spaces within the GRAS scenario, according to different distributional characteristics (maps: bumblebees after Rasmont et al., 2015a, butterflies after Settele et al., 2008).

HHHR (extremely high climate change risk); HHR (very high climate change risk); HR (high climate change risk); AUC (Area under curve – the closer the value is to 1, the better is the model).

For many species, especially the very cold-adapted ones in Alpine and Artic regions such as the bumble bees *B. alpinus, B. balteatus and B. polaris*, or the butterflies *Boloria chariclea*, *Euphydryas iduna* and *Colias hecla* (see Figure 2.6.1), their dispersal abilities are actually irrelevant for the assessment of their future fate because climate change will only lead to reductions of areas with suitable climatic conditions while no new suitable regions will emerge. These projected changes can be expected to lead to changes in the threat status as has been currently assessed by the IUCN Red List (Rasmont et al., 2015b). The additional threat posed by climate change would lead to an increased number of threatened bumble bee species. Currently, there are 18 species (of a total of 69 species) considered as threatened in Europe in the IUCN Red List, mostly because of climate change. However, under the moderate change scenario (SEDG) the number of threatened bumble bee species may be clearly above 20 while under the most severe change scenario (GRAS) it could rise to above 40 threatened species.

Due to considerable differences in larval resources among the different pollinator groups, it is uncertain whether the impacts of climate change on bees and syrphid flies will show similar patterns as those for butterflies and bumble bees (Settele et al., 2008, Rasmont et al. 2015a). Carvalheiro et al. (2013) show that bumble bees and butterflies are far more prone to local and regional extinction than other bees or hoverflies, and Kerr et al. (2015) show the drastic effects of climate change on bumble bees. However, in all groups, landscape connectivity, the mobility of species and effects on plants and on floral resources are important and widely unknown factors, which might drastically change the expected future impacts.

2.6.2.4 Further climate change impacts on pollinators

Climate change might modify the balance between honey bees and their environment (including diseases). Le Conte and Navajas (2008) state that the generally observed decline of honey bees is a clear indication for an increasing susceptibility against global change phenomena, with pesticide application, new diseases and other stress (and a combination of these) as the most relevant causes. Honey bees also have shown a large capacity to adjust to a large variety of environments (not at least as they are often managed and hence may be buffered accordingly) and their genetic variability should allow them to also cope with climatic change, which is why the preservation of genetic variability within honey bees is regarded as a central aim to mitigate climate change impacts (Le Conte and Navajas, 2008). Also, due to the great capacity of the honey bee to regulate the temperature inside the colony (hive) by thermogenesis or cooling, this species seems not directly threatened by global warming.

While for the majority of species climate space itself is already limiting (e.g., on the pollinators' physiology), all pollinators that more or less depend on certain plants, potentially suffer indirectly because of climate change impacts on these plants (Schweiger et al., 2008; Schweiger et al., 2010; Schweiger et al., 2012). In butterflies the nectar plants are more independent from the insect in their development (as there is mostly no specific link for the plants' pollination), while one might expect impacts in "tighter" pollination systems. The absence of a pollinator could mean absence of a pollination-dependent plant and vice versa (Biesmeijer et al. 2006). These effects can be expected only for the rare cases of high specialization, and indeed Carvalheiro et al. (2013) reanalyzed the data from Biesmeijer et al. (2006) and found that declines are not parallel in time. Hoover et al. (2012) have shown, in a pumpkin model system, that climate warming, CO_2 enrichment and nitrogen deposition non-additively affect nectar chemistry (among other traits), thereby altering the plant's attractiveness to bumble bees and reducing the longevity of the bumble bee workers. This could not be predicted from isolated studies on individual drivers.

Generally, it can be assumed that climate change results in novel communities, i.e. creation of species assemblages that have not previously co-existed (Schweiger et al., 2010). As these will have experienced a much shorter (or even no) period of coevolution, substantial changes in pollination networks are to be expected (Tylianakis et al., 2008; Schweiger et al., 2012). This might generally result in severe changes in the provision of services (like pollination), especially in more natural or wild conditions (Montoya and Raffaelli, 2010).

Climate change-induced changes in habitat encompass i) shifts in habitat distributions that cannot be followed by species, ii) shifts in distribution of species that drive them outside their preferred habitats and iii) changes in habitat quality (Urban et al., 2012). However, these phenomena are not yet widespread, while models of future shifts in biome and vegetation type (and species distributions, see previous sections) suggest that within the next few decades many species could have been driven out of their preferred habitats due to climate change (Urban et al., 2012). Wiens et al. (2011) also find that climate change may open up new opportunities for protecting species in areas where climate is currently unsuitable. Indeed, in some cases climate change may allow some species to move into areas of lower current or future land use pressure (Bomhard et al., 2005). These and other studies strongly argue for a rethinking of protected areas networks and of the importance of the habitat matrix outside protected areas to enhance the ecosystem connectivity as a key to and guarantee for migration and long-term survival of species (Perfecto et al., 2009).

Over longer periods, habitat types or biomes may shift their distributions due to climate change or disappear entirely (Settele et al., 2014) and climates with no analog in the past can be expected to occur in the future (Wiens et al., 2011). However, because species can show substantial capacity to adapt to novel habitats, the consequences of this non-analogy on species abundance and extinction risk are difficult to quantify (Prugh et al., 2008; Willis and Bhagwat, 2009; Oliver et al., 2009). Effects of climate change on habitat quality are less well studied than shifts in species or habitat distributions. However, several recent studies indicate that climate change may have and probably will alter habitat quality and functions (e.g., Martin and Maron, 2012).

2.6.2.5 Climate change-induced extinctions

Global species extinctions are now at the very upper limits of observed natural rates of extinction in the fossil record (Barnosky et al., 2011) and have mostly been attributed to habitat loss, invasive species or overexploitation throughout the last centuries (Millennium Ecosystem Assessment, 2005). The attribution of extinctions to climate change is much more difficult, but there is a growing consensus that it is the interaction of other global change pressures with climate change that poses the greatest threat to species (Hof et al., 2011; Vanbergen and the Insect Pollinators Initiative, 2013; González-Varo et al., 2013; see also section 2.7).

Estimates of future extinctions are nowadays based on a wide range of methods (incl. the ones described above). Generally, large increases in extinction rates are projected compared to current rates and very large increases compared to the paleontological record (Bellard et al., 2012). Lack of confidence in the models used as well as evidence from the paleontological record led to questioning of forecasts of very high extinction rates due to climate change as being overestimated (Botkin et al., 2007; Willis and Bhagwat, 2009; Dawson et al., 2011; Hof et al., 2011; Bellard et al., 2012). However, as most models did not consider species interactions, potential tipping points in terrestrial ecosystems or future extinction risks may also have been substantially underestimated (Leadley et al., 2010; Bellard et al., 2012; Urban et al., 2012). This is even the case when many pollinators are able to move in response to climate change at the same speed as the plants they depend on, as e.g., the directions of the movements might be different for plants and pollinators concerned (Schweiger et al. 2008, 2012). While there is no scientific consensus concerning the magnitude of direct impact of climate change on extinction risk, there is broad agreement that climate change will contribute to and result in shifts in species abundances and ranges. In the context of other global change pressures this will contribute substantially to increased extinction risks over the coming century (Settele et al., 2014). Also, the results of a very recent analysis by Urban (2015) suggest that extinction risks will accelerate with future global temperatures, threatening up to one in six species under current policies. His study revealed that extinction risks were highest in South America, Australia, and New Zealand, and risks did not vary by taxonomic group (but no differentiation has been made among invertebrates). Studies on the impacts of extreme events (e.g. hurricanes) on pollinators and pollination are rare (but see Rathcke, 2000).

2.6.3 Conclusions

Many plant and pollinator species have moved their ranges, altered their abundance, and shifted their seasonal activities in response to observed climate change over recent decades. They are doing so now in many regions and will very likely continue to do so in response to projected future climate change. The broad patterns of species and biome shifts toward the poles and higher altitudes in response to a warming climate have been observed over the last few decades in some well-studied species groups such as butterflies and can be attributed to observed climatic changes, while knowledge on climate change effects generally is sparse in groups like bats (Kasso and Balakrishnan, 2013) or birds (but see e.g. Abrahamczyk et al., 2011 on hummingbirds).

Under all climate change scenarios for the second half of the 21st century, (i) community composition will change as a result of decreases in the abundances of some species and increases in others, leading to the formation of novel communities; and (ii) the seasonal activity of many species will change differentially, disrupting life cycles and interactions among species. Both composition and seasonal change will alter ecosystem structure and function in many instances, while in other cases the phenology of pollinators (e.g., generalist bee species) will keep pace with shifts in forage-plant flowering.

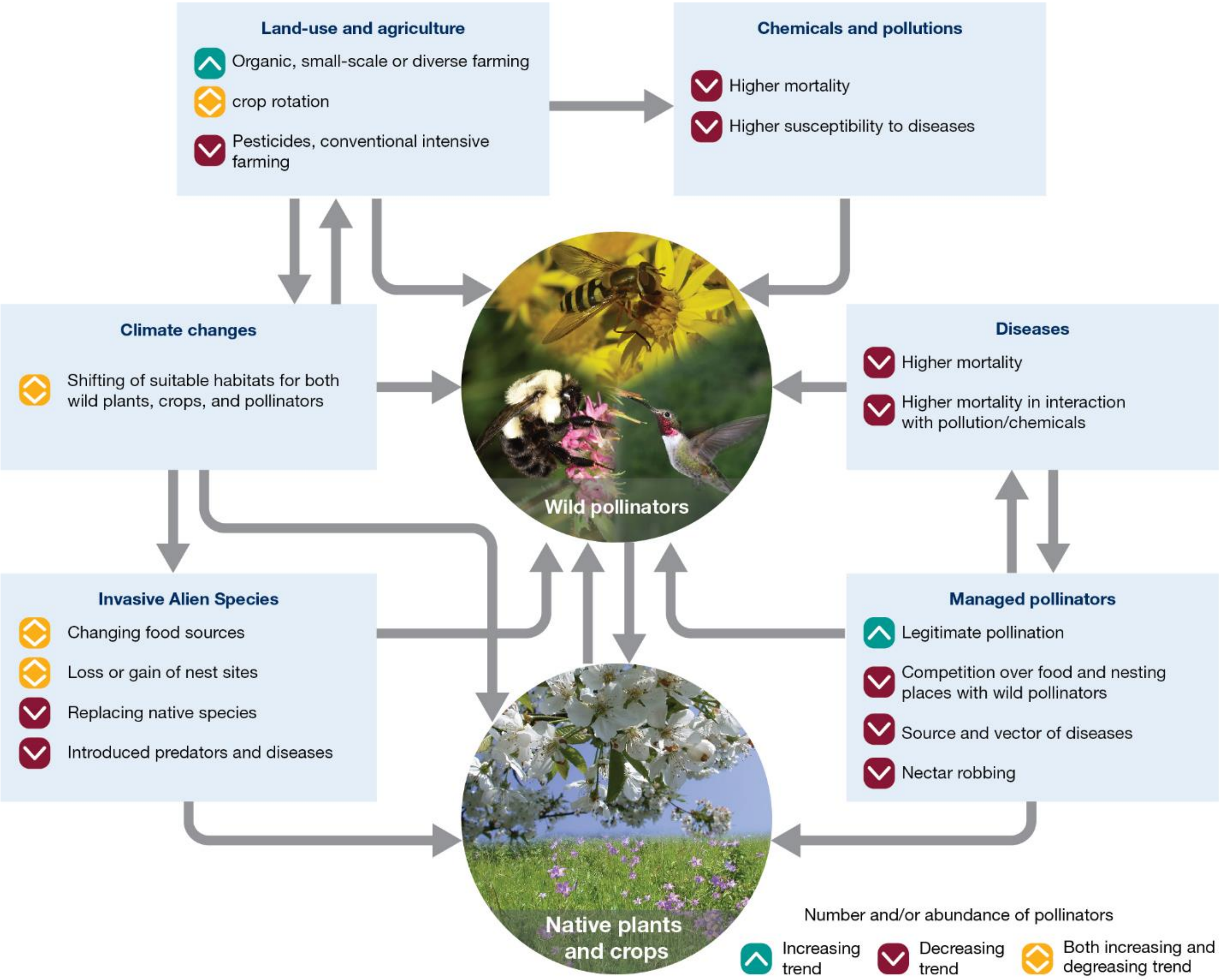
Climate change impacts may not be fully apparent for several decades (Settele et al., 2008; Rasmont et al., 2015a), owing to long response times in ecological systems. In high-altitude and high-latitude ecosystems, climate changes exceeding low end scenarios (e.g., SEDG –see box 2.6.1; or Representative Concentration Pathway 2.6; http://sedac.ipcc-data.org/ddc/ar5_scenario_process/RCPs.html) will lead to major changes in species distributions and ecosystem function, especially in the second half of the 21st century. Honey bees do not appear directly threatened by climate change because of their large capacities of thermoregulation.

For many pollinator species the speed of migration is unknown (including bees, for which foraging ranges are known once the nest is stablished, but not their dispersal ability). For those where more knowledge exists, the rate of movement of the climate across the landscape will exceed the maximum speed at which pollinators can disperse or migrate, especially in order to reach new areas of suitable habitats where climate and other requirements are fulfilled in synchrony. Populations of species that cannot keep up with their climate niche will find themselves in unfavorable climates. Species occupying extensive flat landscapes are particularly vulnerable because they must disperse over longer distances than species in mountainous regions to keep pace with shifting conditions in climates and habitats. Large magnitudes of climate change will particularly affect species with spatially restricted populations, such as boreo-alpine relicts and those confined to small and isolated habitats (e.g., bogs), as they may no longer find suitable habitats, or mountain tops (no upwards move possible), even if the species has the biological capacity to move fast enough to track suitable climates.

A large fraction of pollinator species may face increased extinction risk under projected climate change during the 21st century, especially as climate change interacts with other pressures, such as habitat modification, overexploitation, pollution, and invasive species.

2.7 Multiple, additive or interacting threats

Changes in land use or climate, intensive agricultural management and pesticide use, invasive alien species and pathogens affect pollinator health, abundance, diversity and pollination directly (Sections 2.2-2.6). Moreover, these multiple direct drivers also have the potential to combine, synergistically or additively, in their effects leading to an overall increase in the pressure on pollinators and pollination (González-Varo et al., 2013; Goulson et al., 2015; Vanbergen and the Insect Pollinators Initiative, 2013). These drivers differ in being a physical, chemical or biological threat, in the spatial or temporal scale at which they impact, and in whether they interact simply (additive interactions), or in complex or non-linear ways (e.g., synergistic or antagonistic). For instance, drivers may constitute a chain of events such as when indirect drivers (e.g., increases in economic wealth, changes in consumption) lead to a direct driver (e.g., agricultural intensification) that changes pollinator biodiversity and pollination (Figure 2.7.1). Another possibility is that a direct driver's impact on pollinators and pollination (e.g. climate changes decouple plant and pollinator distributions) might also be manifested through interaction with a second driver (e.g., climate change exacerbates invasive alien species or disease spread) thereby compounding the impact (González-Varo et al., 2013; Ollerton et al., 2014; Potts et al., 2010; Vanbergen and the Insect Pollinators Initiative, 2013). Moreover, certain drivers of change (e.g., conventional agricultural intensification) are themselves a complex combination of multiple, factors (e.g., pesticide exposure, loss of habitat, altered pollen and nectar food resources), which affect pollinators and pollination (Ollerton et al., 2014; Potts et al., 2010; Vanbergen and the Insect Pollinators Initiative, 2013).



17. Figure 2.7.1: Evidence based and potential pathways for single and combined impacts of different pressures on pollinators and pollination.
Arrows indicate pressures resulting in both either decrease or increase of pollinator number and abundance depending on the studied context. The signs in the text boxes indicate increase (+) or decrease (-) of pollinator number and abundance.

This inherent complexity (Figure 2.7.1) means that, to date, this phenomenon of a multifactorial impact on pollinators and pollination has only been demonstrated in comparatively few studies, limited in the scope of species (i.e. honey bees and bumble bees) or combinations of pressures considered (Table 2.7.1). Consequently, the current empirical evidence base is relatively poor due to a relative scarcity of data. It is rarely possible to rule out a single, proximate cause for changes in pollinators and pollination in a particular locality, for a given species or under a certain set of circumstances. However, it seems likely that in the real world a complex interplay of factors is affecting pollinator biodiversity and pollination, although the exact combination of factors will vary in space, time and across pollinator species (Cariveau and Winfree, 2015; Goulson et al., 2015; Vanbergen and the Insect Pollinators Initiative, 2013). Therefore, science and policy need to consider equally the separate and combined impacts of the various drivers affecting pollinators and the pollination they provide. Below we report some case studies together with a Table and a Figure summarising the evidence base and potential pathways for combined impacts of different pressures on pollinators and pollination.

2.7.1 Case study 1: Climate change and land-use

Climate changes may provide opportunities and threats for pollinators, and changes to the composition, extent and configuration of habitat in the landscape are likely to pose a challenge to many pollinator species as climate change progresses (Table 2.7.1). For instance, pollinator species currently at the limits of their climatic range may migrate with global warming into new geographic regions, thus increasing the abundance and diversity of recipient communities (Forister et al., 2010; Giannini et al., 2015; Warren et al., 2001). However, if such immigrants are highly invasive there may be an attendant risk of further ecological changes, for example through alteration of pre-existing plant-pollinator relationships, interspecific competition for food or transfer of pests and diseases (2.4, 2.5). Rates of migration are likely to differ among pollinator and plant species, raising the prospect of a spatial dislocation of plants and their pollinators; recent evidence of climate change impacts on bumble bees suggests there are adaptive limits to the capacity of this pollinator group to track climate change (Kerr et al., 2015; Schweiger et al., 2010). While pollinators with broad diets have the capacity to switch to alternative food plants, thereby maintaining populations and pollinator network structure (Kleijn and Raemakers, 2008; Valdovinos et al., 2013), other evidence suggests specific food-plant diets may underpin pollinator declines (Biesmeijer et al., 2006; Scheper et al., 2014). Habitat loss and fragmentation arising from land-use changes (e.g., habitat conversion to agriculture) may also limit compensatory species migration in the face of climate change (Forister et al., 2010; Giannini et al., 2015; Warren et al., 2001). In general, lower connectivity between habitat remnants combined with future climate shifts may reduce population sizes and increase extinction likelihoods of pollinators (Figure 2.7.1), especially species that are poor dispersers or habitat specialists (Burkle et al., 2013; Giannini et al., 2015; Warren et al., 2001). Land-use driven changes to landscape structure coupled with climate changes might therefore lead to increasingly species-poor plant-pollinator communities dominated by highly mobile, habitat generalist species (Burkle et al., 2013; Hegland et al., 2009).

Furthermore, the delivery of pollination may also be affected by spatial and temporal shifts in pollinator populations or communities. For example, a long-term study of pollinator network structure and pollination delivery to a wild plant species (*Claytonia virginica* L.) revealed that a combination of climate and land-use change reduced pollinator species numbers, affecting network structure and, leading to reduced flower visitation by bee species (Burkle et al., 2013). In addition to affecting species spatial distributions, climate changes may alter the synchrony between pollinator activity and timing of flowering, diminishing or curtailing nectar and pollen food supplies (Memmott et al., 2010; Memmott et al., 2007) (see section on climate change). Therefore, there is potential for climate-driven changes in the availability pollinator foods over time to be exacerbated by the reduced nutritional resources that seem to be a feature of large-scale monocultures (Carvell et al., 2006; Kleijn and Raemakers, 2008).

However, there is likely to be considerable variation in how pollinator populations and communities respond to the combined effects of climate and land-use changes, due not only to variation in life-history traits among pollinator species and guilds, but also the complexity of wider ecological interactions in the ecosystem (Figure 2.7.1). This is illustrated by a plot-scale field experiment that mimicked the combined effects of habitat fragmentation (distance to semi-natural habitat in the landscape) and climate change (manipulation of advanced flower phenology and plant growth) on plant pollination (Parsche et al., 2011). It showed that bee visitation was affected by isolation from other habitat patches, whereas pollinating flies were unaffected by isolation; while advancement of floral phenology did disrupt pollinator mutualisms, this was offset by the plant's escape from herbivore enemies, meaning that net plant reproductive success was unaffected (Parsche et al., 2011).

In summary, there remain relatively few published assessments of the combined effect of land use and climate change on pollinators and pollination (Burkle et al., 2013; Forister et al., 2010; Giannini et al., 2015; Kerr et al., 2015; Parsche et al., 2011; Warren et al., 2001). This is partly because of a lack of suitable data due to the spatial and temporal scales at which these drivers operate, which make inter-correlation likely and their experimental or statistical testing difficult. Nonetheless, our understanding of the separate effects of land-use (see section 2.2) and climate (see section 2.6) changes enables us to predict to a high level of confidence that a combined impact on pollinators is likely in the real world (Table 2.7.1 and Figure 2.7.1). A major source of uncertainty lies in whether such a combined impact lowers the inherent robustness or resilience of pollinator networks (diversity, modularity, etc.) to the point where pollination delivery is affected.

2.7.2 Case study 2: Pathogens and chemicals in the environment

The combined impacts of pathogens and insecticides have implications for the physiological health of individual honey bees and potentially up to the colony scale (Table 2.7.1 and Figure 2.7.1). Laboratory studies have shown increased larval or worker honey bee mortality and energetic stress due to the additive or synergistic interactions between sub-lethal doses of either neonicotinoid or phenylpyrazole insecticides and infection by the microsporidian fungus *Nosema ceranae* or black queen cell virus (BQCV) (Alaux et al., 2010; Aufauvre et al., 2012; Doublet et al., 2014; Retschnig et al., 2014; Vidau et al., 2011). There is some evidence that the synergism between insecticide exposure and *N. ceranae* infection may be contingent on the actual insecticide dose to which the insect is exposed (Retschnig et al., 2014). Less studied are the impacts on pollinators of interactions between insect pathogens and either the miticides used to control invertebrate mites and pests (e.g., *Varroa*) of managed honey bee hives or the herbicides/fungicides applied to crop fields to control weeds and fungal pathogens. One recent study showed the presence of miticides (amitraz and fluvalinate) and fungicides (chlorothalonil and pyraclostrobin) in pollen consumed by honey bees led to a reduction in the individual bee’s capacity to resist *N. ceranae* infection (Pettis et al., 2013). There is some evidence that pesticide exposure impairs the function of the insect immune system, which offers a potential mechanism for combined pesticide-pathogen impacts on bee health (Collison et al., 2015; Goulson et al., 2015).

The synergistic interaction between the neonicotinoid imidacloprid and *N. ceranae* also reduced the activity of an enzyme (glucose oxidase) that is used by worker honey bees to sterilize colony and brood food stores (Alaux et al., 2010). This implies that the effects of pathogen infection and sub-lethal chronic pesticide exposure observed on the individual worker bee’s physiology has the potential to be up-scaled, through worker behaviour, to limit the ability of a bee colony to combat pathogen transmission. Currently, there is limited evidence at the honey bee colony level of disease-pesticide interactions, because only a single study has shown that colony exposure to sub-lethal levels of the neonicotinoid insecticide (imidacloprid) resulted in higher *N. ceranae* infection levels in individual honey bees (Pettis et al., 2012). In a bumble bee species (*B. terrestris*) there is some evidence that chronic exposure to the neonicotinoids thiamethoxam and clothianidin and the trypanosome parasite *C. bombi* in the laboratory can reduce queen survival, a crucial colony trait (Fauser-Misslin et al., 2014). In contrast, the exposure of *B. terrestris* individuals and colonies to laboratory treatments combining *C. bombi* infection and a field-relevant dose of the pyrethroid insecticide λ-cyhalothrin had no effect on worker bee susceptibility to infection or survival (Baron et al., 2014). There remains a need to understand better such combined impacts on social bees at the colony scale.

To date all empirical studies of pathogen-chemical interactions have focused solely on a few eusocial bee species, mainly honey bees. Sociality through the colony hierarchy may to an extent protect reproductive individuals (queens) from exposure to such stressors (Maavara et al., 2007), but see Fauser-Misslin et al., (2014) for an example of bumble bee queen mortality, whilst the impacts on thousands of solitary bee species worldwide remain unstudied.

Differences in experimental design (e.g., doses of chemicals, microorganisms and host tested) and different physiological detoxification pathways among organisms and chemical compounds will tend to lead to alternative outcomes (Collison et al., 2015). For example, most studies to date that have produced some evidence of synergistic pesticide-pathogen interactions have focussed on *Nosema* (Alaux et al., 2010; Aufauvre et al., 2012; Pettis et al., 2012; Pettis et al., 2013; Retschnig et al., 2014; Vidau et al., 2011). The few studies to date on *C. bombi* have produced different results (Baron et al., 2014; Fauser-Misslin et al., 2014) and it remains to be seen whether other pathogens tested in combination with pesticide stressors conform to the general pattern of negative synergistic impacts seen for *Nosema*. Furthermore, most studies have to date been performed under laboratory or semi-field conditions, and the outcome of co-occurring pesticide and disease stress is likely to be further influenced by variations in pesticide dose, and number and combinations of pesticides, actually experienced by pollinators in the field (Collison et al., 2015; Godfray et al., 2014; Retschnig et al., 2014).

In summary, there is some evidence that the interaction between chemicals, especially pesticides, and pathogens may represent a threat to individual bee health and survival. Research is needed to understand disease-pesticide impacts across levels of biological organization (i.e., genetic to colony or population) and combinations of stressors (Table 2.7.1 and Figure 2.7.1), especially in field realistic situations, to obtain insight into their contribution to pollinator losses (Alaux et al., 2010; Collison et al., 2015; Goulson et al., 2015; Pettis et al., 2012; Vanbergen and the Insect Pollinators Initiative, 2013).

2.7.3 Case study 3: Bee nutrition and stress from disease and pesticides

Pollinators such as bees need an optimum balance of nutrients across the individual and colony life-cycle to support their growth and reproduction (Paoli et al., 2014). Global environmental changes (land-use, climate, invasion and pollution) have and continue to result in declines in the diversity and abundance of flowering plants that provide pollinators with pollen and nectar foods (Biesmeijer et al., 2006; Carvalheiro et al., 2013; Carvell et al., 2006; Goulson et al., 2008; Stevens et al., 2006) and with alterations in their composition and quality (Barber and Gorden, 2014; Hladun et al., 2013; Lopezaraiza-Mikel et al., 2007; Stout and Morales, 2009). These changes to pollinator nutritional resources in contemporary landscapes may lead to malnutrition of pollinator individuals and colony stress, which in turn may increase their vulnerability to multiple stressors such as pesticides and pathogens (Archer et al., 2014; Goulson et al., 2015; Vanbergen and the Insect Pollinators Initiative, 2013). Malnutrition in bees is known to affect bee immune function and potentially the function of enzymes used to break-down toxins in diet, so there is thus a risk that this may exacerbate the individual and combined impact of pesticides and pathogens on bees (Goulson et al., 2015; Vanbergen and the Insect Pollinators Initiative, 2013). Immune system activation has a metabolic cost to the individual, and together with exposure to chemicals (section 2.3) and disease (section 2.4), can impair behaviours important in locating floral resources, thereby intensifying the underlying nutritional stress (Goulson et al., 2015; Vanbergen and the Insect Pollinators Initiative, 2013).

2.7.4 Conclusion

Multiple pressures individually impact the health, diversity and abundance of many pollinators across levels of biological organisation spanning genetic to regional scales (Cariveau and Winfree, 2015; González-Varo et al., 2013; Goulson et al., 2015; Potts et al., 2010; Vanbergen and the Insect Pollinators Initiative, 2013).

To date, evidence for a combined impact of different pressures on pollinators and pollination is drawn from relatively few laboratory experiments or correlative field studies that only reflect a small subset of possible scenarios. Doubtless, the precise interactions among different pressures may vary with location, the balance of pressures involved, and among pollinator species according to their different genetics, physiology and ecology (Cariveau and Winfree, 2015; Vanbergen, 2014). Nonetheless it is likely that changes in pollinator biodiversity and pollination are being driven by both the individual and combined effects of multiple anthropogenic factors.

The potential consequences for future food security, human health and natural ecosystem function mean it is crucial that new experiments in field settings (e.g., Hoover et al., 2012) are launched to disentangle the relative effects of different drivers on pollinators and pollination (Cariveau and Winfree, 2015; González-Varo et al., 2013; Potts et al., 2010; Vanbergen and the Insect Pollinators Initiative, 2013). Aside from this important challenge to advance knowledge of the multifactorial pressure on pollinators and pollination, there is an urgent need for decision makers to consider how policy decisions are framed with regard to pollinators and pollination. This may require joint framing across policy and other sectors (e.g., science, business, NGOs) to capture the individual and combined effects of different drivers. The result may lead to more inclusive policy development, taking into account the needs of various stakeholders and advances in science.

2.8 Indirect effects in the context of globalization

Indirect drivers are producing environmental pressures (direct drivers) that alter pollinator biodiversity and pollination. Major indirect drivers relevant to this assessment include the growth in global human population size, economic wealth, globalised trade and commerce, the less stringent environmental regulations in those nations where other markets exist, and technological and other developments, e.g., increases in transport efficiency, or new impacts on land use and food production through climate change adaptation and mitigation (Watson, 2014). These have transformed the climate, land cover and management intensity, ecosystem nutrient balance, and biogeographical distribution of species, and continue to produce consequences for pollinators and pollianation worldwide (2.2-2.7).

Humans now exploit approximately 53% of the Earth's terrestrial surface. For example, croplands are expanding at continental and global scales, with predictions of a net forest loss associated with a 10% increase in the area of agricultural land by 2030, mainly in the developing world. Urban areas are also projected to expand with 66% (vs. 54% today) of the increasing global human population expected to be living in urban areas by 2050 (Ellis et al., 2010; Ellis, 2011; Foley et al., 2005; Foley et al., 2011; Steffen et al., 2011). Increased incomes in emerging economies have driven increased land devoted to pollinator-dependent crops (Monfreda et al., 2008).

International trade is an underlying driver of land-use change, species invasions and biodiversity loss (Hill et al., 2015). The global expansion of industrialised agriculture (e.g., conventional and organic) driven by increased or changing consumption in the developed and emerging economies will continue to drive habitat changes or losses in the developing world, and this can be expected to affect pollinators and pollination. For example, whilst framed around carbon emissions, Persson et al. (2014) showed that much of tropical forests are cleared for export markets. However, direct drivers of change in pollinators and pollination such as land management and landscape structure are also strongly influenced by the local or regional socio-cultural or economic context (Bravo-Monroy et al., 2015). Food sovereignty may offer an alternative direction than ever-increasing trade for feeding the world and reducing negative impacts on ecosystems (Moon, 2011; Billen et al., 2015; Pirkle et al., 2015).

Pesticide regulations, especially in Europe and the US, led to business decisions to shift pesticide sales to alternative markets during the last four decades (Galt, 2008). The less-stringent environmental regulations in those nations where alternative markets occur have the potential to exacerbate local impacts on pollinators (e.g., section 2.3.1.3), yet data are generally lacking, making accurate assessment difficult. Furthermore, pesticides banned in developed nations have, in the recent past, often been used widely on export crops in developing nations, leading to the re-importation of the pesticides into developed nations as a contaminent of the imported food: the so-called "circle of poison" (Galt, 2008). This has been halted on a large scale due to global changes in pesticide regulation, production, trade, sales, and use driven by a number of dynamic economic, social, and ecological processes (Galt, 2008). Nonetheless, countries still differ in their regulation of pest management practices, which creates regulatory asymmetries with unintended economic and environmental consequences (Waterfield and Zilberman, 2012). There is a risk that developing countries may engage in a "race to the bottom"[4] in terms of environmental standards, a socio-economic phenomenon where governments deregulate the business or tax environment to attract or retain economic activity in their jurisdictions (Porter, 1999; Asici, 2013). Furthermore, where national support of programmes to reduce pesticide use has been removed or reduced this has been immediately followed by increased marketing of pesticide products by international and local companies, almost independent of actual need and without consideration of IPM practices (Thorburn, 2015).

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[4] "The race to the bottom is a socio-economic phenomenon in which governments deregulate the business environment or taxes in order to attract or retain economic activity in their jurisdictions", https://en.wikipedia.org/wiki/Race_to_the_bottom

2.9 References

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Chapter 3: The status and trends in pollinators and pollination

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Executive Summary

Many wild bees and butterflies have been declining in abundance, occurrence and diversity at local and regional scales, as it has been recorded in North West Europe and North America (*established but incomplete*). Loss of pollinators has negative impacts on the reproduction of wild plants (*established but incomplete*). In agricultural systems, the local abundance and diversity of wild bees have been found to decline strongly with distance from field margins and remnants of natural and semi natural habitat at scales of a few hundred metres (*well established*) (3.2.2, 3.2.3). At larger spatial scales, declines in bee diversity and shrinkage of geographical ranges, e.g. of bumble bees, have been recorded in highly industrialized regions of the world, particularly Europe and North America, over the last century (*well established*). Beyond records of species-specific population declines, e.g. Franklin's bumble bee (*Bombus franklini*) in Western USA and the giant bumble bee (*Bombus dahlbomii*) in Southern South America, trends are mainly unknown for other regions and continents because of a lack of baseline datasets and monitoring schemes (3.2.3). Declines have been also recorded in other insect pollinator groups, such as butterflies and moths, and among some vertebrate pollinators, particularly hummingbirds and bats (*established but incomplete*) (3.2.1, 3.2.2). Local declines in pollinator abundance and diversity have been linked to decreasing trends in wild plant pollination and seed production in habitat fragments (*well established*), and to declines in the diversity of pollinator-dependent wild plant species at regional scales (*established but incomplete*) (3.6.3).

The International Union for Conservation of Nature (IUCN) Red List assessments indicate that 16.5 per cent of vertebrate pollinators are threatened with global extinction (increasing to 30 per cent for island species). There are no global Red List assessments specifically for insect pollinators. However, regional and national assessments indicate high levels of threat for some bees and butterflies. In Europe, 9% of bees and butterfly species are threatened and populations are declining for 37% of bees and 31% of butterflies (excluding data deficient species, which includes 57% of bees). Where national Red List assessments are available, they show that often more than 40 per cent of bee species may be threatened (3.2.1, 3.2.3).

Large and well-connected plant-pollinator networks, i.e., those with many links between the plants and pollinators, are needed to guarantee satisfactory levels of pollination for wild plants and crops, as well as sufficient availability of food for pollinators (*established but incomplete*). Wild and domestic pollinators involved in crop pollination also require floral resources provided by wild plants (*well established*). Therefore, large, and well-connected plant-pollinator networks are associated with viable populations and diverse communities of pollinators (*established but* *incomplete*). Habitat disturbance tends to lead to loss of interaction links and species from plant-pollinator networks (*established but incomplete*) (3.5.2, 3.5.3).

There is a loss of indigenous and local knowledge and sustainable bee management practices within local communities (*established but incomplete*). Shifts in social systems, cultural values, and accelerated loss of natural habitats have been associated with a decrease in the transfer of knowledge within and between generations. This has led to a decline in stingless bee husbandry in the Americas and Africa and changes in habitat management for wild honey bee species in Asia by local and indigenous communities (*established but incomplete*). Whether this has led to loss of pollination of crops and wild flowering plants remains unknown (3.9.1).

The number of managed western honey bee hives is increasing at the global scale, although seasonal colony loss is high in some European countries and in North America (*well established*). FAO data show that the number of western honey bee hives has increased globally by about 50% over the last five decades, despite a temporary drop during the 1990s after the dissolution of the Soviet Union and Eastern-European Soviet Bloc (*well established*). It is unknown whether this decline is an artefact of how data were collected and reported, or the result of a true decrease in honey bee hives that resulted from the political and economic disruption caused by the Soviet collapse (*unresolved*). FAO data also show that national trends vary widely among countries, with contrasting trends (increases, decreases, no change) found among countries within continents (*well established*). On the other hand, the status of shifts in abundance and distribution of wild honey bees (*Apis mellifera* and other *Apis* species) is largely unknown, with the exception of some records on the spread of the Africanized honey bee in the Americas (*established but incomplete*) (3.3.2).

Trade in and movement of the managed western honey bee, *Apis mellifera*, has led not only to it being a global presence (Antarctica excluded), but also to spillover of pathogens and parasites. Particularly, the shift of the varroa mite (*Varroa destructor),* originally a parasite of the Asian honey bee *Apis cerana*, to the western honey bee has led to severe loss of beehives and makes beekeeping more difficult and costly in many regions (*established but incomplete*). The *Varroa* mite is associated with viruses, such as the deformed wing virus, which is now spreading to bumble bees and wild bees with yet unknown consequences (*unresolved*). The impact of the invasion of honey bees, such as the Africanized honey bee in the Americas, on native bee communities and animal-pollinated plants remains largely unclear (*unresolved*) (3.4.2, 3.4.3).

Commercial management, mass breeding, transport and trade in pollinators outside their original ranges have resulted in new invasions, transmission of pathogens and parasites and regional extinctions of native pollinator species (*well established*). Recently developed commercial rearing of bumble bee species for greenhouse- and field-crop pollination, and their introduction in other continents outside of their original ranges, have resulted in numerous cases of biological invasion, pathogen transmission to native species and the decline of congeneric (sub-)species (*established but incomplete*). A well-documented case is the severe decline in and extirpation from many areas of its original range of the giant bumble bee, *Bombus dahlbomii,* since the introduction and spread of the European *B. terrestris* in southern South America (*well established*) (3.2.3, 3.3.3, 3.4.2, 3.4.3).

More food is produced every year and global agriculture's reliance on pollinator-dependent crops has increased in volume by more than 300 per cent over the last five decades (*well established*). Because the degree of yield dependency on pollinators varies greatly among crops, pollinators are responsible, in a direct way (i.e., the production of seeds and fruits we consume), for a relatively minor fraction (5-8%) of total agricultural production volume (*well established*). However, pollinators are also responsible for many indirect contributions, such as the production of many crop seeds for sowing but not consumption (*well established*). The small fraction of total agricultural production that depends directly on pollinators has increased four-fold during the last five decades, whereas the fraction of food production that does not depend on pollinators has only increased two-fold. Therefore, global agricultural is now twice as pollinator-dependent compared to five decades ago, a trend that has been accelerating since the early '90s (*well established*). Agricultural production, in terms of volume, of some Mediterranean and Middle East countries is particularly dependent on pollinators because of the cultivation of a large variety of temperate and subtropical fruit and seed crops. Rapid expansion of many of these crops in other countries (e.g., China) and cultivation of some genetically-engineered and moderately pollinator-dependent crops, like soybean (e.g., Argentina, Paraguay, Uruguay, and Bolivia) and canola (oilseed rape) (Canada) are responsible for the large increase in the pollinator dependency of global agriculture (*established but incomplete*) (3.7.2, 3.7.3).

Decreased crop yield relates to local declines in pollinator diversity (*well established*), but this trend does not scale up globally (*established but incomplete*). At the local scale, yield of many pollinator-dependent crop species is positively related to wild pollinator diversity. As a consequence, reductions in crop yield have been found in agricultural fields with impoverished bee faunas despite high honey bee abundance (*well established*). While pollination efficiency varies considerably between species and crops, wild bees as a group have been found, on average, to increase crop yield twice as much as honey bees on a per-visit basis (*well established*). A Global analyses of FAO data did not show slowing in yield growth of pollinator-dependent crops relative to pollinator-independent crops over the last five decades (1961-2007) (*established but incomplete*), although the trend in declines of some native bees may change this situation (3.8.2, 3.8.3).

Globally, yield growth and stability are, between 1961-2008, negatively associated with the increasing dependency of crops on biotic pollination (*well established*). Despite no sign of deceleration in average yield growth among pollinator-dependent crops over time, FAO data revealed that yield growth, and yield stability are all negatively related to increasing crop pollinator-dependency (*well established*) (3.8.3).

Cultivation of pollinator-dependent crops largely accounts for the 30% expansion of the global cultivated area occurring during the last fifty years (*well established*). FAO data revealed that crops that largely depend on pollinators have experienced the fastest global expansion in cultivated area (*well established*). However, these crops exhibited the slowest average growth in yield and highest inter-annual yield variability (*well established*) (3.7.3, 3.8.3).

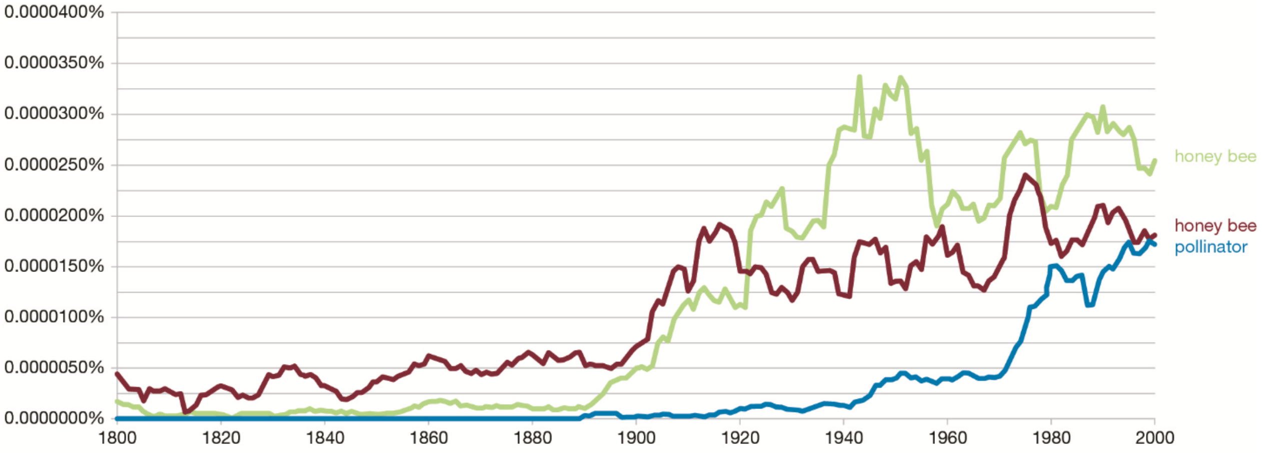
3.1 Introduction

In recent years, the widely-publicized decline of pollinators and its implications for global food security and natural ecosystems has seized popular and scientific attention. Is this widespread concern justified? This chapter presents an overview of the trends in pollinators and pollination worldwide. It addresses the spatial and temporal status and trends in wild pollinators, managed pollinators, introduced invasive pollinators and plants, the structure of pollination networks, wild plant pollination and reproductive success, agricultural pollinator dependence, crop pollination and yields.

This chapter focuses on the following questions: For wild and managed pollinators, is there an on-going, long-term decline? What changes have actually been observed (i.e., decline in abundance, decline in species diversity, or changes in community composition)? What are the consequences of these changes for the reproduction of wild plants and crop yields? Specifically, does the evidence indicate clear spatial or temporal trends in pollinator abundance and diversity, changes in pollinator composition, in mean flower visitation rates and their variability, or in pollination deficits? Moreover, it is important to understand whether and how current trends can be extrapolated to larger scales and new areas, or used as the basis for predictive analyses.

In the process of pollination, there is a clear link between flowering plants and pollinators, both of which diversified in the mid-Cretaceous ca. 120-150 million years ago (Hu et al., 2008). This means that there is a long evolutionary history for the ecosystem function of pollination, which may even predate the flowering plants (Ollerton and Coulthard, 2009). The first historical observation of pollinators being important, namely affecting crop yield, includes depictions in ancient art (Buchmann and Nabhan, 1997). Numerous other examples have since become known, and can be found referenced in the subsequent sections of this chapter and in previous chapters. Without a doubt pollinators are an essential component of biodiversity, yet relatively few studies address the threat of (local) extinctions. Most emphasis is on the characterisation of the interactions between plants and pollinators (Bond, 1994). Some examples from island systems are exceptions in this regard, where disruptions of animal-plant interactions have been more readily quantified (Cox and Elmqvist, 2000) and there are several documented pollinator extinctions (Ollerton et al., 2014; Cox and Elmqvist, 2000; Fleischer et al., 2008). Recently, examples document regional declines and local extinctions of native pollinators as a consequence of the international commercial traffic in bees and plants (Stout and Morales, 2009).

Over the past 200 years, attention from academic researchers, and to some degree the general public, has shifted from (managed) honey bees to pollinators in general, with a steep increase starting in the 1970s (Figure 3.1).



18. Figure 3.1. Temporal trend in the use of the terms ‘pollinator’, ‘honeybee’ and ‘honey bee’ generated using Google Ngram.
This shows the trend in the currently databased collection of Google Books between the years 1800 and 2008 (percentage of all books published in the USA in English that contain the designated term). Note the recent (post 2000) switch to the term ‘pollinator’ appearing more frequently in publications than ‘honey bee’. This may be attributed to the increased number of publications that are focused on the breadth and diversity of pollinators as providers of an essential ecosystem function.

Concern about pollinator decline is relatively recent (Kevan, 1999; Raw, 2001; Spira, 2001; Committee on the Status of Pollinators in North America, 2007; Williams, 1982), but there is a growing perception among both scientists and the general public that at least some populations and species are declining in at least some areas. Much of this concern comes from well-documented declines in managed honey bee (*A. mellifera*) populations in North America and Europe, as well as more recent reports of declines and even local or global extinctions of some native bees, such as bumble bees (*Bombus* species) (Bommarco et al., 2012; Bartomeus et al., 2013; Williams et al., 2009). However, the number of managed colonies of *A. mellifera*, the major commercial pollinator worldwide, has increased over the past 50 years (Aizen and Harder, 2009a). Likewise, the diversity of additional native bee species nowadays managed for pollination (e.g., *Osmia*, *Megachile*, *Anthophora*, *Bombus*) has increased, partly because of their greater efficiency compared to honey bees in pollinating specific crops (e.g., Freitas and Pereira, 2004). The fact that almost half the studies on pollinator decline comes from only five countries (Australia, Brazil, Germany, Spain and the USA), with only 4% of the data from the continent of Africa (Archer et al., 2014), highlights the bias in information and the lack of data for some regions. Even among studies that address pollination as an ecosystem function or service, there is substantial variation in how this is measured, and therefore it is difficult to compare studies and derive management recommendations (Liss et al., 2013). The scale of sampling that would be required to provide an answer to whether pollinator populations are declining in a specific region has been estimated at around 200–250 sampling locations, each sampled twice over five years to provide sufficient statistical power to detect small (2–5%) annual declines in the number of species and in total abundance, and would cost US$2,000,000 (LeBuhn et al., 2013). These conclusions were drawn from analysis of studies that used seven different sampling techniques (pan traps, Moericke traps, visual counts of the number of animals, malaise traps, hand netting, funnel traps, and baits) in relatively small study sites.

In addition to concern about individual species, there is increased concern about the effects of pollinator decline on plant communities (Lever et al., 2014). A recent study shows that loss of a single pollinator species can reduce floral fidelity in the remaining pollinators, “with significant implications for ecosystem functioning in terms of reduced plant reproduction, even when potentially effective pollinators remained in the system” (Brosi and Briggs, 2013). Below we provide detailed summaries of the state of the science in each of the above-mentioned areas.

3.2 Trends in wild pollinators

3.2.1 Outline of section

Wild pollinators are a diverse group, and include vertebrate species such as birds, mammals, and reptiles, and invertebrates such as bees, butterflies, flies, moths, beetles, ants, and wasps. This very diverse group includes a few species that are very well known, such as the European honey bee (*Apis mellifera*), some bumble bees (*Bombus* spp.), butterflies, and hummingbirds, but for the vast majority of pollinators there are tremendous knowledge gaps about their life histories, distribution, and abundance that hinder our analysis of trends.

The regulation of animal populations in the wild has been the object of research by ecologists and conservation biologists for many years, but the application of these ideas to non-pest insect species such as pollinators is relatively recent. For example, it was not until 1912 that Sladen (1912) published a treatise on bumble bees, and only in the 1950s did studies begin to appear about their colony dynamics and foraging behavior (Free and Butler, 1959). In 1963 the first study was published about bumble bee diseases (Skou, 1963), and in that same decade studies about the ability of bumble bees to increase pollination of clover and alfalfa appeared (Free, 1965; Holm, 1966; Bohart, 1957), as did a paper about how to rear bumble bees in captivity (Plowright and Jay, 1966), opening up the possibility of managing their populations. Heinrich published a monograph about bumble bee ecology and foraging energetics in 1979.

Only recently, a study assessed the limited evidence of how food resources and risks regulate wild bee populations (Roulston and Goodell, 2011). For vertebrate pollinators, and even more so for most insect species, there are few studies investigating the environmental factors, and biotic interactions such as competition, predation, parasitism, and disease that influence their populations. Among bird pollinators, information about ecological interactions is available for hummingbirds (Trochilidae: Gill, 1988; Sandlin, 2000; Tiebout, 1993; Fleming, 2005), sunbirds (Nectariniidae: Carstensen, 2011), and honeyeaters (Meliphagidae: Craig et al., 1981; McFarland, 1996; Paton, 1985; Pyke et al., 1996), and a little for lorikeets (Loriinae: Richardson, 1990). Some information is also available for bat pollinators (Chiroptera: Winter and von Helversen, 2001; Fleming et al., 2005). More generally, the insights that ecologists have gained for regulation of animal populations in general can also shed light on pollinator populations (e.g. Knape and de Valpine, 2011).

The changes in pollinator populations described in sections 3.2.2 (distribution) and 3.2.3 (abundance), and future challenges they face, are in part the consequences of the changing climate and changing landscapes. Pollinator responses to the changing climate are likely to include changes in their latitudinal and altitudinal distributions, producing changes in species occurrence and hence diversity at any particular locality. Evidence of such shifts and their consequences is beginning to accumulate, with declines recorded for both managed and wild bee populations in both Europe and North America (Becher, 2013), altitudinal and latitudinal range changes for butterflies (Heikkinen et al., 2010; Casner et al., 2014), and altitudinal shifts for bumble bees (Ploquin et al., 2013; Pyke et al., 2016).

The standard objective assessment of the status of a species, e.g. a pollinator, is the IUCN Red List assessment. Global assessments are available for many vertebrate pollinators, e.g. birds and bats. Most insect pollinators have not been assessed at a global level. In total 16.5% of vertebrate pollinators are threatened with global extinction (increasing to 30% for island species; Aslan et al. 2013). The trend is generally towards more extinctions. Regional and national assessments of insect pollinators indicate high levels of threat particularly for bees and butterflies (often >40% of species threatened) (IUCN Red List for Europe; www.iucn.org; van Swaay et al. 2010). The recent European-scale red lists revealed that 9% of bees and 9% of butterflies are threatened and populations are declining for 37% of bees and 31% of butterflies (excluding data-deficient species). Note, however, that for the majority of European bees data are insufficient to make IUCN assessments. Many if not most of the data-deficient species are likely to have a very limited (endemic) distribution or are very rare, traits often found in threatened species. At national levels numbers of threatened species tend to be much higher than at regional levels, e.g., more than 50% for bees in some European countries. In contrast, crop-pollinating bees are generally common species and rarely threatened species. Of 130 common crop-pollinating bees (Kleijn et al., 2015) only 58 species have been assessed either in Europe or North America. Only two species are threatened (*Bombus affinis, Bombus terricola),* two are near threatened (*Andrena ovatula*, *Lasioglossum xanthopus)*, 42 are doing well (all assessed as Least Concern), whereas for 12 of these species data are insufficient for assessment. Of 57 species mentioned as crop pollinators in Klein et al. (2007) only 10 species have been formally assessed, of which one bumble bee species, *Bombus affinis,* is critically endangered. However, at least 10 other species, including three honey bee species, are known to be very common.

Human-altered landscapes can reduce gene flow in pollinator populations (Jha, 2015), and the interaction between land use and fragmentation (Hadley and Betts, 2012) can also have negative impacts (Kenefic et al., 2014). Land use intensity has also been shown to correlate with pollinator populations (Clough et al., 2014). A recent paper has reviewed the effects of local and landscape effects on pollinators in agroecosystems (Kennedy et al., 2013); bee abundance and richness were higher in diversified and organic fields (e.g., Holzschuh et al., 2007) and in landscapes comprising more high-quality habitats, while bee richness on conventional fields with low diversity benefited most from high-quality surrounding land cover (e.g., Klein et al., 2012). Stresses from pesticides and parasites (Chapter 2) can also alter pollinator distributions and abundance. Increases in nitrogen inputs can also affect flower production, pollinator visitation, and fruit set (Muñoz et al., 2005).

3.2.2 Evidence for spatial shifts and temporal changes in species occurrence

Information about wild pollinator populations is primarily available from two sources, either historical information from museum collections and records collected by amateur naturalists and scientists, or very recent surveys initiated in response to concerns about current declines that can now provide baseline information for future comparison. For example, Biesmeijer et al. (2006) compiled almost 1 million records for bee and hoverfly observations for Britain and the Netherlands from national entomological databases to compare areas with extensive sets of observations before and after 1980. They found significant declines in the bee species richness in many areas, and also that outcrossing plant species that are reliant on insect pollinators (United Kingdom) or bee pollinators (Netherlands) also declined relative to species with wind- or water-mediated pollination. These results strongly suggest, but do not prove, a causal connection between local extinctions of functionally-linked plant and pollinator species. Another example of how museum records can be used to gain insights is a re-survey of bee fauna and associated flora from a grassland site in Brazil, originally surveyed 40 years ago and again 20 years ago, which found that bee species richness has declined by 22% (Martins et al., 2013). Some previously abundant species had disappeared, a trend that was more accentuated for large rather than small bees. However, one study found that the abundance of common bee species was more closely linked to pollination than bee diversity (Winfree et al., 2015).

A recent long-term study of relative rates of change for an entire regional bee fauna in the northeastern United States, based on >30,000 museum records representing 438 species (Bartomeus et al., 2013), found that over a 140-year period native species richness decreased slightly, but declines in richness were significant ($p = 0.01$) only for the genus *Bombus*. “Of 187 native species analyzed individually, only three declined steeply [in abundance], all of these in the genus *Bombus*. However, there were large shifts in community composition, as indicated by 56% of species showing significant changes in relative abundance over time.” At the community level some of the decline was masked by the increase in exotic species (increased by a factor of 9, to a total of 20, including species of *Anthidium*, *Hylaeus*, *Lasioglossum*, *Megachile*, *Osmia*, etc.), with an accompanying trend toward homogenization. The study also provided insights into the traits associated with a declining relative abundance: small dietary and phenological breadth and large body size, which may provide clues to identify which species are likely to be susceptible to declines in other areas as well. It is somewhat reassuring that, despite marked increases in human population density in the northeastern USA and large changes in anthropogenic land use in that area, Bartomeus et al. (2013) found that aggregate native species richness declines were modest outside of the genus *Bombus*; the number of rarefied non-*Bombus* bee species per time period has declined by 15%, but the trend is not statistically significant ($p = 0.07$).

A third example of re-sampling of bees, in Colorado, USA, used a century-old record of bee fauna that had found 116 species in grassland habitats (Kearns and Oliveras, 2009a). The re-sampling, a five-year effort, recorded 110 species, two genera of which were not present in the original 1907 collection. Their comparison was hampered by the lack of information about the sampling techniques of the original study, and taxonomic changes, but the authors concluded that the conservation of most of the original species had been facilitated by the large amount of preserved habitat in the study area (Kearns and Oliveras, 2009b). An even longer re-sampling period of 120 years in Illinois, in temperate forest understory, found a degradation of interaction network structure and function, with extirpation of 50% of the original bee species (Burkle et al., 2013). The authors attributed much of this loss to shifts in both plant and bee phenologies that resulted in temporal mismatches, nonrandom species extinctions, and loss of spatial co-occurrences between species in the highly modified landscape. Thus negative changes in the degree and quality of pollination seem to be ameliorated by habitat conservation.

Examination of museum specimens has also been shown to provide insights into reasons for bee population declines. Pollen analysis from 57 generalist bee species caught before 1950 showed that loss of preferred host plants was strongly related to bee declines, with large-bodied bees (which require more pollen) showing greater declines than small bees (Scheper et al., 2014).

In a meta-analysis of long-term observations across Europe and North America over 110 years, Kerr et al. (2015) looked for climate change–related range shifts in bumble bee species across the full extents of their historic latitudinal and thermal limits, and changes along elevation gradients. They found consistent trends from both continents with bumble bees failing to track warming through time at their northern range limits, range losses from southern range limits, and shifts to higher elevations among southern species. These effects were not associated with changing land uses or pesticide applications.

A monitoring program for butterflies in the Flanders region of Belgium (Maes and Van Dyck, 2001) provides evidence for that region having the highest number of butterfly extinctions in Europe, with 19 of the original 64 indigenous species having gone extinct. Half of the remaining species are now threatened with extinction. The authors attribute these losses to more intensive agricultural practices and the expansion of building and road construction (urbanization), which increased the extinction rate more than eight-fold in the second half of the 20th century.

In the absence of population trend data, studies of species diversity can also provide some information about the status of pollinators. Studies such as those of Keil et al. (2011) for Syrphidae, and another study of species of bees, hoverflies (Syrphidae) and butterflies (Carvalheiro et al., 2013) are examples of this. Carvalheiro et al. (2013) looked at these three groups of pollinators in Great Britain, Netherlands, and Belgium for four consecutive 20-year periods (1930-2009). They found evidence of extensive species richness loss and biotic homogenization before 1990, but those negative trends became substantially less accentuated during recent decades, even being partially reversed for some taxa (e.g., bees in Great Britain and Netherlands). They attributed these recoveries to the cessation of large-scale land-use intensification and natural habitat loss in the past few decades. Most vulnerable species had been lost by the 1980s from the bee communities in the intensively farmed northwestern European agricultural landscapes, with only the most robust species remaining (Becher, 2013; Heikkinen et al., 2010, Casner et al., 2014; Holzschuh, 2008). New species are continuously colonizing north-western Europe from the much richer Central and South European regions. This may also contribute to increases of insect pollinator richness. Bartomeus et al. (2013) found that bee species with lower latitudinal range boundaries were increasing in relative abundance in the northeastern USA, and Pyke et al. (2016) compared altitudinal distributions of bumble bees in the Colorado Rocky Mountains from 1973 and 2007 and found that queens had moved up in altitude by an average of 80m. Also, uphill shifts in bumble bee altitudinal distributions have been recorded in the Cantabrian Cordillera of northern Spain during the last 20 years leading to local extinctions and bee fauna homogenization where previously there were distinct community differences (Ploquin et al., 2013).

Temperature increases can directly affect bee metabolism but there have also been significant temperature-related changes in the phenology of floral resources important for pollinators, including earlier flowering of most species, and changes in the seasonal availability of flowers that may also affect pollinator survivorship (Aldridge et al., 2011). Forrest (2015) reviewed research on plant–pollinator mismatches, and concluded that although certain pairs of interacting species are showing independent shifts in phenology (a mismatch), only in a few cases have these independent shifts been shown to affect population vital rates (seed production by plants) but this largely reflects a lack of research. Bartomeus et al. (2011) combined 46 years of data on apple flowering phenology with historical records of bee pollinators over the same period, and found that for the key pollinators there was extensive synchrony between bee activity and apple peak bloom due to complementarity between the bees' activity periods. Differential sensitivity to temperature between plants and their pollinators can also affect butterflies; flowering time of butterfly nectar food plants is more sensitive to temperature than the timing of butterfly adult flight (Kharouba and Vellend, 2015).

Bedford et al. (2012) focused on evidence for geographical range shifts among butterflies in Canada. They collected data for 81 species and measured their latitudinal displacement between 1960 and 1975 (a period prior to contemporary climate change) and from 1990 – 2005 (a period of large climate change). They identified an unexpected trend, given the mobility of butterflies, for species' northern borders to shift progressively less relative to increasing minimum winter temperatures, suggesting that even these mobile pollinators have been unable to extend their ranges as quickly as would be required to keep pace with climate change; this might be because of their dependence on larval host plants, which may not be shifting quickly either (Bedford et al., 2012).

A similar study of 48 butterfly species in Finland found that they shifted their range margins northward on average by 59.9km between 1992-1996 and 2000-2004, with non-threatened species showing a larger change than the more stationary threatened species (Pöyry, 2009). Such poleward shifts (Parmesan, 1999) are probably a common feature of many pollinator species geographical distributions in recent years (although not much is known about southern hemisphere species), and are likely being matched by altitudinal shifts as well, as seen for both butterflies and bumble bees

(Forister et al., 2011; Pyke et al., 2012; Wilson et al., 2007). However, Kerr et al. (2015) found in a survey of historical data for bumble bee distributions in both Europe and North America that there were consistent trends in failures to track warming through time at species' northern range limits, range losses from southern range limits, and shifts to higher elevations among southern species. So this important group of pollinators is being affected negatively by this response to climate change. Responses to climate change are also compounded by changes in habitat. For example, Warren et al. (2001) found that 75% of 46 butterfly species expected to be expanding their range north are declining in abundance, and attributed this to negative responses to habitat loss that have outweighed positive responses to climate warming. Adverse effects of nitrogen deposition on butterfly host plants may also be taking a toll on that group of pollinators (Feest et al., 2014).

The changing climate may also pose challenges for avian pollinators. One study of the potential changes in distribution that will result considered South Africa, where some of the migratory pollinator species may be at particular risk (Simmons et al., 2004; Huntley and Barnard, 2012), and a study of hummingbird migration in North America found that if phenological shifts continue at current rates, hummingbirds will eventually arrive at northern breeding grounds after flowering begins, which could reduce their nesting success (McKinney et al., 2012).

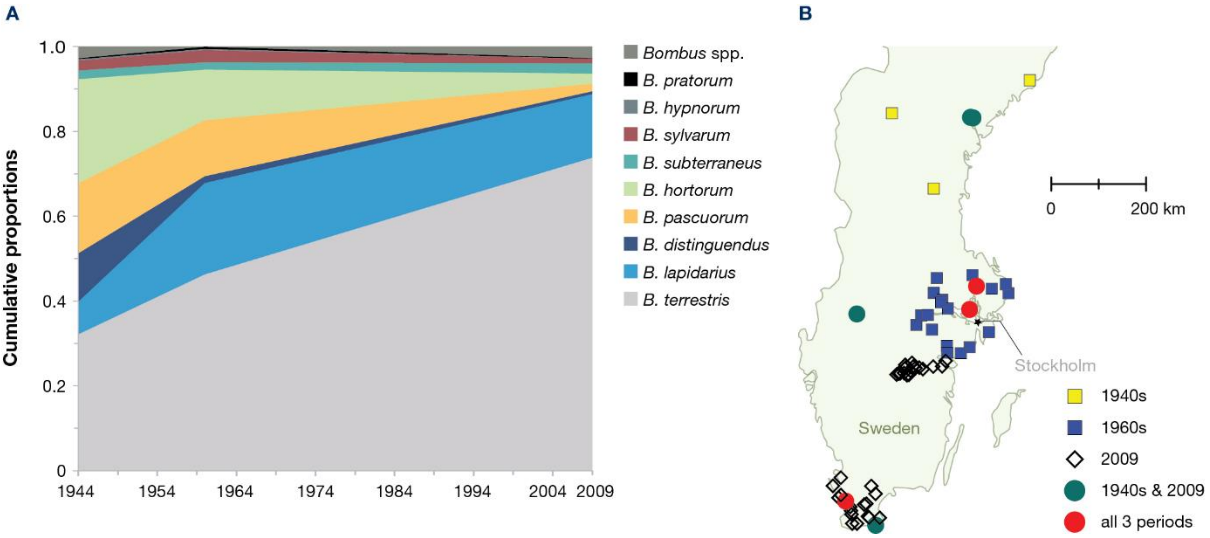
3.2.3 Shifts in pollinator abundance

All animal populations fluctuate in abundance and pollinator populations are no exception. That said, there is evidence that some pollinator populations are now changing in abundance to such a degree that they have exceeded the range of variation previously recorded (Cameron et al., 2011); a few have suffered local or even global extinctions (Cox and Elmqvist, 2000; Maes and Van Dyck, 2001; Grixti et al., 2008; Mortensen et al., 2008; Ollerton et al., 2014). Although there is some evidence for changes (see references cited in previous section), this is a topic for which much additional work is needed before we have a clear picture for trends on a global scale.

Insect populations are notoriously variable in abundance (Andrewartha, 1954), and with few exceptions we do not fully understand the underlying causes for this variation in insect pollinator populations. Despite our ignorance of the exact causes of variation in most pollinator populations, we do know that diseases (Colla et al., 2006; Koch and Strange, 2012; Fürst et al., 2014; Manley, 2015), parasites (Antonovics and Edwards, 2011; Arbetman et al., 2013), pesticides (Gill et al., 2012; Stokstad, 2013; Johansen, 1977; Canada, 1981), a lack of diverse food sources (Alaux et al., 2010), and habitat loss (not always separated from fragmentation; Hadley and Betts, 2012) (Schüepp et al., 2014), which reduces both nest sites and floral resources (Kearns et al., 1998), can all potentially affect pollinators negatively, including species of particular concern for crop pollination (Stephen, 1955). (See Chapter 2 for additional information.)

Bumble bees (Hymenoptera)

Very few studies assess shifts in pollinator abundance, mainly because historic population counts are not available. A remarkable exception is that of clover pollination by bumble bees in Scandinavian countries (Bommarco et al., 2012; Dupont et al., 2011). Drastic decreases in bumble bee community evenness (relative abundance of species), with potential consequences for the level and stability of red clover (*Trifolium pratense*) seed yield, were observed in Swedish clover fields over the last 90 years (Bommarco et al., 2012; Figure 3.2.). Two short-tongued bumble bees (*Bombus terrestris* and *Bombus lapidarius*) increased in relative abundance from 40 to 89 per cent and now dominate the communities. Several long-tongued bumble bees declined strikingly over the same period. The mean number of bumble bees collected per field was typically an order of magnitude higher in the 1940s and 1960s compared with the most recent data. Associated with this, average clover seed yield declined, while yield variability doubled. The authors infer that the current dependence on few species for pollination has been especially detrimental for the stability of seed yield. In parallel to this, bumble bee abundances and species composition have shifted in Danish red clover (*Trifolium pratense*) fields as well (comparing the 1930s with present data; Dupont et al., 2011). Abundance (bees observed per m^2) of short-tongued bumble bees did not change significantly. Long-tongued bumble bee species, however, showed consistent and large declines in species richness and abundance throughout the flowering season. Of 12 *Bombus* species observed in the 1930s, five species were not observed in 2008-2009. The latter were all long-tongued, late-emerging species (Dupont, 2011).



19. Figure 3.2. Map of visited sites and detected proportional shifts in bumble-bee community composition in red clover seed fields in the last 70 years.
Blue circles, all three periods; green circles, 1940s and present; blue squares, 1940s; yellow squares, 1960s; green squares, present. Proportion of bumble-bee abundance for the different species is presented as cumulative proportions for the communities averaged among sites and years within each period.

Bommarco et al. (2010) found that the effects of habitat loss on wild bee populations in Sweden, Germany, and Finland were dependent on dispersal capacity and diet breadth. Small generalist bees tended to be more strongly affected by habitat loss as compared with small specialists, and social bees were negatively affected by habitat loss irrespective of body size. Habitat loss thus led to clear shifts in the species composition of wild bee communities. It seems likely that this effect of habitat loss would be found in other pollinator communities.

A survey of bumble bee populations in North America found that relative abundances of four species have declined by up to 96% and that their geographic ranges have contracted by 23–87%, some within the last 20 years (Cameron et al., 2011). Colla and Ratti (2010) studied the abundance of *Bombus occidentalis* in blueberry fields in western Canada and found that abundance of that species had declined from 27 and 22% in 1981-82 to 1% in 2003-04, indicating that at least that species had become much rarer. Quantitative data are also available for transects surveyed in the Colorado Rocky Mountains in 1974, and again in 2007 (Pyke et al., 2011; Pyke et al., 2016).

In the case of bumble bees, the development of a commercial international trade has led to the phenomenon of "pathogen spillover", whereby introduced colonies infected with disease-causing parasites (e.g., *Crithidia bombi*, *Nosema bombi*) have spread those diseases to wild populations in North America (Colla et al., 2006; Otterstatter and Thomson, 2008). A 2012 review of the status of North American bumble bees (Schweitzer et al., 2012) suggested that pathogen spillover might be a primary factor in the decline of eight species from three subgenera that have declined drastically during the last 15 to 20 years. These include three species that are obligate parasites on other declining species.

Szabo et al. (2012) found weakly significant correlations between losses of *B. terricola* and *B. pensylvanicus* and vegetable greenhouse density in some native populations ($R^2 = 0.17$, $P = 0.0048$ for *B. terricola*; and $R^2 = 0.08$, $P = 0.0034$ for *B. pensylvanicus*), including local extinctions. Importation of disease-carrying bumble bees has also been documented recently in South America (Arbetman, 2013; Schmid-Hempel et al., 2014) and elsewhere (Graystock et al., 2013). Declining populations in North America have significantly higher infection levels of the microsporidian pathogen *N. bombi* and lower genetic diversity compared with co-occurring populations of the stable, non-declining, species (Cameron et al., 2011).

A study of bumble bees in Illinois, using museum data, found that bumble bee species richness declined substantially during the middle of the 20th century (1940–1960), with local extinctions of four species, and range contractions in four other species (Grixti et al., 2008). The authors concluded that half of the bumble bee species found historically in Illinois have been locally extirpated or have suffered declines, supporting observations of broader declines in North America. These declines coincided with large-scale agricultural intensification between 1940 and 1960, which would have reduced nesting habitat and floral resources, and increased exposure to pesticides. In Europe, 24% of bumble bee species are threatened with extinction, according to a recent IUCN study assessing the species (Nieto et al., 2014). Eight species are listed as Vulnerable, seven as Endangered, and one as Critically Endangered. Bumble bees are of concern in part because they are most abundant and diverse in colder (high altitude, high latitude) climates that are very susceptible to climate change (Williams and Osborne, 2009; Rasmont et al., 2015). Well- documented cases of species-specific bumble bee declines are Franklin's bumble bee (*Bombus franklini*) in Western USA (Thorp, 2005) and the giant bumble bee (*Bombus dahlbomii*) in Southern South America, the latter clearly linked to the introduction of European *B. ruderatus* and *B. terrestris* (Morales et al., 2013).

A meta-analysis of studies on bumble bees in Britain, Canada, and China found that decline susceptibility is generally greater for species that have greater climatic specialization, for species in areas where they occur closest to the edges of their climatic ranges, and for species that have queens that become active later in the year (Williams et al., 2009). The later initiation of colonies may become a problem if there is a mid-season decline in resource abundance, as was found recently in a montane study site in Colorado (Aldridge et al., 2011). Some European bumble bees have also been found to be declining in abundance over the past 60 years (Goulson et al., 2008), with these changes driven primarily by habitat loss and declines in floral abundance and diversity resulting from agricultural intensification. The declines in bees in Brazil reported by Martins et al. (2013) were also attributed in part to habitat change (urbanization). Competition for floral and nest site resources can negatively affect bumble bee diversity in urban areas (McFrederick and LeBuhn, 2006).

Box 3.1 – Bumble bee monitoring programs

BeeWalk is a UK national recording scheme run by the Bumble Bee Conservation Trust to monitor the abundance of bumble bees on transects across the country. These transects are conducted by volunteers, who identify and count the bumble bees they see on a monthly walk along a set route from March to October. http://beewalk.org.uk/

The Irish Pollinator Initiative http://www.biodiversityireland.ie/projects/irish-pollinator-initiative/ offers a similar opportunity for monitoring bumble bees and other pollinators.

Canadian collaborators with the Grey Bruce Centre for Agroecology facilitate pollinator monitoring in Ontario. http://gbcae.com/pollinators.html

The Xerces Society has organized a citizen science effort, Bumble Bee Watch, to monitor these pollinators in USA and Canada. http://bumblebeewatch.org/contents/about/

Wasps (Hymenoptera)

Wasps are not common pollinators for very many plant species, but are involved in some interesting sexual deception pollination systems of orchids (e.g., Peakall and Beattie, 1996, Schiestl et al., 2003). They are perhaps best known as obligate specialist pollinators of figs (*Ficus spp.*), which produce fruits that are important resources for many herbivores (Herre et al., 2008). The susceptibility of the wasps to changes in flowering patterns induced by drought was documented in northern Borneo, when an El Niño Southern Oscillation (ENSO) event led to the local extinction of the pollinators because of a gap in the availability of flowers (Harrison, 2000). In general, however, almost nothing is known about the size and variability of pollinating wasp populations.

Flies (Diptera)

Flies can be efficient pollinators of both wild and crop plants (Jauker and Wolters, 2008; Howlett et al., 2009; Rader et al., 2009; Howlett, 2012; Jauker, 2012), and a great diversity of them have been recorded as flower visitors (Larson et al., 2001; Speight, 2010; Woodcock et al., 2014). Some crops, such as onions, that are not visited well by bees can be pollinated by flies (Currah, 1984), and they can be used in greenhouses for sweet pepper pollination (Jarlan et al., 1997). Species from at least 86 families of Diptera have been observed visiting flowers, and over eleven hundred species of plants from 172 families have been reported as being visited by flies (Kearns, 2001, 2002; Inouye et al., 2015). One species is even available commercially for pollination; *Lucilia sericata* (common green bottle fly) are available as "Natufly". Flies are particularly important at high latitudes (Totland, 1993; Woodcock et al., 2014) and high altitudes, especially in areas where bumble bees are not present, such as alpine Australia (Inouye and Pyke, 1988).

Despite their obvious importance, there are very few data available on population sizes and trends. Keil et al. (2011) looked for temporal change in species richness of hoverflies (Syrphidae) from the UK and the Netherlands, comparing museum specimen data prior to and post 1980. They were particularly interested in the effects of spatial scale, and compared grid resolutions from 10 x10 km to 160x160 km. Trends differed across spatial scales, but species richness increased in the Netherlands and decreased in the UK at the fine scale (10 x 10km), while trends differed between countries at the coarsest scale (positive in UK, no change in Netherlands). Thus Keil et al. (2011) concluded that explicit considerations of spatial (and temporal) scale are essential in studies documenting past biodiversity change or attempting to forecast future changes.

The Delhi Sands Flower-Loving Fly (*Rhaphiomidas terminatus abdominalis*) is one of the most endangered animals on the planet and was listed under the US Endangered Species Act in 1993 (http://www.xerces.org/delhi-sands-flower-loving-fly/). It is a habitat specialist, and the dunes where it occurs in southern California have largely disappeared due to development. The flies collect nectar of at least one plant in that habitat (*Eriogonum fasiculatum*) (U.S. Fish and Wildlife Service, 1997).

Iler et al. (2013) analysed a 20-year record of Syrphidae from a Malaise trap maintained at the Rocky Mountain Biological Laboratory (2,900 m altitude in Colorado, USA). Their primary focus was on phenology of fly emergence (several species), for which they found no significant trend, suggesting that the historic interaction with food plants is being maintained in the face of climate change. A 15-yr study of Syrphidae in the UK (Owen, 1989) found that hoverfly populations are more stable than those of other terrestrial arthropods, that there are strong correlations ($r=0.51 - 0.54$, $p<.0001$) between abundance and distribution, and that abundance is not affected by body size. However, they found that larger species tend to have more stable populations. Many species in this family have very specialized habitat requirements for the larval stage (Rotheray, 2011), which may put them at risk from habitat alteration.

Butterflies and moths (Lepidoptera)

Butterflies are not as important as bees as pollinators in many ecosystems, because they often visit flowers less frequently than do bees and may also deposit less pollen per visit (Winfree et al. 2011), and their abundance and pollen loads (Proctor et al., 1996) are often lower. However, they can deposit high-quality pollen on the stigmas because frequently flying relatively long distances between consecutive flower visits (Herrera, 1987). The Xerces Society for Invertebrate Conservation began the North American 4th of July Butterfly Count in 1975, and it was passed on to the North American Butterfly Association in the late 1980s. That one-day count is now the largest butterfly monitoring program in existence in terms of geographic scope. In 2013, 424 U.S. counts were held in 44 states plus the District of Columbia. The 27 Canadian counts in 2013 were held in Ontario (22) and Saskatchewan (5); no data were collected in Mexico. Although there has been little analysis yet of these data, there is evidence of decline in at least some species (http://www.naba.org/pubs/ab141/ab141count_column.pdf).

NatureServe has assessed all 800 species of butterflies in the United States and has found that 141 (17%) are at risk of extinction (NatureServe, 2014). Twenty-six species of butterflies in the United States are listed as threatened or endangered under the Federal Endangered Species Act (U.S. Fish and

Wildlife Service, 2014). Many of the declining species are rare endemics, with a narrowly limited geographic range or very specific habitat requirements, but the decline in populations of the monarch butterfly (*Danaus plexippus*) in North America shows how a widely-distributed species can also be at risk (Brower et al., 2011). In Canada, a 2009 survey found that about one-third of the 300 species found there are at some level of risk (Hall, 2009). All five endemic species are at some level of risk and 23 are globally at some level of risk.

The United Kingdom Butterfly Monitoring Scheme, initiated in 1976, now records data from over 1,000 sites annually and has recorded declines in many species (https://www.gov.uk/government/statistics/england-biodiversity-indicators). Similar schemes have been active for more than ten years in the Netherlands, Belgium, and Germany, and the European Grassland Butterfly Indicator (http://www.eea.europa.eu/publications/the-european-grassland-butterfly-indicator-19902011) reports that from 1990 to 2011, grassland butterfly populations declined by almost 50%; change in rural land use is identified as a primary cause. There is also a French butterfly count focused on gardens, showing relatively stable populations from 2006 – 2013: http://www.noeconservation.org/index2.php?rub=12&srub=31&ssrub=98&goto=contenu. The European Red List of Butterflies (http://ec.europa.eu/environment/nature/conservation/species/redlist/downloads/European_butterflies.pdf) summarizes information about their conservation status; 8.5% (37 species) are designated as Threatened (including 0.7% being Critically Endangered), and another 10% as Near Threatened). The report concludes that most of these are declining rapidly in parts of their range and are in need of conservation action. The major drivers of butterfly habitat loss and degradation are related to agricultural intensification, although climate change plays a role, as do changes in management of forested and grassland areas that affect butterfly host plants and nectar resources.

Although some moth species are also important pollinators, there are even fewer studies of their population dynamics outside of economically important pest species. Some moths have closely coevolved relationships with their nectar plants, with a close correspondence between proboscis length and corolla size (Nilsson, 1998), although in Kenya Martins and Johnson (2013) found that adult hawkmoths are routinely polyphagous and opportunistic, regardless of their proboscis length. Many families of large moths, including sphingids, erebids, noctuids and geometrids, are very species-rich and also contain a large number of nectar-feeding species that are potential pollinators, but our knowledge of these primarily nocturnal pollinators is scant. More seems to be known about their distribution than their significance as pollinators, or population trends, but data on larger moths in Britain (http://butterfly-conservation.org/files/1.state-of-britains-larger-moths-2013-report.pdf) show a 28% decline from 1968 – 2007, with two-thirds of 337 species of common and widespread larger moths declining over the 40-year study.

Forister et al. (2011) and Casner et al. (2014) analyzed data from a decades-long study of butterfly distributions along an altitudinal transect in California's Central valley. They found that declines in the area of farmland and ranchland, an increase in summer minimum temperatures and maximum temperatures in the fall negatively affected net species richness, whereas increased minimum temperatures in the spring and greater precipitation in the previous summer positively affected species richness. Changes in land use contributed to declines in species richness (although the pattern was not linear), and the net effect of a changing climate on butterfly richness was more difficult to discern, but given the dramatic changes in the climate of that area (probably the most severe drought in 500 years – Belmecheri et al., 2015) it is not surprising that butterfly populations are being affected.

Most of these studies reporting changes in species richness or species abundance are not able to identify specific causes for declines. For one high-altitude butterfly species, *Speyeria mormonia*, Boggs and Inouye (2012) found that snowmelt date explained a remarkable 84% of the annual variation in population growth rate, but studies successfully identifying environmental factors driving population size remain rare.

Beetles (Coleoptera)

Beetles are the largest order of insects, and although they are relatively uncommon as pollinators, they have had a long evolutionary history with flowers (Gottsberger, 1977). They have also been overlooked in comparison to other groups of pollinators (Mawdsley, 2003). Beetle (weevil) pollinators are very important for oil palms, and they have been successfully introduced to tropical areas where these plants have been introduced; they now replace hand-pollination that was initially required (Greathead, 1983). They are also pollinators of some minor crops such as Annona (Podoler et al., 1984). There do not appear to be any studies of the trends in beetle pollinator populations.

Vertebrate pollinators

Two recent papers address the conservation status of vertebrate pollinators and the consequences of their loss. Aslan et al. (2013) estimated the threat posed by vertebrate extinctions to the global biodiversity of vertebrate-pollinated plants. While recognizing large gaps in research, their analysis identified Africa, Asia, the Caribbean, and global oceanic islands as geographic regions at particular risk of disruption of pollination (and dispersal). Plants that lose their mutualists are likely to experience reproductive declines of 40–58%, potentially threatening their persistence. A recent survey (Regan et al., 2015) of bird and mammal pollinators was undertaken using IUCN Red List data that are probably the best source for global information about extinction risk for threatened species. Of the 901 bird species reported as pollinators that they considered, 18 were uplisted (e.g., from Endangered to Critically Endangered) during the period 2008 – 2012, while 15 of the 341 mammal pollinators qualified for uplisting or were added to the list during the period 1996 – 2008. Thus, it appears that these two groups of vertebrate pollinators are suffering significant declines. This conclusion is also supported by reports of overhunting of flying foxes (Brooke and Tschapka, 2002), which are important pollinators and seed dispersers on some oceanic islands (Cox et al., 1991; Elmqvist et al., 1992).

Hummingbirds are charismatic pollinators in the New World. Some data for hummingbirds are available from the Breeding Bird Survey in the USA and Canada. Although sample sizes are relatively small, the time period surveyed (1962-2012) is long, and the data appear to be the best available for trends in population size. Three species (Table 3.1) show increases of between 1-2%/yr, while four others seem to be declining at 1-5%/yr. These are migratory species, which overwinter in Mexico or further south in Central America (e.g., Calder, 2004), and no data are available for their overwintering populations (it is not even clear where most of these birds are going in winter). However, based on the extent of habitat loss, it is estimated that the Mexican hummingbird populations may have declined by 15-49% in the past century (Berlanga et al., 2010). For some species there are extensive records available of the phenology of their migration in the USA and Canada, as observers across a large latitudinal gradient report their first sightings each spring (e.g., http://www.hummingbirds.net/map.html). Habitat loss in their overwintering and summer breeding grounds, and in the migration corridor, may pose threats to the migratory species, and there is the potential for effects of climate change on flowering phenology that may also create challenges for phenology of migration (McKinney et al., 2012). Hummingbirds are most diverse in the Neotropical and important pollinators of that flora, but information on population trends are completely lacking.

12. Table 3.1. Data on migratory hummingbird population trends from 1966 – 2012 from the Breeding Bird Survey data from USA and Canada.

Means for number of birds observed per survey route are shown with 95% Confidence Intervals. From http://www.mbr-pwrc.usgs.gov/bbs/ (data retrieved 15 September 2015).

Common name	**Species**	**USA annual trend (N)**	**Canada annual trend (N)**
Ruby-throated Hummingbird	*Archilochus colubris*	+1.6 (1,910) CI 1.3 – 1.8	+2.2 (387) first yr 1968 CI 1.3 – 3.0
Black-chinned Hummingbird	*Archilochus alexandri*	+1.1 (418) CI 0.2 – 1.9	+0.2 (10) CI -5.4 – 6.6

Anna's Hummingbird	*Calypte anna*	+2.0 (220) CI 1.3 – 2.7	
Costa's Hummingbird	*Calypte costae*	-1.9 (99) CI -6.0 – 1.1	
Calliope Hummingbird	*Stellula calliope*	-1.0 (168) CI -2.3 – 0.2	0.9 (52) CI -0.6 – 2.5
Rufous Hummingbird	*Selasphorus rufus*	-2.3 (214) CI -3.0 – -1.5	-1.9 (131) CI -3.1 – -1.6
Allen's Hummingbird	*Selasphorus sasin*	-4.1 (55) CI -5.7 – - 2.6	

Bird occurrence has been monitored in South Africa in two large citizen science projects, the first Southern African Bird Atlas Project (SABAP1: 1987–1991) and the second Southern African Bird Atlas Project (SABAP2: 2007-present); data at http://www.gbif.org/dataset/282d0ccb-4fa0-40f9-8593-105c77e88417. A recent comparison of these two data sets finds that the families Pycnonotidae and Ploceidae, which include nectar as a small component of their diet, have increased in abundance in 66% and 61% of geographical grid cells respectively, whereas the families Nectariniidae (Sunbirds) and Promeropidae (Sugarbirds), both of which include nectar as a major component of their diet, have increased in 52% and 33% of grid cells respectively. Because very few grid cells remain unchanged, these data indicated that the Promeropidae show a decline in about 67% of grid cells (Loftie-Eaton, 2014).

Bats are another important and diverse group of vertebrate pollinators in many parts of the world (Fleming and Mucchala, 2008). Population estimates are available for a few species of pollinating bats, but in general little is known about trends, in part because they are difficult to survey (http://digitalcommons.unl.edu/usgsstaffpub/35/). In some areas bats are important pollinators of food resources, such as cactus fruits in Mexico (Arias-Cóyotl et al., 2006), agave species (including those used for tequila and mezcal) (Rocha, 2005; Trejo-Salazar et al., 2015), species of mango, wild species of banana, durian, and guava (http://www.bats.org.uk/pages/why_bats_matter.html). The migratory species in Central and North America face many of the same challenges described above for migratory hummingbirds, as well as the additional constraint of needing caves for roosting (Slauson, 2000). One study found that an island population of a columnar cactus may be moving toward insect pollination because of a paucity of bats, possibly a consequence of hurricanes (Rivera-Marchand and Ackerman, 2006).

Pollinator extinction, reintroduction, and replacement

Local and global extinctions of pollinators have occurred (Cox and Elmqvist, 2000; Cameron et al., 2011), and some conservation efforts have been implemented to re-introduce missing species or replace their functions as pollinators. An example of re-introduction (See Chapter 6 for additional information on re-introductions) is the case of the United Kingdom bumble bee species *Bombus subterraneus*, which was declared extinct in the UK in 2000. An initial attempt to use queens from New Zealand in 2011 for reintroduction was unsuccessful (Howlett et al., 2009). However, a subsequent programme to reintroduce *B. subterraneus* with queens from Sweden is ongoing following restoration of appropriate habitat and food plants (Gammans, 2013), and although workers have been observed, production and successful overwintering of queens has not yet been proved. This re-introduction protocol developed for *B. subterraneus* in the UK may be useful in other parts of the world experiencing similar bumble bee extinctions (e.g., *B. occidentalis* in parts of its former range in North America) (Cameron et al., 2011). A fortuitous replacement of pollination occurred in Hawaii, where the introduced Japanese White-eye (*Zosterops japonica*) assumed the role of extinct bird species as a pollinator of the ieie vine (*Freycinetia arborea*) (Cox, 1983).

The current status of almost all wild pollinator populations is unclear and difficult to assess due to the lack of data. A few of the re-surveys mentioned in sections 3.2.2 and 3.2.3 suggest that pollinator populations (diversity and abundance) can be maintained over long periods of time if habitat that provides nesting sites and food resources are conserved. General trends across studies indicate that the challenges posed by habitat loss or alteration, introduction of diseases, alien competitors and invasive plant species, and increasing pesticide use, are resulting in substantial shifts and often declines in pollinator populations that have prompted concern for their future. One important trend that can be extrapolated from comparative surveys between disturbed and undisturbed sites (e.g., Chacoff and Aizen, 2006; Quintero et al., 2010) is that massive habitat disturbance could not only lead to impoverished pollinator faunas, but also to a spatial homogenization of bee communities (decreased beta diversity) (see also Carvalheiro et al., 2013 and Chapter 2).

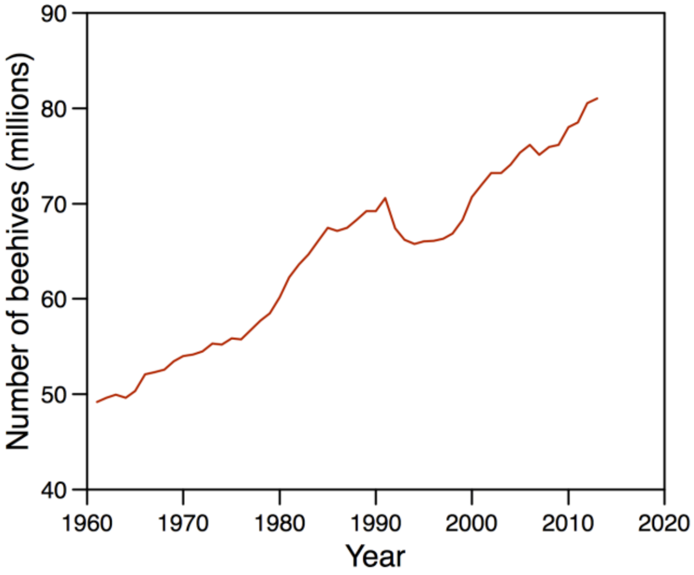
3.3 Trends in managed pollinators

3.3.1 Outline of section

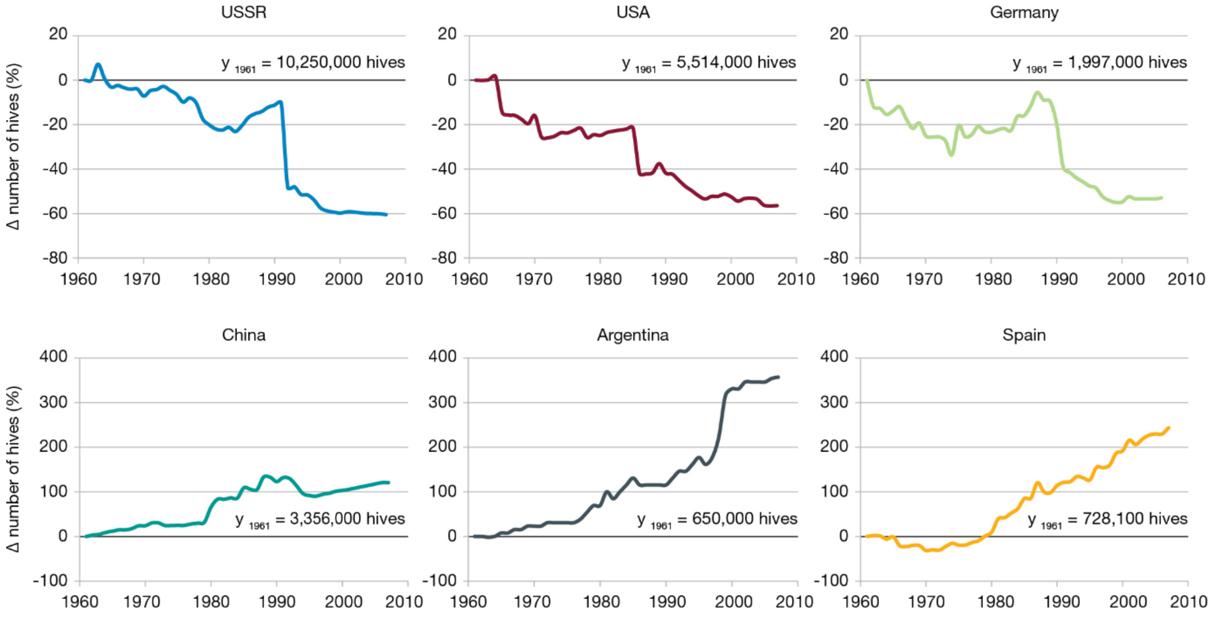
Managed pollinators include the well-known honey bee (*A. mellifera*) as well as a growing number of other bee species and other insects such as flies. The number of managed colonies of the western honey bee (*A. mellifera*), the major commercial pollinator, has increased on a global scale over the past 50 years. The diversity of additional bee species nowadays managed for pollination, including bumble bees, stingless bees, solitary bees as well as other insects, has also increased (Bosch, 2005; Velthuis and van Doorn, 2006). This is in part because of their greater efficiency in pollinating specific crops. It is also in part to reduce reliance on a single managed pollinator, the honey bee, which is perceived to incur rising management costs due to treatment against emerging pathogens and increasing rates at which colonies die and need to be replaced. Local species should be chosen to avoid risks associated with importation of non-native species. Research is also needed to identify efficient pollinators of crops under threat of pollination shortfall (e.g. Giannini et al., 2015b) and then to develop methods to ensure sustainability of pollinator numbers, either through appropriate land management or development of rearing techniques.

3.3.2 Honey bees (*Apis* spp.)

There are inherent difficulties in determining trends in the number of honey bee colonies for biological and sociological reasons, and these trends are often conflated with rates of colony mortality. Specifically, it is difficult to determine the number of honey bee colonies in a geographic locality because, firstly, unlike other livestock, a honey bee colony can be divided by a beekeeper into two or more parts during the active season to multiply colony numbers and, conversely, colonies can be united into one in periods of flower dearth or cold temperatures. Secondly, an entire honey bee colony may depart (abscond) or be acquired as a passing swarm. Thirdly, beekeeping is a labour-intensive activity and colonies are often not registered. Fourthly, there are unknown numbers of wild honey bees, e.g., in Africa, and feral Africanized honey bees in South, Central and southern North America. Fifthly, there is probably variation across nations, and even across years within a country, in how data on colony numbers are collected. These factors conspire to hamper acquisition of colony numbers (the total number of colonies at any one point in time) and annual rates of colony mortality (the proportion of colonies that die in one year). Indeed, rates of colony mortality have recently been reported to be much higher than the usual rate of ca. 10%, and up to 30% or more since the winter of 2006-to-2007 in some parts of the temperate Northern Hemisphere (Oldroyd, 2007; see Chapter 2, sections 2.5 and 2.6), and may be equally high in South Africa (Pirk et al., 2014). Information from the FAO database nevertheless suggests an increasing world number of managed colonies of honey bees (Figure 3.2), a trend driven by Asia (Aizen and Harder, 2009b, Barron, 2015, see Chapter 2, Figure 3.3), with a current world stock of greater than 81 million hives, each comprising 10,000-40,000 or more worker honey bees (Figure 3.2). Within this global increase in stock, some countries have suffered declines whereas others have seen growths (Figure 3.3, Aizen and Harder, 2009b, Potts et al., 2010a, b, Smith et al., 2013). Even neighbouring countries (e.g., the USA and Canada) may exhibit contrasting growth rates in the stock of honey-bee hives (Figure 3.4).



20. Figure 3.2. Number of managed colonies of *Apis mellifera* in the world.
Extrapolation of the line suggests ca. 100 million managed colonies by 2050. Data compiled from FAOSTAT (www.faostat3.fao.org).

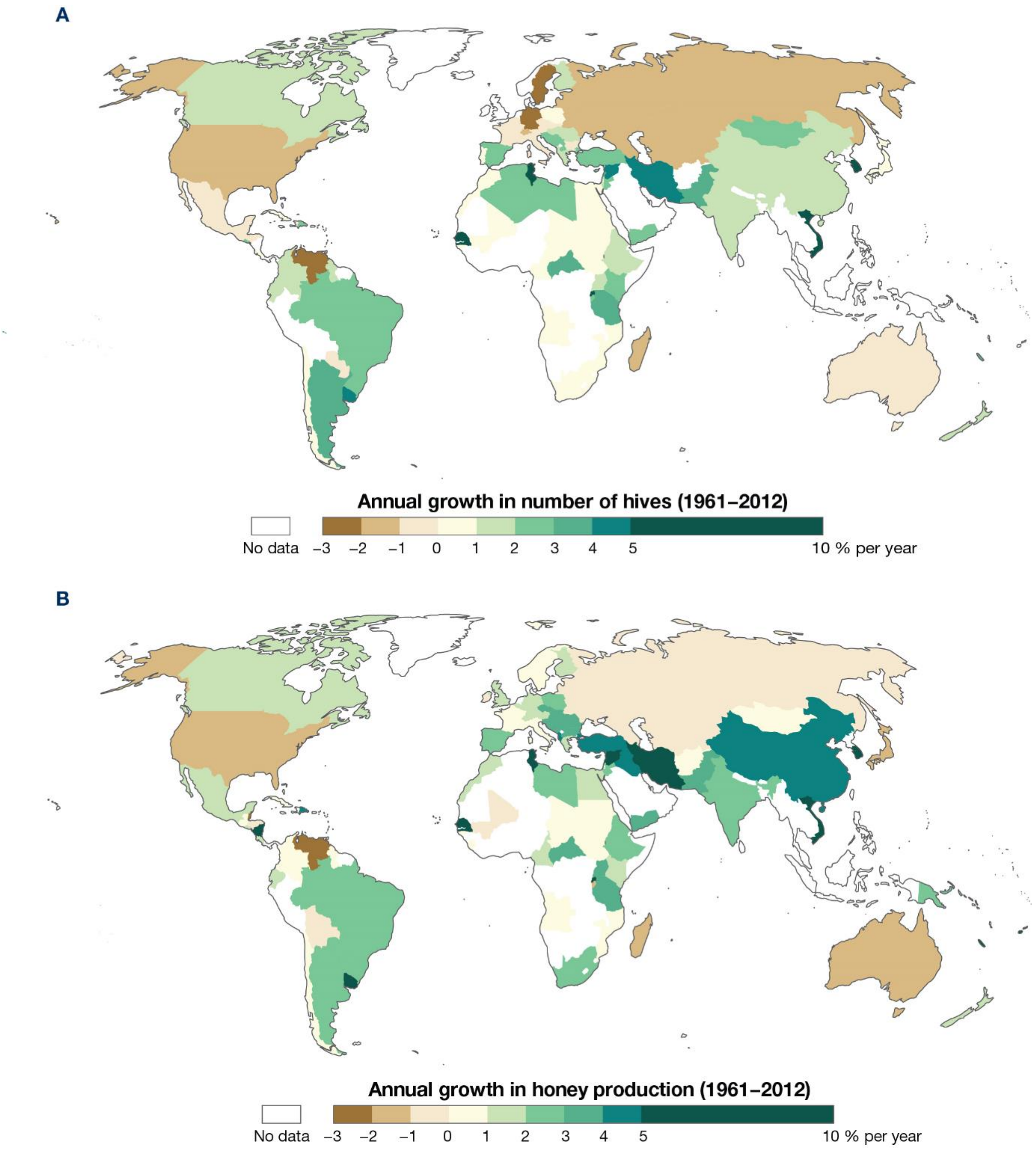


21. Figure 3.3. Number of managed colonies of *Apis mellifera* in selected countries,
showing overall losses in some countries and gains in others (from Aizen and Harder, 2009a, Suppl. Materials).

This overall pattern of increasing numbers of honey bee colonies worldwide is rather robust and may reflect an increasing market value of honey and of honey bee colonies as pollination 'units', though not withstanding regional trends global honey production has consequently increased with the number of colonies to ca. 1.6 million tonnes annually in 2013, though the risk of a drop in production varies widely across the world (Chapter 6, Table 6.2.2). Some developed countries in the temperate Northern Hemisphere have witnessed an on-going decrease in colony numbers since the 2nd World War (Figure 3.3, Aizen and Harder, 2009a), possibly a consequence of societal changes (e.g. increasing wealth, collapse of communism; see Moritz et al., 2010, vanEngelsdorp and Meixner, 2010, Smith et al., 2013), intensification of land use, and emerging pests and diseases (see Chapter 2, sections 2.5.1 and 2.6). Indeed, wild (where they are native) and feral (where they are introduced) *A. mellifera* have almost disappeared from the temperate Northern Hemisphere (Jaffé et al., 2010). However, socioeconomic factors affecting the honey market seem to be the primary cause for long-term trends in the growth in the number of honey-bee hives (Aizen and Harder, 2009b). For instance, countries exhibiting negative growth rates in the number of hives can exhibit positive rates in honey production (e.g., Germany, France; Figure 3.4), which is inconsistent with a scenario of declines in the stock of honey-bee hives being driven by disease. Furthermore, there has been an increasing global trend in honey production per hive over the last five decades, so that today an average hive produces approximately 50% more honey than 50 years ago (Aizen and Harder, 2009a).

Genomic analyses indicate that managed honey bees have not suffered from a reduction in genetic diversity, either where they are native in Europe (Wallberg et al., 2014) or where they have been introduced to North America (Harpur et al., 2012). These introductions have also affected other honey bee species; for example, in South Korea, *A. cerana* was widespread in beekeeping into the 1980s, but the current trend is toward managed *A. mellifera* to the loss of managed *A. cerana* (Jung and Cho, 2015).

The wisdom of importing *A. mellifera* into locations where it is non-native, or even moving it between locations where it is native, has been called into question for many reasons (see Chapter 2, Section 2.5). There has consequently been increasing interest in managing other species for crop pollination. From a review of the current literature presented below, it is estimated that approximately 1% of the world’s 20,000 bee species are under consideration and 0.1% are currently under active management for commercial pollination. This trend in increasing interest in or use of other insect species for crop pollination may also have been driven by perceived or real shortfalls in the pollination of crops.



22. Figure 3.4. World map showing the annual growth rate (%/yr) in the number of honey bee colonies and honey production for countries reporting those data to FAO between 1961 and 2012 (FAOSTAT 2013). Data from the countries that were part of the former Soviet Union, the former Yugoslavia, or the former Czechoslovakia were combined.

3.3.3 Bumble bees (*Bombus* spp.)

Once techniques for commercial rearing of one bumble bee species, *Bombus terrestris*, were developed in the 1980s, the number of managed colonies of this species traded annually rose to one million in 2006 (Velthuis and van Doorn, 2006). Nowadays they are used commercially to pollinate tomatoes (Velthuis and van Doorn, 2006) and over 240 crops worldwide. Acquiring information on current (2014) numbers of *Bombus* colonies traded annually is problematic because such information is withheld by rearing companies and there is no obligation to report commercially sensitive information. An estimated two million *Bombus* colonies are traded annually across the world for pollination of mainly crops grown under enclosure (e.g., tomatoes in glasshouses), but increasingly also for open field pollination (see Chapter 2, Table 2.4.2 for a list of commercialized *Bombus* species). In some countries, native *Bombus* species are commercialised (e.g. *Bombus impatiens* in eastern North America, *Bombus ignitus* in Japan), reducing risks associated with the importation of non-native subspecies (i.e., genetic introgression) and species (i.e., competition among pollinator species, pathogen transmission, and the spread of insect-pollinated weeds; see Chapter 2, Table 2.3). In many other cases, bumble bees have been imported as an exotic pollinator (e.g., *B. impatiens* to the west and south of its native range in North America and further afield), with strong evidence of negative consequences for native *Bombus* species through the unwitting introduction of exotic pathogens on imported exotic bumble bee species (e.g., *B. terrestris* in Chile and Argentina; Morales et al., 2013, and see Chapter 2, Sections 2.5 and 2.6). This so-called pathogen spill-over is also established for commercially-reared bumble bee species transported within their native ranges (Colla et al., 2006, Murray et al., 2013).

3.3.4 Stingless bees

Other bee species could be developed for pollination using local knowledge, particularly social species such as stingless bees in the tropics (Meliponini; e.g. Slaa et al., 2006; Gannini et al., 2015a; Jaffé et al., 2015) with nests including 100s to 10000s of individuals (Roubik, 1989), a large workforce of potential pollinators. As detailed in Chapter 2 (Section 2.5.2), stingless bees comprise a group of several hundred species distributed across the tropics, some of which have been traditionally managed in clay or wooden pots and harvested for honey (Free, 1982; Crane, 1983, 1999, see Cortopassi-Laurino et al., 2006 and Vit et al., 2013 for details of managed species). The Maya of the Yucatan Peninsula and adjacent lowlands comprising present-day Belize, Guatemala and Mexico developed sophisticated management of one species, *Melipona beecheii,* that sufficed for local needs for sugar (honey) and wax such that Spanish Settlers did not import honey bees from Europe to the Yucatan Peninsula in the 17th Century, unlike in other localities in Central and South America (Quezada-Euán et al., 2001). In marked contrast to the growth of other managed pollinators, indigenous local knowledge strongly suggests that *M. beecheii* colonies are declining in number in Yucatan, as are traditional meliponicultural practices, reflecting what indigenous people perceive as ‘an imbalance with nature’ (Quezada-Euán et al., 2001, and see Chapter 5). Reduction in the number of colonies may in part be due to habitat degradation through deforestation (Freitas et al., 2009), a problem that is thought to compromise meliponiculture with other stingless bees in Mexico and elsewhere across the tropics (Cortopassi-Laurino et al., 2006). Brazil provides another example; stingless beekeepers scattered across Brazil were asked in a recent survey to assess the status of wild stingless bee populations (Jaffé et al., 2015), and 92% of the interviewed beekeepers replied there are now less wild stingless bees than 50 years ago. Although this was not a quantitative assessment, the authors stressed the value of beekeepers' opinion, given that they have a close relationship with their bees, constantly assess the natural resources they use, and frequently collect wild colonies. These findings suggest that many wild stingless bee populations have declined in Brazil during the last decades.

3.3.5 Solitary bees

Several leafcutter and mason bees (family Megachilidae, genera *Megachile* and *Osmia* respectively) have been produced in artificial nesting media (e.g., drinking straws, bamboo canes, drilled wood blocks and polystyrene boards) and are managed on a small scale (see Chapter 2, Section 5.4, Table 4). A simple approach is to place appropriate nesting substrate out in the field, close to crops requiring pollination, and to allow natural populations of these bees to build up in numbers over successive years (Free, 1993; Delaplane and Mayer, 2000; Howlett et al., 2009).

In some instances, cocoons of leafcutter and mason bees can be harvested and traded (though we note potentially negative consequences of exportation because of the risk of spread of exotic diseases, see Chapter 2, Section 5.4). Indeed, the vast majority of cases of managed pollination with *Megachile* and *Osmia* involve the release of managed populations into the field or orchard. These populations are reared under appropriate temperature conditions throughout development and wintering and their release is timed to the bloom of the target crop the following year (Richards, 1984; Bosch and Kemp, 2001). A significant pollinator industry has built up around one species, the alfalfa leafcutter bee *Megachile rotundata*, in the USA and Canada, where it is non-native (Stubbs and Drummond 2001; Stickler and Cane, 2003), with excellent guides to the management of this bee species (e.g., Richards, 1984). Alfalfa (*Medicago sativa*) is a Eurasian crop introduced to North America as an important foodstuff for cattle, but honey bees are often poor pollinators of alfalfa (Free, 1993). Following a presumably earlier unintentional introduction of *M. rotundata* to the east coast of North America around 1930 from Eurasia, where it is native, the species had made its way to central and western USA by the 1950s, where large areas of alfalfa were grown for seed. Not only is *M. rotundata* an excellent pollinator of alfalfa, it also nests gregariously in artificial domiciles. Targeted research revealed important aspects of its biology, and a viable industry in alfalfa leafcutter bee management became established (Stephen 1961, 1962; Bohart, 1962; see also Pitts-Singer and Cane, 2011).

Official figures on the size of the industry (number of bees produced) are lacking, but an estimated 800 million alfalfa leafcutter bees are traded commercially per year and a further 1,600 million are encouraged in and around alfalfa fields by bee-friendly farming practices and provision of nesting medium in the USA (Peterson et al., 1992, Reisen et al., 2009), with a sizable industry in Canada, too, that also supplies *M. rotundata* from largely pathogen-free areas in Canada to the US market. Land surrounding alfalfa fields in the USA is also occasionally managed for the ground-nesting alkali bee *Nomia melanderi*, which is also an efficient alfalfa pollinator. Management involves not only the enhancement of natural nesting sites but also the actual building of completely artificial nesting sites, called bee beds (Johansen and Mayer, 1976). The species has not, though, been commercialised to any extent (Cane, 2008), and neither has *Rhophitoides canus*, another ground-nesting solitary bee successfully managed for alfalfa seed production in eastern Europe (Ptacek, 1989 in Bosch, 2005). Pollination of alfalfa makes a strong case for the diversification of managed pollinators.

Leafcutter and mason bees are all solitary, and the diversity of these and other species employed in (semi-)commercial enterprises remains small. In Japan, the native *Osmia cornifrons* has been successfully managed since the 1940s for improved apple pollination (Yoshida and Maeta, 1988; Maeta, 1990), where it is traded and used to pollinate 70% of the apple production area (Maeta, 1990). In Europe, an estimated one million *Osmia bicornis* (=*rufa*) bees are traded per year for apple and other fruit pollination by 10-20 small companies, while *Osmia cornuta* in central and southern Europe and *Osmia lignaria* in the US and Canada are being traded to the same or greater extent for the pollination of orchard crops (Bosch and Kemp, 2002). In the tropics, other largely solitary species such as carpenter bees (genus *Xylocopa*) have been experimentally managed as potential pollinators of crops such as passion fruit (*Passiflora edulis*, Junqueira et al., 2012; Junqueira et al., 2013), whose flower morphology does not allow efficient pollination by honey bees. In Australia, the native blue-banded bee *Amegilla chlorocyanea* is as efficient as bumble bees in pollinating tomatoes grown in glasshouses (Hogendoorn et al., 2006). This list in not exhaustive.

3.3.6 Other managed pollinators

The commercial management of other insect pollinators has great potential (Kevan et al., 1990, Howlett, 2012), but is currently on a much smaller scale than that of honey bees, bumble bees or solitary bees. Flies were occasionally used for strawberry pollination in the 20th Century (Free, 1993). However, this practice has been largely replaced by *Bombus* pollinators, considered more efficient crop pollinators than flies. Bumble bees need to gather large quantities of pollen and nectar for their offspring and so are far more consistent flower visitors than flies (Free, 1993). Blowflies and syrphid flies can also be important pollinators of crops grown for seed in cages (to control cross-pollination), e.g. the blowfly *Calliphora vomitori* for the pollination of onion grown for seed (Currah and Ockendon, 1983), and are also available commercially. As mentioned above, another species of fly, *Lucilia sericata* (common green bottle fly), is available commercially for pollination (section 3.2.3).

3.4 Trends in introduced pollinators and transmitted pathogens

3.4.1 Outline of section

This section provides a general overview of trends in pollinator introductions to novel habitats and its ecological effects, especially those related with disease transmission. It discusses the different concepts related to pathogen transmission and the existing evidence for honey bees, bumble bees, and leaf cutter bees. The frequency of introduced species and the prevalence of different infectious diseases on a worldwide scale is summarised according to the existing evidence.

3.4.2 Ecological effects of introduced pollinators

In this section, the term ‘introduced species’ will be used as synonymous with ‘non-native species’ to denote a species that lives outside its original distributional range, which has arrived there by human activity, either deliberate or accidental, and is able to survive and reproduce in the new habitat without human assistance. An introduced species might become an invasive species if it can outcompete native species for resources such as nutrients, light, physical space, water, or food. This definition excludes most garden and farm organisms, which will be denoted as ‘managed species’.

Introduced pollinators may affect native species and ecosystems through various mechanisms, such as (a) exploitative or interference competition for flower resources and nesting sites, (b) transmission of parasites or pathogens to native populations, including the co-introduction of natural enemies, (c) inadequate pollination of native flora leading to changes in the reproduction of native plants, and (d) undesirable pollination of exotic flora. Each mechanism can, in principle, propagate its effects to the rest of the community through indirect interactions and cascading effects (Goulson et al., 2008). Although the introduction of pollinators for commercial purposes is often considered positive, various reports have documented detrimental effects. For example, the introduction of the honey bee has been correlated with a decline in native bee and bird species, especially on islands, presumable due to competitive effects (e.g., Hansen et al., 2002; Dupont et al., 2003). Introduced bees are also known to reduce the reproduction of native plant species (Gross and Mackay, 1998), and enhance pollination and establishment of exotic weeds (Barthell et al., 2001; Stout et al., 2002). The final fitness impact, measured as survival and/or reproductive success, of introduced pollinators on plants will depend on several factors, among which the time elapsed since pollinator introduction may be a critical one (Esterio et al., 2013).

In spite of the fact that introduced pollinators seem to have a broad detrimental effect on ecological variables of novel habitats, most evidence is of a correlative nature and suffers from methodological limitations (Paini, 2004; Kenis et al., 2009; Jeschke et al., 2012). In consequence, the extent to which introduced pollinators have a truly detrimental impact on native communities and biodiversity remains controversial (but see Thomson, 2006; Kenta et al., 2007). Although this limitation may reflect the difficulty of carrying out experimental studies under field conditions rather than the absence of phenomena, the low number of experimental studies precludes unequivocal generalizations at the community level at present.

3.4.3 Spread of diseases through introduced pollinators

Parasitism is thought to be an important driver of population declines in pollinator species. While mostly studied in honey bees and bumble bees, the impact of parasitism is not limited to insect pollinators. For example, white-nose syndrome is a bat disease caused by the fungus *Geomyces destructans* that has killed millions of hibernating bats in North America (Turner et al., 2011), with unknown consequences for ecological functions in terms pollination and seed dispersal.

Recently, there has been an increased awareness of the effects of parasite spread from original to novel hosts, due to the intentional or accidental introduction of honey bees and bumble bees to novel habitats. Parasite-mediated competition is the indirect ecological interaction that occurs when a host species out-competes, and eventually may lead to extinction of, a second host species by transmitting a novel parasite to which the second host has not evolved appropriate defensive mechanisms (Price et al., 1988).

Spillover, that is, the spread from a heavily infected “reservoir” host species to a sympatric ‘non-reservoir’ species has been identified as a risk factor that may affect population persistence of non-reservoir species under natural conditions (Otterstatter and Thomson, 2008; Fürst et al., 2014; Graystock et al., 2013). As millions of commercially-produced honey bee and bumble bee colonies are grown and traded annually for pollination, there is a real threat of pathogen spillover from commercially-produced colonies to natural populations (Fürst et al., 2014). The expectation of 100 million managed colonies of *Apis mellifera* in the world by the year 2050 (Figure 3.2) suggests spillover could be a major mechanism of parasite transmission (Fürst et al., 2014).

Just as introduced managed species can transmit pathogens to wild species, transmission could also occur in the opposite direction from wild bee populations to managed bee species, and these in turn may return the infectious disease to native species in an amplified and perhaps more virulent way. Diseases may be kept at low prevalence under natural conditions. However, the arrival of densely-populated commercial apiaries may amplify disease prevalence and cause commercial species to behave as reservoirs that transmit the pathogen back to wild bees, a phenomenon known as spillback. Even though evidence for this phenomenon is still scarce in the literature (Kelly et al., 2009; Schwarz et al., 2014), this mechanism could account for a higher frequency of disease transmission than is currently recognized. While the role of parasites has begun to be understood only in recent years, current evidence suggests they might have been involved in the conspicuous decline of bee and bumble bee species of North America, and South America (e.g., Otterstatter and Thomson, 2008; Cameron et al., 2011; Evison et al., 2012; Schmid-Hempel et al., 2014; Fürst et al., 2014).

Bumble bees

Bumble bees can cause a variety of problems for local pollinator communities when they are moved around the world (Dafni et al., 2010). At least four species of *Bombus* have been introduced to new countries to enhance crop production. For example, *B. hortorum, B. terrestris, B. subterraneus*, and *B. ruderatus* were introduced from the UK to New Zealand. *Bombus terrestris* has been also directly introduced from Europe to Israel, Chile, Asia, Central America, Northern Africa, and secondarily introduced from Israel to Chile, and from New Zealand to Japan and Tasmania. *Bombus ruderatus,* in turn, was introduced from the UK to New Zealand, and secondarily from New Zealand to Chile. Both *B. terrestris* and *B. ruderatus* spread secondarily from Chile to Argentina (Montalva et al., 2011). Figure 3.5 summarizes the main routes of invasion of *Bombus* species in the world. There is a clear primary source of invasion originating in Europe. A secondary source of invasion started in New Zealand. A conspicuous non-intentional spread has occurred from Chile to Argentina, with a subsequent spreading in the Argentinean territory, a process that is currently ongoing (Morales et al., 2013).

The distribution of native and introduced bumble bees is depicted in Figure 3.6. The North Neotropical and South Nearctic regions have 10 and 19 species, respectively, with only one introduced species in both cases, *Bombus terrestris*. The West Nearctic region has a high diversity of native bumble bees (44 species), and marked declines in the abundance of *Bombus affinis, Bombus terricola,* and *Bombus occidentalis* appear to be related to the presence of nonnative fungi *Nosema bombi* and *Nosema ceranae* (Microsporidia: Nosematidae) and the trypanosome *Crithidia bombi* (Kinetoplastida: Trypanosomatidae). Such diseases were probably introduced when colonies of *B. occidentalis* and *B. impatiens* were shipped to Europe in the early 1990s, and then imported back to the U.S. for commercial greenhouse pollination (Evans et al., 2008). A more recent analysis based on IUCN Red list criteria revealed that over 25% of all North American bumble bee species are facing some degree of extinction risk (Hatfield et al., 2014), suggesting that parasites may be also responsible for population declines in other bumble bee species. The only introduced species to Japan is *Bombus terrestris* which, as in Southern South America, New Zealand, Israel, and Tasmania, has become naturalized. In Japan, the tracheal mite *Locustucarus buchneri* has been transmitted from the introduced *Bombus terrestris* to the native bumble bee species. In contrast, the Japanese mite was once introduced into Europe through the transport of the Japanese bumble bee queens for commercialization, and it infested *B. terrestris* colonies in the production plant (Goka et al., 2001; Goka et al., 2006), with unknown effects for bumble bee survival and reproduction.

The spread of natural enemies of wild bumble bee colonies may be aided by introduced bumble bees or honey bees, which can act as vectors for bumble bee parasites (e.g., Colla et al., 2006; Ruiz-González and Brown, 2006; Otterstatter and Thomson, 2008). In this regard, attention should be paid to regions receiving high numbers of introduced honey bees and bumble bee species, as it is feasible they harbor diseases otherwise absent in native bumble bee (and other wild pollinator) populations. A review of published data on bumble bees reveals that New Zealand has received the highest number of bumble bee species (4 species) (Figure 3.6), albeit transmission to other congeneric species is zero as no native bumble bee species exists there. However, this does not mean that potential diseases are precluded from transmission among invasive bumble bees and to other native bee species because host shifts have been probably underestimated in natural populations (see reviews in Woolhouse et al., 2005; Potts et al., 2010a). A more critical scenario occurs in southern South America where two out of seven bumble bee species are introduced (*Bombus terrestris* and *Bombus ruderatus*), which suggests a high potential for disease transmission to native species (see Morales et al., 2013; Schimd-Hempel et al., 2014).

Even though more information on disease prevalence is required, the current information available suggests infection is variable across regions. The prevalence of diseases in bumble bees is shown in Figure 3.7. Inspection of patterns reveals that *Bombus* of southern South America, England, and southwest North America harbour the highest diversity of parasites, suggesting a high risk of transmission to other native bee species. The most frequent pathogens are *Nosema*, a microscopic spore-forming microsporidian. *Nosema bombi* is a cosmopolitan species that has been found widely across *Bombus* species, though at varying prevalence. Likewise, *Nosema ceranae* is associated both with *Apis mellifera* and *Bombus* species (e.g., Plischuk et al., 2009; Fürst et al., 2014). *Crithidia* is a trypanosome protozoan that can be contracted at flowers via fecal transmission (Durrer and Schmid-Hempel, 1994). *Crithidia bombi* has been detected in England and different *Bombus* species in the Neotropics (Figure 3.7). *Apicystis*, a protozoan present in commercial *Bombus* colonies and transmitted to native *Bombus* species (e.g., Arbetman et al., 2013), has been found in England, southwest North America, and southern South America. Recently, the honey bee viral pathogen deformed wing virus, has been found to be both virulent and widespread in UK populations of *B. terrestris* and *B. lapidarius* (Fürst et al., 2014).

Honey bees

A variety of bee species have been deliberately released in parts of the world to which they are not native. However, with more than 25 subspecies recognized at present (Engel, 1999), the honey bee, *Apis mellifera,* thought to be native to Africa, western Asia, and Southeast Europe (Michener, 1974), has been managed and introduced to almost every country in the world. With the exception of Antarctica, it has a global distribution (Goulson, 2003). One of the most rapid and spectacular biological invasions is that of the African subspecies *Apis mellifera scutellata*, introduced into Brazil in 1957 in an attempt to improve the productivity level provided by the European honey bee subspecies. African swarms escaped into the wild and established a solid population that has spread throughout Latin America up to the southwest of North America, reaching a population density of 6-100 colonies/km^2 in the Neotropics (Roubik, 1983).

The remarkable ability of this bee to displace other poorly adapted populations of the European honey bee, and the threats it poses for human activities has stimulated an enormous amount of research (reviewed in Schneider et al., 2004). A variety of factors seem to determine the rapid spread of Africanized honey bees across the Neotropics. For instance, the colonies of this subspecies can grow at an extremely fast rate of 16-fold per year (Schneider et al., 2004), which coupled with the long foraging distance (Goulson, 2003), make this bee an efficient colonizer of new habitats. In addition, it has been documented that *Apis mellifera* is highly generalist, visiting a hundred or more different plant species within a region, and nearly 40,000 different plant species worldwide (Crane, 1999). This extremely high level of generalization confers populations a high degree of adaptability to new environmental conditions (Butz Huryn, 1997; Schneider et al., 2003).

Regarding mating behavior, European queens tend to mate disproportionately with African over European males (Schneider et al., 2004), which leads to a dominance of African alleles for some characters. One of these traits is the higher resistance shown by the African subspecies to the parasitic mite *Varroa destructor* and the tracheal mite *Acarapis woodi* (Schneider et al., 2004).

Honey bees and their nests support viruses and a high diversity of microorganisms such as bacteria, fungi and protozoa whose effects range from antagonistic to mutualistic. Diverse pathogens have been suggested, often in interaction with pesticides, to cause important detrimental effects such as “colony collapse disorder (CCD)" (Durrer and Schmid-Hempel, 1994; Johnson et al., 2009; Gillespie, 2010; Evison et al., 2012; Graystock et al., 2013); the honey bee chalkbrood, caused by the fungus *Ascosphaera apis*, foulbrood caused by the bacterium *Paenibacillus larvae*, the microsporidian *Nosema apis*, and the mite *Varroa destructor* (Goulson, 2003). The small hive beetle *Aethina tumida*, transported from Africa to North America has become an important pest of commercial colonies (Evans et al., 2000).

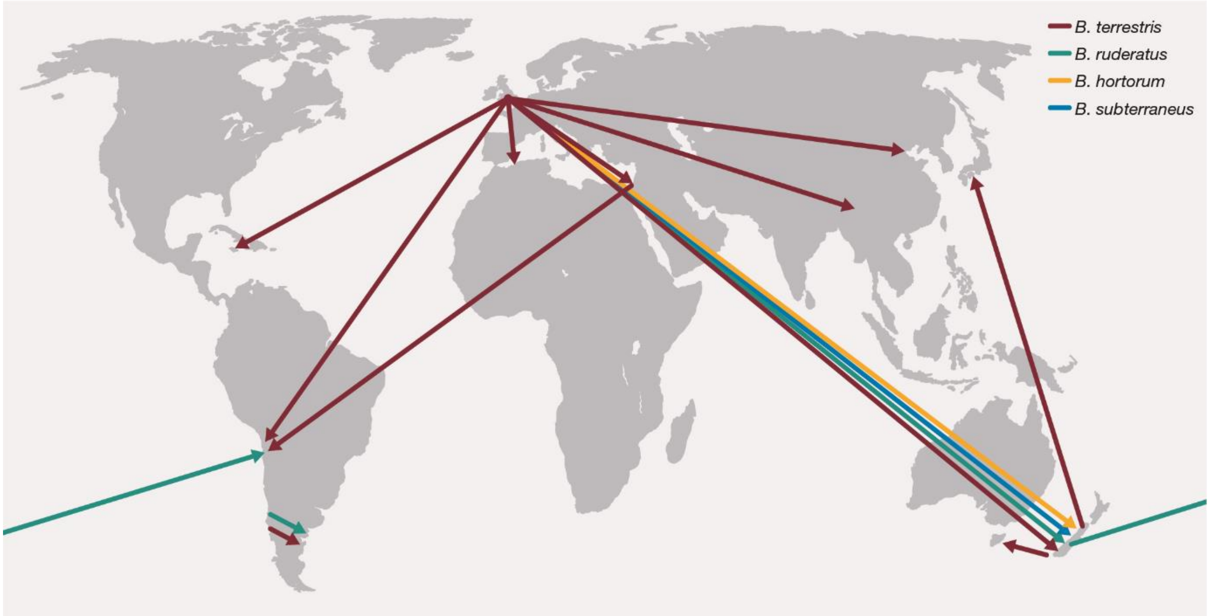
Leafcutter bees

Six exotic Megachilidae bees have been introduced to North America for crop pollination: *Megachile rotundata* (native to Eurasia), *M. apicalis* (native to Eurasia), *M. concinna* (probably introduced from Africa), *Osmia cornifrons* (native to East Asia), *O. cornuta* (native to Europe), and probably *O. taurus* (native to East Asia) (Frankie et al., 1998; Gibbs and Sheffield, 2009). *Megachile rotundata*, the alfalfa leafcutter bee, is by far the most studied megachilid species. It was accidentally introduced to North America by the 1940s. Because of its high efficiency in alfalfa (*Medicago sativa*) pollination, this species has been introduced to Australia and New Zealand (Goulson, 2003). The value of *M. rotundata* as pollinator of field crops is only surpassed by the honey bee, *Apis mellifera*.

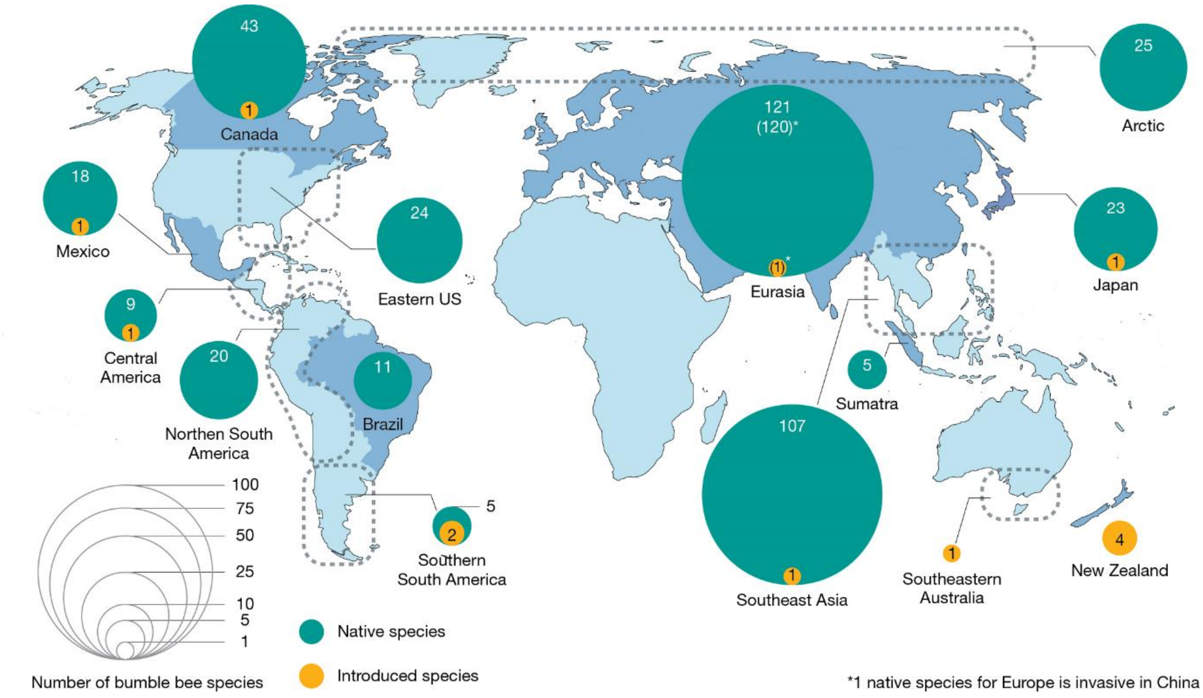
Megachile rotundata is attacked by disease pathogens, from which the most common is the fungus *Ascosphaera aggregata* (Ascomycete), which causes chalkbrood. Even though *M. rotundata* host many natural enemies native to Eurasia (Eves et al., 1980), no pest that accompanied *M. rotundata* introduction has been detected in North American native bees (Pitts-Singer and Cane, 2011). Most knowledge of parasitism and predation on this species comes from managed populations, where up to 20% of cells can be parasitized by wasps (Pitts-Singer and Cane, 2011).

Even though the variables involved in the management of *M. rotundata* for alfalfa pollination and seed production have been extensively studied, information about its ecological impact on native plant and bee species is almost inexistent. Surveys carried out in USA have reported that *M. rotundata* is rare, occupying 3-4% of available nesting sites, or absent in wild conditions. In spite of its low abundance in the wild, *M. rotundata* may eventually contribute to the invasion of weedy species in North America as it shows preference for sweet clovers (*Melilotus alba* and *M. officinalis*) and purple loosestrife (*Lythrum salicaria*) over alfalfa in choice tests (Small et al., 1997).

In comparison to honey bees and bumble bees, there is a clear knowledge gap on the ecological consequences of megachilid introduction that need to be addressed in future studies.

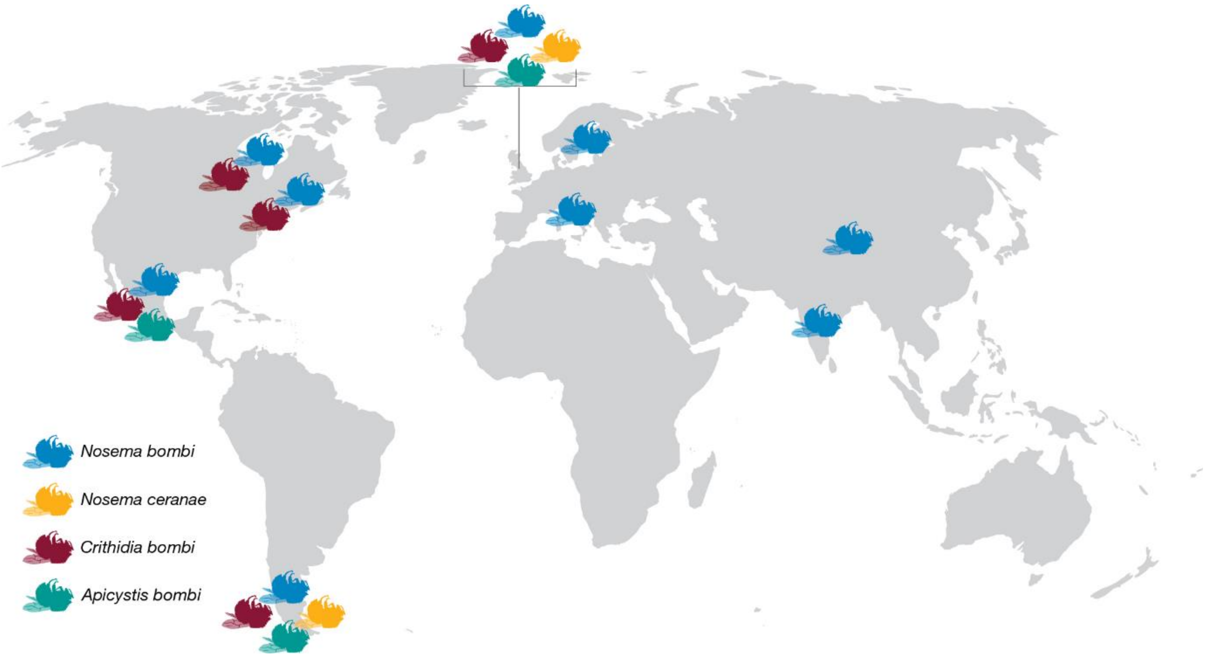


23. Figure 3.5. Global introductions of European bumble bees, *Bombus* spp.



24. Figure 3.6. Number of introduced (yellow) and native (teal) bumble bee species in biogeographic regions described by Williams (1996).

Bubble size reflects the bumble bee species richness at each region. Data updated by P. Williams, British Natural History Museum (www.nhm.ac.uk/research-curation/projects/bombus/).



25. Figure 3.7 Presence or absence scheme for the most frequent parasite species in bumble bees. Regions without pies have not been examined for parasites and do not represent parasite-free regions.

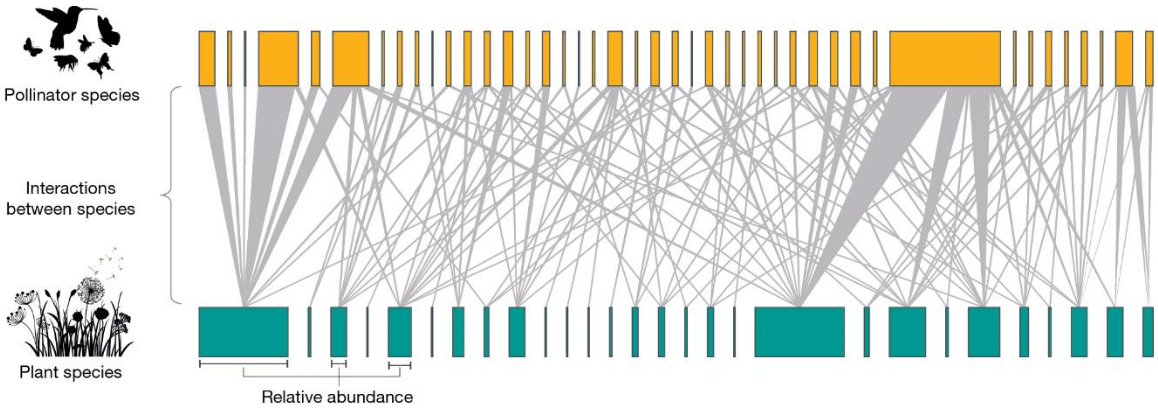
3.5 The structure of pollination networks

3.5.1 Outline of section

This section aims to give a brief general overview of trends in pollination networks and their metrics. It discusses how pollination networks change at small scales, with disturbances like grazing, the introduction of non-native plants or pollinators, and habitat fragmentation, and at global scales, with latitude. It also summarises what is known of how pollination networks change over time.

3.5.2 A brief introduction to pollinator networks

Pollination networks (or pollination webs) are bipartite networks of mutualistic interactions between pollinators and plants within a system, and the interactions between pollinators and plants are the links between the nodes. Pollination networks, therefore, contain information about which plant species and which pollinators interact, and how strong the interactions are (i.e., the relative abundance of interactions between a particular pollinator species with a particular plant species, Figure 3.8).



26. Figure 3.8. An example of a pollination network.
The lower line of (green) blocks represents different plant species, the upper (orange) line represents different pollinator species. The width of the blocks represents relative abundance of plants of pollinators. The lines connecting the flower visitors and flowers indicate the interactions between species. The width of the connections is proportional to the strength of the interaction.

Pollination networks have certain characteristics. They tend to be asymmetrical (i.e., specialists tend to interact with generalists, and if one species is heavily reliant on another, the species on which it depends is only weakly reliant on it; Vázquez and Aizen, 2004; Bascompte et al., 2006; Stang et al., 2007). They also tend to be highly nested (i.e., a core of generalist species interact, i.e. are directly involved, with each other, and the majority of specialists in the network only interact with these generalists; Bascompte et al., 2003). The nested and asymmetrical properties of pollination networks allow them to be more tolerant of species loss associated with habitat transformation and disturbance (Memmott et al., 2004; Bastolla et al., 2009). Nestedness has been shown to increase network resilience in mutualistic networks like plant-pollinator networks (Thébault and Fontaine, 2010). Despite the value of having many species in networks, it seems that it is the abundant species that deliver the bulk of pollination (Winfree et al., 2015).

The study of networks has generated a number of metrics for assessing networks, and Blüthgen (2010) provides a summary of these along with cautionary advice for their interpretation. If these cautions are heeded, network metrics could prove useful in biological conservation monitoring and assessment (Tylianakis et al., 2010; Kaiser-Bunbury and Blüthgen, 2015). It should be noted that there is a distinction between “flower visitation networks” and “pollination networks”, because not all flower visitors are effective pollinators (King et al., 2013), and observation of visitation alone often misses the full spectrum of plant species that pollinators have visited (Bosch et al., 2009). Most studies construct flower visitation networks, because they record which flower visitors were seen at which plant species, but not whether pollination takes place.

Although visitors need to transfer pollen for their visits to be effective, and pollinator visitation rate is not always correlated with pollinator effectiveness (Watts et al., 2012), there is a relationship between fruit set and the number of interactions (Vázquez et al., 2005; Garibaldi et al., 2013). Therefore, interaction frequencies are often used as a proxy for pollination, despite obvious limitations.

The study of how pollination and flower visitation networks change in space and time is still relatively new (Burkle and Alarcón, 2011), but provides useful insights. Network analyses can identify consequences of certain actions for the plant-flower visitor community, for example, the removal of non-indigenous species (e.g., Carvalheiro et al., 2008), livestock grazing (Vanbergen et al., 2014), or disturbance (Nielsen and Totland, 2014). Network analyses can be based on either field data (e.g., Bascompte et al., 2006) or modelled communities (e.g., Lever et al., 2014).

3.5.3 Spatial changes in flower visitation networks

Disturbances influence species composition and relative abundance in space and time, thus affecting flower visitation networks. There is general agreement that the nested and asymmetric nature of networks makes them relatively robust to disturbance (Memmott et al., 2004; Bascompte et al., 2006; Petanidou et al., 2008; Ramos-Jiliberto et al., 2009). Although species composition and interactions between species may change, metrics of network structure (e.g., connectance, i.e. the number of links between flower visitors and flowers out of the total number possible; and nestedness) tend to be conserved (Alarcón et al., 2008; Olesen et al., 2008; Petanidou et al., 2008; Dupont et al., 2009; Olesen et al., 2011; Dupont and Olesen, 2012), suggesting that flower visitation networks may be less vulnerable to negative effects of disturbance than previously thought (Muñoz et al., 2008). This is because species within networks often have the ability to form new connections (referred to as "rewiring") with new species in the absence of former partners (Ramos-Jiliberto et al., 2012), or might change from being specialists to being generalists (Petanidou et al., 2008). The new relationships formed when networks are disturbed may not be as efficient as those they replace, however, and seed or fruit set may therefore be reduced (e.g., Alarcón, 2010; Brosi and Briggs, 2013). Furthermore, recent research suggests that continued loss of species eventually leads to a threshold at which the network collapses (Lever et al., 2014).

Human disturbance of habitats can change productivity, changing plant and flower-visitor species composition, with implications for flower visitation networks. For example, in a desert environment in Israel, gardens containing a number of exotic plant species had more water available and greater bee abundance than the surrounding arid environment; bee species in gardens were more generalist than those found in the neighbouring desert, however, and despite there being fewer bees in the desert, bee species richness was greater (Gotlieb et al., 2011). A study over three years found that addition of nitrogen (N) to plants changed floral abundance and plant species composition, but not the abundance and species composition of flower visitors (Burkle and Irwin, 2009). Additional N also changed flower visitation rates, but the core generalist plant and pollinator species remained the same, and network parameters like degree of nestedness did not change (Burkle and Irwin, 2009).

A study conducted in France addressed the effects of urbanization on flower visitation networks. Urbanization was associated with fewer flower-visitor-plant interactions relative to semi-natural or agricultural environments, and smaller-bodied, specialist flower visitors were particularly negatively affected; also, reproductive success of focal plant species was lower in urban environments, as was network asymmetry (Geslin et al., 2014). The detrimental effect for specialist flower visitors and the decline in plant reproductive success as a consequence of reduced pollinator diversity in urban environments is perhaps expected, yet the negative impact on short-bodied species is not an obvious outcome. Large-bodied species require more resources, have larger home ranges and reproduce more slowly than small species and so should be more impacted by habitat fragmentation (Tscharntke et al., 2002). It may be that short-bodied insects with concomitantly small flight ranges might not thrive in urban environments, where spatial connectivity between floral resources is low relative to natural environments. Urban areas with gardens or parks that can provide nesting sites and forage may therefore not experience these losses to the same degree.

The effects of livestock grazing and pasture management on flower visitation networks have mostly been investigated in temperate systems. Intensively-managed meadows have been found to have lower floral and pollinator diversity, and fewer specialist pollinators (Weiner et al., 2011). In another study, where long-term grazing was investigated, flower-visitor networks in grazed plots were more species rich, but had become less nested, implying they are more vulnerable to loss of specialist species in future (Vanbergen et al., 2014). Elsewhere, grazing, and high stocking rates in particular, have been associated with declines in specialist plants or flower visitors (Vázquez and Simberloff, 2003; Yoshihara et al., 2008). Relatively few studies have investigated the effects of fragmentation and habitat loss on flower visitation networks. Models predict that at levels of habitat conversion of 50% or more, pollination networks experience extinctions (Keitt, 2009), although this threshold may be greater if crop lands that replace natural habitat can supply pollen and nesting sites for pollinators, although wild plant species will still be lost (Keitt, 2009). Field studies have found that habitat transformation can trigger non-random loss of interactions, with infrequent and specialized interactions being the most vulnerable (Aizen et al., 2012). Habitat loss has been associated with not only a loss of species but also increased connectance (i.e., overall generalisation) within networks (Spiesman and Inouye, 2013). A comparison of habitat fragments of different sizes (isolated hills in an agricultural matrix in the Argentinian pampas) found that the number of species and number of links between species (link diversity) both declined with decreasing fragment area, but that the rate of decline of link diversity was double that of species diversity, suggesting that interactions can be lost faster than species are lost (Sabatino et al., 2010). The generality of this finding is not known, however, but should be investigated as this would have implications for patch size and restoration efforts.

Invasive Alien Species (IAS) can be defined as "Animals, plants or other organisms introduced by man into places out of their natural range of distribution, where they become established and disperse, generating a negative impact on the local ecosystem and species" (IUCN, 2015). The majority of studies into the effects of invasive alien species (IAS, both pollinators and plants) find that IAS tend to be generalists (or even super-generalists; Aizen et al., 2008; Vilà et al., 2009). IAS become integrated into networks by forming links with native generalists (Memmott and Waser, 2002; Padrón et al., 2009; Kaiser-Bunbury et al., 2009). When IAS become integrated into networks, the links within the networks can change from generalist native species to super-generalist invasive species (Aizen et al., 2008), which can reduce connectivity among native species (Valdovinos et al., 2009; Sugiura and Taki, 2012; de M. Santos et al., 2012; Traveset et al., 2013; Ferrero et al., 2013; Grass et al., 2013). The effect of both plant and pollinator IAS becomes more marked with degree of invasion (Aizen et al., 2008; Kaiser-Bunbury et al., 2011). Oceanic islands may be an exception to the trends observed for alien invasion, with native endemic species being super-generalists, having responded to reduced competition to fill a variety of niches, and they may facilitate invasion by including invasive plants or pollinators into networks (Olesen et al., 2002). This seems to be supported by a study comparing oceanic and continental islands: there were more super-generalist species on oceanic islands, which ultimately yielded higher values for connectance and nestedness on oceanic islands, and which are therefore assumed to be more stable (Castro-Urgal and Traveset, 2014). A recent review comparing invaded and uninvaded networks showed that invaded networks were more resilient to simulated removal of either plants or pollinators, because invasive species, through their generalist nature, tend to connect more species within the network to each other; but when these connected IAS are removed, the network is more prone to collapse (Albrecht et al., 2014). As with all pollinator network studies, the question remains as to whether the observed interactions in invaded networks really do involve the transfer of pollen, and thus are beneficial to both the flower visitor and plant.

At global scales, patterns in specialisation within flower visitation networks as a function of latitude are not clear. Some studies have found specialisation to be greater in the tropics (Dalsgaard et al., 2011; Trøjelsgaard and Olesen, 2013), or for plants but not their flower visitors (Olesen and Jordano, 2002). The opposite trend, of increasing specialisation away from the tropics has also been found (Schleuning et al., 2012); and other studies have found no relationship between specialisation and latitude (Ollerton and Cranmer, 2002). The patterns are likely driven by both ecological and evolutionary processes, and resolution to this debate will likely be obtained as additional datasets from more regions across the globe become available and analyses become more refined.

3.5.4 Changes in flower visitation networks with time

The majority of studies on networks are essentially “snapshots” in time, but those studies that do consider longer periods (Alarcón et al., 2008; Petanidou et al., 2008) find that species may be less specialised than they initially appear. Flower visitor species and their relative abundance can vary significantly across years and within seasons (Price et al., 2005; Basilio et al., 2006; Dupont et al., 2009; Burkle and Irwin, 2009; Albrecht et al., 2010). The contrast can be striking; for example, Petanidou et al. (2008) found that more than 90% of species that seemed to be specialists in one year behaved as generalists in later years. These changes are driven primarily by variations in both plant and flower visitor species composition and abundance, associated with the normal cycle of succession, and secondarily by flexibility in flower visitors’ behaviour; flower visitors may change the species they visit (“rewiring” the network), depending on interspecific competition or flower availability (e.g., Fontaine et al., 2008; Brosi and Briggs, 2013). Even at short time scales, succession in flower visitation networks can be associated with considerable variation, and in some systems, the species composition of the network can change daily (Olesen et al., 2008). Current networks are also influenced by the distant past: for example, historical climate change (i.e., in the Quaternary) has been found to influence network structure on continents but not on islands (Dalsgaard et al., 2013). All of this suggests that flower visitation networks are highly dynamic and highlights the importance of viewing networks over appropriate time scales (Burkle and Alarcón, 2011).

Notwithstanding the ability to rewire, systems may not be exempt from losses. A study considering changes in plant-pollinator interactions over 120 years in Illinois, USA, found only 50% of the bee species originally recorded in the late 1800s by Charles Robertson (1928); although novel interactions had arisen, the total number of interactions in the remaining network was only half that of the original (Burkle et al., 2013). In addition, the overall network structure had become less nested, indicating that pollination is more vulnerable to future perturbations (Burkle et al., 2013). Notably, new partnerships formed in rewiring may be less effective at ensuring seed set than previous relationships (Alarcón, 2010; Brosi and Briggs, 2013).

Climate change is likely to see changes in rainfall distribution and amount, wind patterns, and temperature; in addition, the occurrence of extreme weather events is likely to increase in frequency with climate change (IPCC, 2013). These in turn will affect the phenology, distributions and relative abundances of plants and their pollinators. One of the first studies to consider the implications of changes in phenology for flower visitation networks predicted disruptions in food supply for between 17-50% of flower visitors, with specialised species most vulnerable (Memmott et al., 2007). The consequences of these findings may not be as detrimental as initially thought, if species that appear to be specialists can indeed switch to alternative forage sources. We have little idea of how climate change might affect flower visitor life stages, however, nor do we know the effects of extreme weather events (e.g., droughts) on flower visitor and plant population dynamics and networks (Memmott et al., 2007). There can be selection for various phenological responses to climate change in plants, with the net effect that some species advance their flowering time, or change the time of peak flowering or length of flowering period, whilst others show no such changes (Rafferty and Ives, 2011; CaraDonna et al., 2014). Future studies on the effects of climate change on pollination-networks will need to consider the plastic responses of both plants and flower visitors, the constraints on each, as well as the efficiencies of new-formed relationships, against a background of environmental stochasticity.

The above examples demonstrate the value of network analysis in providing insight into plant-pollinator interactions. They are particularly useful, once baseline data have been recorded, in allowing investigation of the consequences of future changes in flowers or their pollinators, either through modelling or by future repeat sampling. Another promising field of study involves merging functional approaches with pollination networks to assess the vulnerability of pollination to disturbance. In this approach, species traits (e.g., proboscis length, body size or flower size) are mapped onto the networks, which can allow prediction of how certain disturbances will affect the representation of life history traits within pollination networks, and therefore allow predictions on how disturbances might affect delivery of ecosystem services (Ibanez, 2012; Díaz et al., 2013; Lavorel et al., 2013; Astegiano et al., 2015). For example, loss of pollinators with a certain proboscis length could affect the likelihood of certain plant species being successfully pollinated (Ibanez, 2012). Another example might be the loss of birds that pollinate economically important plant species, which can be predicted by the bird species' traits that make them vulnerable to loss, like low tolerance to hunting.

In summary, changes in pollination networks in response to various disturbances can yield useful information for management. Networks also can be used to predict the effects of various disturbances, e.g., removal or arrival of non-native species (Carvalheiro et al., 2008; Albrecht et al., 2014), or losses associated with climate change (Memmott et al., 2007). There is also a need for more long-term studies, given that species seem to be able to change their roles in networks, changing the species with which they interact if their partners are no longer present (Inouye, 1978; Alarcón et al., 2008; Petanidou et al., 2008). It would also be important to establish whether the new partnerships formed in the “rewiring” of networks are as efficient as previous partnerships (e.g., Brosi and Briggs, 2013), or whether generalists are more likely to form new associations than specialists. The use of quantitative over qualitative networks (i.e., networks that consider the number of interactions, not just which species interact with which) will also help to identify the importance of certain interactions over others.

Key to much research on networks is knowledge of how many of the interactions in pollination networks are beneficial to both the flower visitor and the plant. Networks will provide more insight when the impact of the interaction (pollen deposited, reward collected, or seed set) is included (Bosch et al., 2009; Tur et al., 2014), rather than just which flower visitors visit which plants (see, e.g., Alarcón, 2010). Finally, networks might also be important in identifying tipping points (Lever et al., 2014) – points at which the system switches to another stable state, from which it may be difficult or impossible to return.

3.6 Wild plant pollination and reproductive success

3.6.1 Outline of section

In this section we explain how a decline in pollinators and pollination can be linked to plant population decline and extinction. We summarize evidence for a decline in seed production linked to pollinator decline, and highlight the few studies that show that pollinator loss can culminate in a decline in plant abundance. Even without a decline in plant abundance, pollinator loss has the potential to erode plant genetic diversity and alter the trajectory of plant evolution.

3.6.2 Concepts

Earlier in this chapter we examined trends in the abundance and diversity of pollinating animals. A logical next step is to ask whether the trends in pollinator abundance and diversity are linked to trends in plant reproduction. Ultimately, we may ask: does pollinator abundance and diversity affect plant population growth rate? Growth rates below zero will lead to population extirpation; growth rates above zero allow population persistence.

The life cycle of an animal-pollinated plant is a sequence of events starting with the arrival of the pollinator and ending with the flowering of the next generation. For pollinator decline to matter for plant population persistence, it must translate into changes in pollination rate, pollen receipt, fertilisation, seed set, the number of seedlings produced, and ultimately the rate of establishment of new plants. At any step in this procession, initial effects may fail to be transmitted and pollinator loss will then not cause a decline in plant abundance. For example, a decline in the abundance of a particular pollinator may not cause a decline in pollination rate, if the lost function is replaced by a competing pollinator (Smith et al., 1995). A decline in pollination rate may not cause a decline in seed set if plants receive an excess of visits or are self-pollinating, and a decline in seed production may not cause a marked reduction in plant population growth rate if the plant species can reproduce vegetatively or is long-lived (Pauw and Bond, 2011). Plant species that are likely to respond rapidly to pollinator decline are those that require specialist pollinators, need cross-pollination for seed production, and need high rates of seed production to compensate for high rates of adult mortality. A self-incompatible annual plant that is a pollination specialist is an example of a species that can be expected to be vulnerable to pollinator decline in the short term (Bond, 1994). From the discussion above, it follows that simply detecting low pollination rates or low levels of seed set is insufficient evidence of a risk of population decline, and may not be a cause for concern. Plant species vary in how much pollination or seed set they need for population persistence.

Two useful concepts are "pollen limitation" and "seed limitation". Pollen limitation occurs when plants produce fewer fruits and/or seeds than they would with adequate pollen receipt, and is measured by comparing reproductive success in unmanipulated plants with that of plants given supplemental pollen (Knight et al., 2005). Seed limitation occurs when population size is smaller than it would be with adequate seed production, and is measured by comparing recruitment in control populations with recruitment in populations to which seed has been added (Turnbull et al., 2000). Sixty-three percent of the 482 data records on percent fruit set showed significant pollen limitation (Knight et al., 2005), and approximately 50% of 90 tested species show evidence of seed limitation (Turnbull et al., 2000).

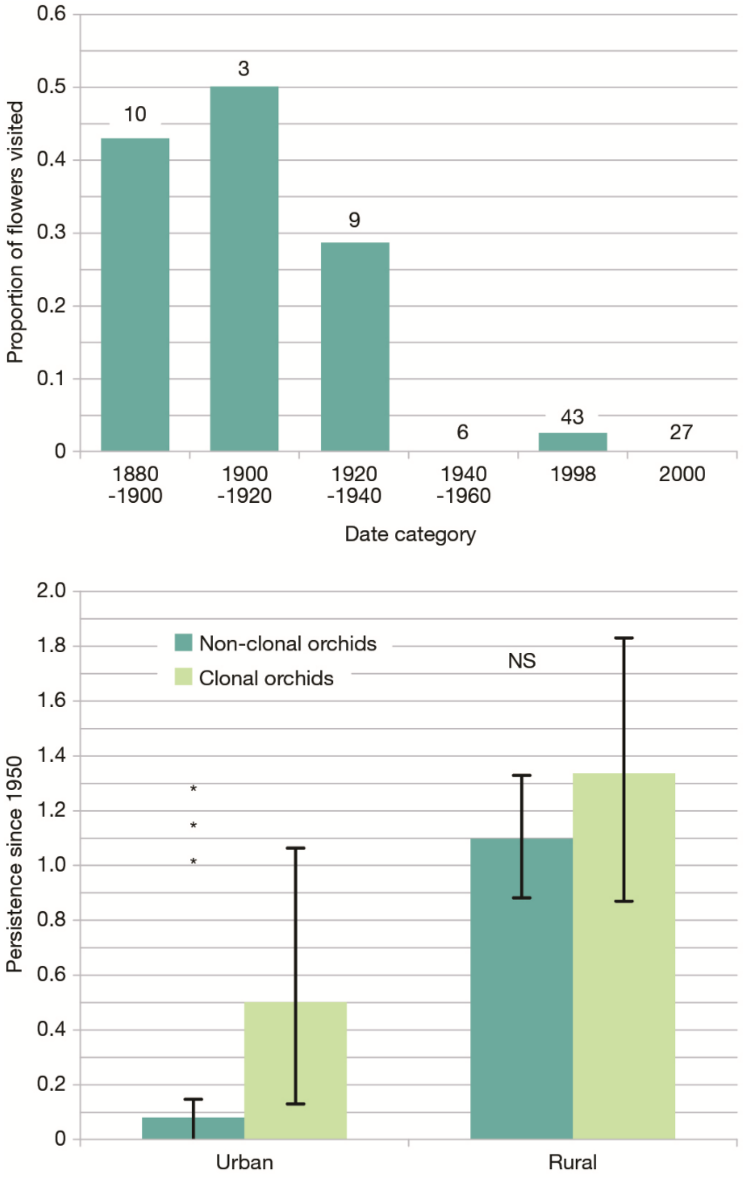
The demographic perspective on the importance of pollinators for plants above can be contrasted with a genetic perspective. In the absence of pollinators, the replacement of outcrossing by self-fertilization or vegetative reproduction may rescue plant populations by ensuring recruitment, but will lead to the loss of genetic diversity. Plant populations need genetic diversity in order to respond to long-term changes such as climate change and to allow for future natural evolution. Thus, situations where there is a decline in outcrossing rate, but no decline in plant reproduction, are also a cause for concern.

3.6.3 Status and trends

The vast majority of wild plant species are dependent on insect pollination for fruit and seed set, with the proportion of animal-pollinated plants rising from a mean of 78% in temperate-zone communities to 94% in tropical communities (Ollerton et al., 2011). In addition, many (62–73%) of the plant populations thus far investigated show pollen limitation, i.e. changes in the abundance and diversity of pollinators are likely to affect their seed production (Burd, 1994; Ashman et al., 2004, Wolowski et al., 2013). Nevertheless, detecting historical trends in plant reproduction is no easy task given the paucity of long-term studies.

Historical species distribution records are a potential source of information. Analysis of such data found a decline in bee species diversity in Britain and the Netherlands and that outcrossing plant species that rely on pollinators have shown corresponding declines, when compared to self-compatible or wind-pollinated plants (Biesmeijer et al., 2006). Of course, correlations such as these do not prove causality or directionality (Biesmeijer et al., 2006) but the strong correlation makes sense intuitively and reintroduction of locally extinct pollinator species might provide experimental confirmation.

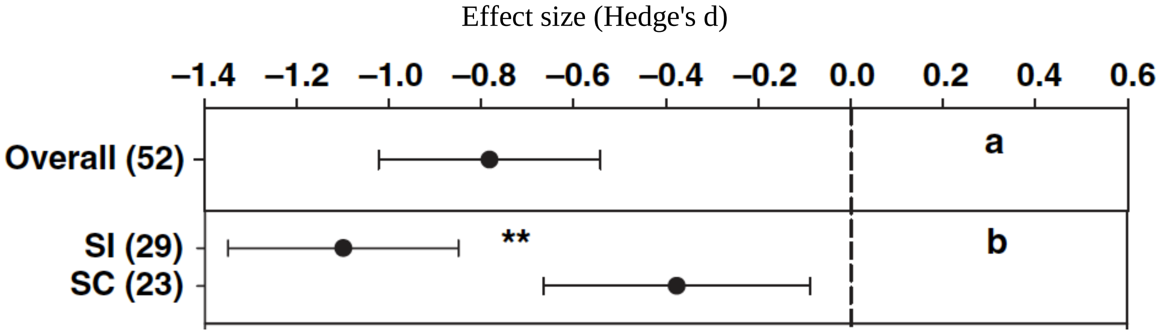
Direct comparisons with historical pollination rates are rare. In one study, century-old herbarium specimens were rehydrated and examined for evidence of pollination. The historical pollination rates were found to be many times higher than current rates from the same location. There was a contemporaneous shift in plant community composition at the site due to the local extirpation of species that were unable to reproduce vegetatively, consistent with their greater dependence on seeds and pollination for population persistence (Pauw and Hawkins, 2011) (Figure 3.9). While the ability of certain plant species to persist into the medium term without pollinators is good news, it can also be seen as a temporary relief, or an extinction debt, which we will pay in the long-term when the failure of seed production finally causes population decline, or the loss of genetic diversity.



27. Figure 3.9. (a) Reconstruction of historical pollination rates from herbarium specimens of *Pterygodium catholicum* collected on Signal Hill, South Africa.
Pressed herbarium specimens contain a record of past pollinator activity in the form of pollinarium removal rates. Sample sizes are above bars. (b) Following the human-caused loss of the pollinator, an oil-collecting bee, the orchid assemblages shifted in favour of greater representation by clonally reproducing species in urban areas, while no such shift occurred in rural areas where the pollinator still occurs. Persistence of 1 indicates that the number of pre- and post-1950 herbarium records is equal. Figure reproduced from Pauw and Hawkins (2011), with permission from John Wiley and Sons.

In most cases, however, historical base-line data are lacking and researchers use space-for-time substitution, i.e. they compare human-altered areas with natural areas, assuming that the human-altered areas historically resembled the natural areas (Aizen and Feinsinger, 1994; Murren, 2002; Steffan-Dewenter et al., 2002; Pauw, 2007). In these studies, it is important to account as far as possible for "natural" spatial variation.

Island-mainland comparisons are the context in which spatial patterns of variation in pollination rate and plant reproduction are most often studied. Small fragments of natural habitat that remain in an agricultural or urban matrix are considered equivalent to habitat 'islands'. Mainlands are larger natural areas nearby. In a meta-analysis of 89 plant species from 53 published articles, Aguilar et al. (2006) found a large and negative effect of fragmentation on pollination and on plant reproduction. The compatibility system of plants, which reflects the degree of dependence on pollinator mutualism, was the only reproductive trait that explained the differences among the species effect sizes. Furthermore, a strong correlation between fragmentation (measured as effect size, see Figure 3.10), pollination and reproductive success suggests that the most likely cause of reproductive impairment in fragmented habitats may be pollination limitation. Thus, this study clearly links a decline in habitat area to a decline in pollination and secondarily to a decline in fruit and seed set (Figure 3.10).



28. Figure 3.10. Weighted-mean effect sizes and 95% bias-corrected confidence intervals of habitat fragmentation on pollination for 52 plant species
(a), and categorized by their compatibility systems (b). The effect size can be interpreted as the difference between the reproductive responses of plants in fragmented habitats versus continuous habitats, measured in units of standard deviations. SI = self-incompatible; SC = self-compatible. Reproduced from: Aguilar et al. (2006).

Land use at the landscape scale can also impact the composition of plant communities indirectly, via pollinators. Clough et al.’s (2014) study of grassland plant and pollinator data from Europe showed that relative pollinator dependence among grassland plants is lower in landscapes with little pollinator-friendly habitat in the surrounding landscape, and consequently fewer pollinators.

Exotic invasive plant species can alter pollinator visitation and, in turn, the sexual reproduction of natives (see Chapter 2). While there is no question that there is a trend of increasing invasive species globally, most studies have used space-for-time substitution to study the effects by comparing invaded with uninvaded areas nearby. Using a meta-analytical approach on a data set of 40 studies, Morales and Traveset (2009) evaluated the effect of invasive plant species on pollinator visitation and reproduction of native co-flowering species. An overall significantly negative effect of invasive species on visitation and reproduction of native plants was detected ($p<0.05$).

A more recent study jointly analyzed the effect of habitat alteration and alien invasion on plant reproduction through a meta-analysis of 58 publications reporting 143 studies (Montero-Castaño and Vila, 2012). Habitat alteration and invasions affected pollinators to the same magnitude by decreasing visitation rates. Visitation rates by vertebrates in altered landscapes and by insects (excluding bees) in invaded areas were the most affected. The result for insects concurs with an earlier study, which showed that, although significant, the magnitude of the effect of habitat fragmentation on bees is not large, possibly because many bee species prefer open habitats (Winfree et al., 2009). It must be noted however, that this analysis included many studies in which the level of habitat fragmentation was moderate (i.e. habitat mosaics), and findings may differ for situations of extreme fragmentation (e.g. Pauw, 2007). The result for vertebrates concurs with Aslan et al. (2013) who estimated that globally 16.5% of vertebrate pollinators (192 species) are threatened with extinction, which is worrying when we consider that an estimated 16,800 plant species are vertebrate-pollinated (Aslan et al., 2013). Threat levels are particularly high for island-based species; the authors estimated that 30.4% of island-based vertebrate pollinators are threatened, with important consequences for island plant reproduction. For example, functional extinction of bird pollinators reduced pollination, seed production, and plant density in the shrub *Rhabdothamnus solandri* (Gesneriaceae) on the North Island of New Zealand but not on three nearby island bird sanctuaries where birds remain abundant (Anderson et al., 2011). The study is one of few which show that the effect of pollinator loss is perpetuated through all the stages of plant reproduction to culminate ultimately in a decline in plant abundance (see Pattemore and Anderson 2013 for a related study). The study contrasts with cases where, introduced species of pollinators are able to replace extinct species (Lammers et al., 1987; Cox, 1983).

Pollinator-driven changes in plant reproduction need to be seen in the context of other pressures experienced by plant populations. In the world of the future, plants may need more seeds; human-driven increases in plant mortality rates due to an increase in fire frequency (van Wilgen, 1981), logging, harvesting or grazing should logically necessitate an increase in seed production if birth rates are to balance death rates (Crawley, 1990). Hence, a pollination deficit may occur without pollinator decline. The predicted droughts from climate change in some parts of the world may also affect pollination patterns, through effects on plants (e.g., nectar and production), pollinators (access to floral resources), or both. Phenological mismatches between plants and pollinators are also likely to become increasingly common (Thomson, 2010; McKinney et al., 2012).

In a long-term view we also need to consider the evolutionary future of a world with an altered pollinator environment (Guimarães et al., 2011). The features of flowers, their scents and colours, are the result of natural selection imposed by pollinators. Faced with increasing pollen limitation, plants may either come under selection to enhance attractiveness, or alternatively to enhance self-pollination (Cox, 1991; Fishman and Willis, 2008; Mitchell and Ashman, 2008; Harder and Aizen, 2010). The latter trajectory is expected to lead to smaller and less attractive flowers, as shown experimentally by Bodbyl-Roels and Kelly (2011). Evidence for such a trend comes from a study of urban versus rural populations of a Japanese *Commelina* species, which display traits that promote self-pollination only in an urban context (Ushimaru et al., 2014). Animal traits may also evolve in response to human-induced changes in the architecture of plant-pollinator interaction networks. For example, Smith et al. (1995) detected an evolutionary change in bill size in the Hawaiian honeycreeper (*Vestiaria coccinea)* resulting from an apparent dietary shift caused by dramatic anthropogenic declines and extinctions of lobelioids, a historically favoured nectar source.

3.7 Agricultural pollinator dependence

3.7.1 Outline of section

This section reviews the dependence of crops and global agriculture on animal pollination, trends of increased pollinator-dependency of agriculture over time, and spatial-temporal variation among among regions in the world. Also, it discusses potential uncertainty associated with the use of FAO data and crop categories of pollinator dependency.

3.7.2 Crop and agriculture pollinator dependency

Animal pollination is critical for, or enhances the reproduction of, many cultivated crops. Some estimates have shown that pollinators (mainly, but not exclusively bees) increase the productivity of ca. 70% of 1,330 tropical crops (Roubik, 1995), 85% of 264 crops cultivated in Europe (Williams, 1994), and about 70% of the world’s 87 leading crops (Klein et al., 2007). Given that pollinator dependence for increasing yield is highly common, there have been breeding programs to make some crops less dependent on animal pollination. For instance, inbred, pollinator-independent varieties of some crops, like tomato (*Solanum lycopersicum*) have been artificially selected (Peralta and Spooner, 2007). Also, self-compatible cultivars of almond, *Prunus amygdalus*, have been developed from crosses between self-incompatible varieties (e.g., Kodad and Socias I Company, 2008), whereas gynoecious (i.e. female) lines of parthenocarpic cucumbers (*Cucumis sativus*) have been obtained through controlled crosses between parents carrying this partially recessive, genetic-based trait (Yan et al., 2008). On the other hand, many entomophilous crops, like sunflower (*Helianthus annuus*) rely on the sowing of commercially-produced hybrid seed harvested on male-sterile plants, a process for which insect pollination is absolutely essential (Perez-Prat and van Lookeren Campagne, 2002). Also, some outcrossing crop species maintained as populations, such as alfalfa and white clover, will become increasingly less productive without abundant and effective pollinators because of increasing inbreeding depression (Jones and Bingham, 1995). Even self-compatible crops that have been highly genetically engineered, like rapeseed (*Brassica napus*), can be largely pollinator-dependent (Morandin and Winston, 2005), the same as largely parthenocarpic crops, like seedless varieties of *Citrus* (Chacoff and Aizen, 2007) or triploid seedless watermelon (Walters, 2005). Because of these opposing examples, there seems not to be a net trend for agriculture to become less pollinator-dependent through crop breeding.

Because there is wide variation among crops and varieties within crops in their degree of pollinator dependency (Klein et al., 2007), the question that follows is not how dependent are individual crops, but rather how dependent is global agriculture on animal pollination. Overall, animal-pollinated crops represent about one-third of global agricultural production volume (i.e., metric tons), but because of only partial pollinator-dependence of those crops (Richards, 2001; Klein et al., 2007), pollinators only account for 5-8% of total production (Aizen et al., 2009). These latter figures are minimum estimates, however, because they only consider the direct role of pollinators in producing the seeds and fruits we consume in terms of weight, but not (i) the indirect role of pollinators in producing the seeds of many vegetable or fibre crops we sow (Klein et al., 2007); (ii) pollinators´ contribution to food quality in terms of the disproportionate concentration of micronutrients, including many vitamins, contained in different organs of animal-pollinated plants (Eilers et al., 2011, Delaplane et al., 2013), particularly in tropical regions (Chaplin-Kramer et al., 2014); (iii) pollinators´ relevance in the pollination of fodder crops and pasture (Fairey et al., 1998); (iv) pollinators´ importance in the production of non-timber forest products (Rehel et al., 2009); and (v) pollinators´ role in the pollination of medicinal plants and plant species of traditional use (Joy et al., 2001). In addition, because of the low yield of many pollinator-dependent crops (compared to non-dependent crops), the relatively small direct production deficit caused by complete pollinator loss would need to be compensated by expanding global agriculture area by about 30-40% (Aizen et al., 2009). Thus, although any potential pollinator decline might not affect food production substantially, it will potentially entail a high environmental cost in terms of natural and semi-natural habitat destruction associated with the compensatory expansion of agricultural land as well as pose other land-use conflicts (Chapter 2).

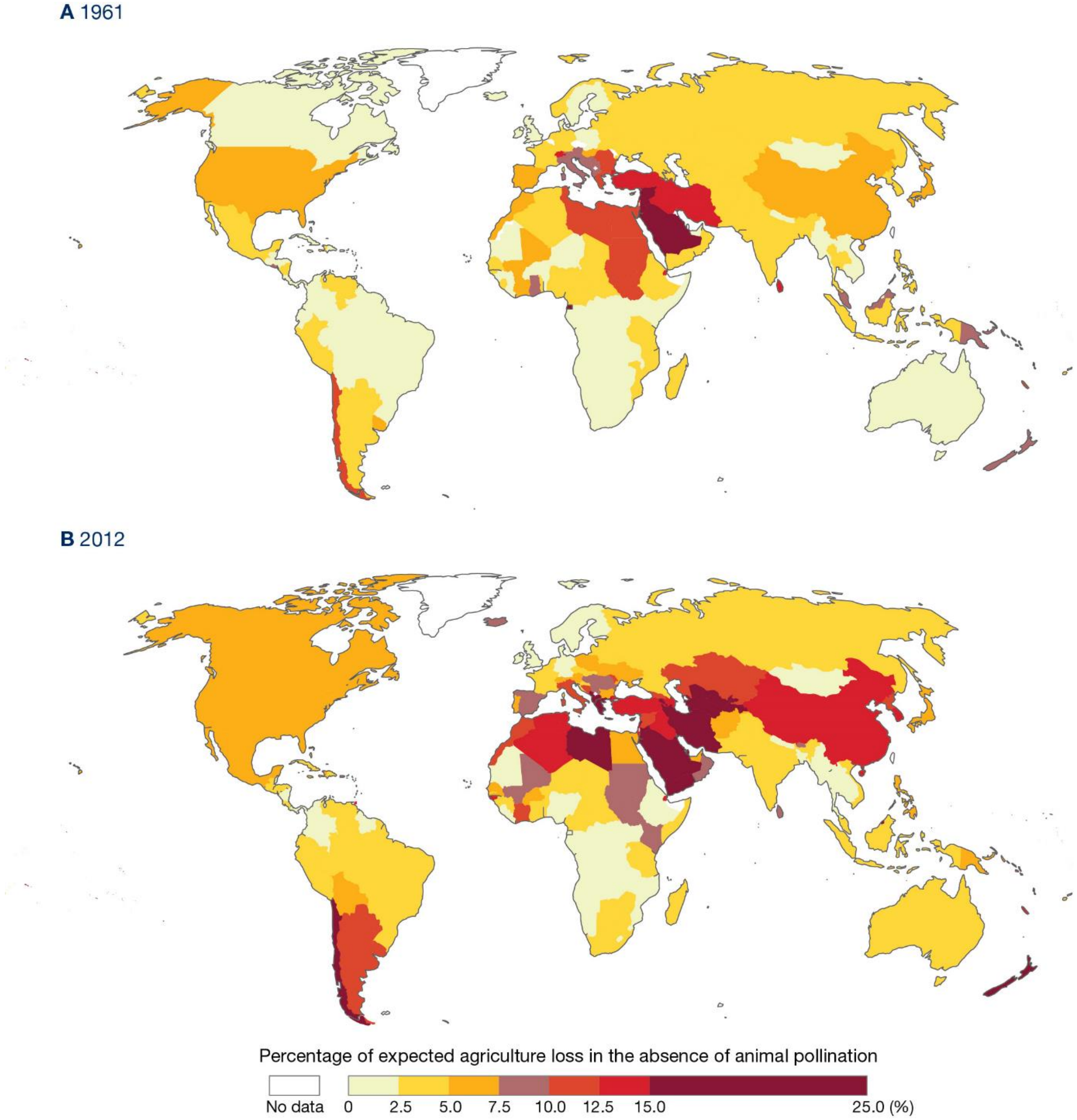
3.7.3 Spatial and temporal trends in agricultural pollinator dependency

Although pollinators are directly involved in the production of a small fraction of our food supply in terms of weight (Aizen et al., 2009a, c), global agriculture has become more pollinator-dependent over the last five decades. Pollinator-dependent production has increased >300% in absolute terms, whereas the much larger non-dependent fraction has less than doubled over this period (Aizen and Harder, 2009a). This increase in pollinator-dependency of agriculture has been steeper in developing countries in Africa, Asia and Latin America than, with some exceptions (e.g., Canada), in developed countries in North America, Europe, Australia and New Zealand. Mediterranean and Middle East countries have had traditionally a highly pollinator-dependent agriculture associated with the cultivation of a large variety of temperate and subtropical fruit and seed crops. However, rapid expansion of many of these crops in other countries (e.g., China) and cultivation of some genetically-engineered and moderately pollinator-dependent crops, like soybean (e.g., Argentina, Paraguay, Uruguay, and Bolivia) and rapeseed (Canada) (Lautenbach et al., 2012), are responsible for the large increase in the pollinator dependency of global agriculture in terms of production between 1961 and 2012 (Figure 3.11, Table 3.2). In turn, this increase in the production of pollinator-dependent crops account for most of the ca. 30% expansion of global agricultural land that occurred during this period (Aizen et al., 2008; Garibaldi et al., 2011b). In 2006, pollinator-dependent crops comprised 33% of developing country and 35% of developed country cropped land area (Aizen et al., 2008). This areal expansion has been basically concentrated in the Developing World, where the cultivation of pollination-dependent crops proceeded at a faster pace than the cultivation of nondependent crops. On the other hand, the total cultivated area changed very little in the Developed World, where the increase in the cultivation of pollination-dependent crops was compensated by a proportional decrease in the cultivation of non-dependent crops, (Figure 3.12; Aizen et al., 2008, 2009). Furthermore, many pollinator-dependent crops are rich in micronutrients critical for human health (e.g., vitamin A, iron, and folate), and thus their production can be directly linked to animal pollination (Chaplin-Kramer et al., 2014). For instance, pumpkin, melon and mango are among the top crops for vitamin A production. Human deficiency of one or more of these micronutrients is most severe in regions of the Developing World, where their production depends the most on pollinators (Chaplin-Kramer et al., 2014). Hence, because of habitat destruction associated with agriculture expansion, pollination provided by wild insects might be more compromised in those regions where they are needed the most.

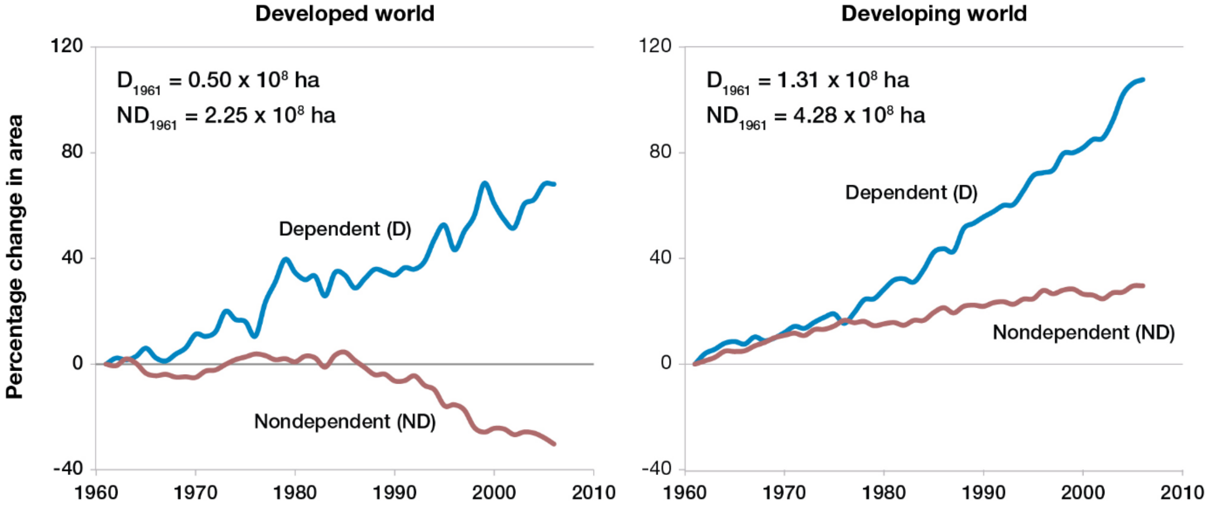
The reported global trend of agriculture increasingly being dependent on animal pollination (Figure 3.12) is most likely driven by socio-economic and political factors. Specifically, these factors involve increasing diversification in the human diet and consumption of high-value crops (Pelto and Pelto, 1983; Gallai et al., 2009), as well as globalization in food trade due to the adoption of market policies by most former Soviet Bloc countries and China after the fall of the Berlin Wall in 1989 (Aizen and Harder, 2009a). This trend might be exacerbated in the future as some rapidly-expanding, insect-pollinated crops (e.g., oil palm and rapeseed) are cultivated for large-scale biofuel production (Somerville, 2007).

3.7.4 Sources of uncertainty associated with FAO data and pollinator dependency

Sources of uncertainty for the section "Trends in agricultural pollinator dependence" and the next section "Trends in crop pollination and yield" are basically related to the accuracy of FAO agriculture data and the degree of pollinator dependence for the different crops (see Klein et al., 2007). Accuracy of FAO data most likely differs among countries and crops. However, under-reporting or over-reporting of agricultural data among countries and crops should be considered as a random error source and should not modify interpretation of temporal and spatial trends to the extent that these biases are not consistent among countries, crops, or years. Even consistent biases would affect neither the shape of the temporal trends nor estimates of pollinator dependency at the country level to the extent that biases do not differ between pollinator-dependent and non-dependent crops. The other component of the estimation of agriculture pollinator dependency, i.e., the extent to which crop yield depends on pollinators, is also subjected to great uncertainty. Crops can be solely classified in broadly-defined categories of pollinator dependency because information is highly fragmentary, anecdotal or scarce for many crops, for many present and past varieties within crops, because the degree of pollinator dependence can be highly variable among and within varieties of a single crop (McGregor, 1976; Klein et al., 2007). Therefore, for some major crops, such as soybean and sunflower, their pollinator dependence status should be viewed as a consensus status for the whole crop based on existing published information. In any case, this component of uncertainty should have a minor role in global or country-level estimations of pollinator dependency because of the weighted average of many individual observations (i.e., crops).



29**. Figure 3.11. World map showing agriculture dependence on pollinators (i.e., the percentage of expected agriculture loss in the absence of animal pollination, categories depicted in the coloured bar) in 1961 and 2012 based on FAO dataset (FAOSTAT, 2013) and following the methodology of Aizen et al. (2009).**



30. Figure 3.12. Temporal (1961-2006) trends in cultivated areas.
Shown is the aggregate cultivated area of all pollinator-dependent and non-dependent crops grown in the Developed and Developing world relative to their respective 1961 values (i.e., Δ area). Absolute area values for 1961 are provided (modified from Aizen et al., 2008).

13. Table 3.2. Pollinator dependency, and world production and global cultivated area (2012) of 16 major crops based on FAO dataset
(FAOSTAT 2013). We also provide estimates of annual growth rates in production and cultivated area (1961-2012). Pollinator dependency categories followed Klein et al. (2007).

Category/Crop	Scientific name	Pollinator dependency	Production (millions of tonnes)	Annual growth in production (%/year)	Cultivated area (millions of hectares)	Annual growth in area (%/year)
Vegetables and tubers						
Potatoes	*Solanum tuberosum*	None	365.37	0.59	19.28	-0.27
Sugar beet	*Beta vulgaris*	None	269.83	1.02	4.91	-0.67
Cassava	*Manihot esculenta*	None	269.13	2.64	2.08	1.52
Tomatoes	*Lycopersicon esculentum*	Little	161.79	3.53	4.80	2.08

Cucumbers and gherkins	*Cucumis sativus*	Great	65.13	3.84	2.11	1.45
Cereals						
Maize	*Zea mais*	None	872.79	2.88	178.55	1.04
Rice	*Oryza* ssp. (mainly *O. sativa*)	None	738.19	2.44	162.32	0.67
Wheat	*Triticum* spp. (mainly T. *aestivum, T. durum, T. spelta*)	None	671.50	2.19	217.32	0.12
Barley	*Hordeum disticum*	None	133.51	1.21	49.57	-0.19
Biofuels and oilseeds						
Sugar cane	*Saccharum officinarum*	None	1842.26	2.81	26.09	2.13
Oilpalm (fruit)	*Elaeis guineensis*	Little	259.42	5.95	17.57	3.15
Soybeans	*Glycine max, G. soja*	Modest	241.14	4.40	104.92	2.95
Rapeseed	*Brassica rapa, B. napus oleifera*	Modest	64.56	5.83	34.10	3.37
Fresh fruit						
Watermelon	*Citrullus lanatus*	Essential	105.37	3.54	3.47	1.13

Apples	*Malus domestica*	Great	76.38	2.98	4.84	2.05
Oranges	*Citrus sinensis*	Little	68.22	2.89	3.82	2.19
Mangoes, mangosteens, guavas	*Mangifera indica, Garcinia mangostana, Psidium guajava*	Great	42.14	2.69	5.17	2.77

3.8 Trends in crop pollination and yield

3.8.1 Outline of section

Deficits in pollination quantity and/or quality often limit crop yield. This section reviews and discusses the relevance of pollinator diversity to narrow pollination deficits, and whether crop pollination deficits have increased along spatial disturbance gradients and over time. It also briefly discusses the impact on agriculture of bees that were introduced outside their native range and have become invasive.

3.8.2 Crop pollination deficits

In pollination, pollen can be insufficient in quantity, e.g. stigmas receive too few pollen grains, or quality, e.g. stigmas receive pollen with low vigour due to genetic (i.e., self or self-incompatible pollen) or environmental factors (e.g., pollen produced by water-stressed or defoliated plants). Both may restrict wild plant reproduction and crop yield (Knight et al., 2005; Chapter 1). Technically, a crop pollination deficit refers to quantitative or qualitative inadequate pollen receipt that limits agricultural output (Vaissière et al., 2011). Even though pollination commonly limits seed production, decreases in pollinator diversity and visitation by effective pollinators may exacerbate chronic pollination deficits experienced by many crops.

A recent worldwide meta-analysis including data for 41 crops grown in 600 cultivated fields distributed across all continents, except Antarctica, reveals that diverse assemblages of wild bees seem to be important to reduce pollination deficits and sustain high yields of many pollinator-dependent crops (Garibaldi et al., 2013). Specifically, this study found that flower visitation by wild bees increases crop fruit and seed set, on average, twice as much as visitation by the domesticated honey bee, *Apis mellifera*, on a per-visit basis. Furthermore, declining pollination provided by wild bees might not be substituted by stocking fields with more honey bee hives, although honey bees can add to the pollination provided by wild bees (Garibaldi et al., 2013). Whereas complementary pollinating activity between wild bees and honey bees can explain this overall additive effect, diverse pollinator assemblages ensure the inclusion of one or more species of efficient pollinators (see also Chapter 1). For instance, yield of marketable French bean production in the Mt Kenya region was found to be positively correlated with the abundance of carpenter bees (*Xylocopa* spp.), despite high abundance of honey bees (Masiga et al., 2014). On the other hand, the risks of relying on a single pollinator species for large-scale crop pollination are exemplified by almond (*Prunus dulcis*) in the US, the country with probably the largest pollination industry. In this nation, more than two million honey-bee colonies are rented and even moved across the country to pollinate crops (Morse and Calderone, 2000). In fact, approximately 1.7 million hives are transported to California for almond pollination during a six-week period (Sumner and Boriss, 2006).

Besides questioning the efficiency of honey bees in pollinating almond flowers compared to wild pollinators (Klein et al., 2012), the continuous drop of the stock of honey-bee hives in the US during the last decades (National Research Council, 2007) questions the rationality and sustainability of such a practice. Furthermore, at a global scale the growth of the stock of domesticated honey-bees hives have proceeded at a much lower rate than demands for pollination (Aizen and Harder, 2009a), stressing the importance of wild pollinators for the productivity of many pollinator-dependent crops (Breeze et al., 2011). Similarly, in some regions of several Asian countries people have resorted to hand pollination of apple following declines in native apple pollinators and unavailability of managed honey bees to perform this function (Partap and Partap, 2007).

A recent global analysis (Kleijn et al., 2015), which includes data from 20 pollinator-dependent crops in about 1400 crop fields, proposes that the contribution of wild bees to crop production is limited to a subset of bee species that are common in agroecosystems. It seems likely that (i) crop pollination deficits are common and (ii) enhanced and sustained yields of many crops can be better ensured by both promoting specific pollinator species and the maintenance and restoration of diverse pollinator communities.

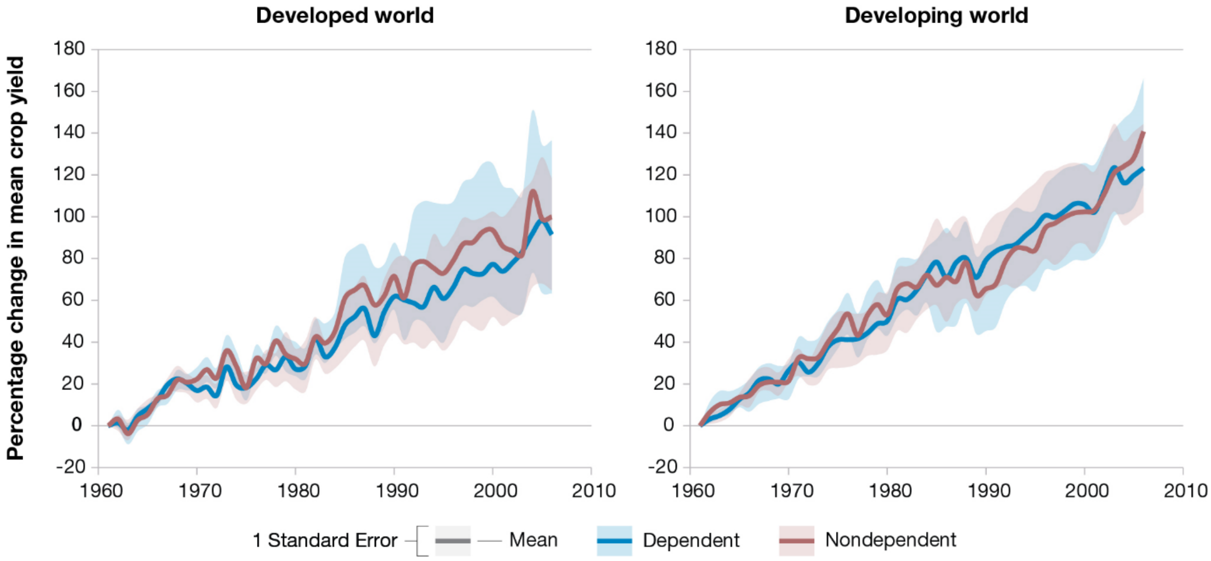
3.8.3 Spatial and temporal trends in pollination deficits

Remnants of natural and semi-natural habitats, hedgerows, and field margins, which supply essential flowering and nesting resources, can become important pollinator sources in different agro-ecosystems (Winfree et al., 2009; Morandin and Kremen, 2013; see Chapter 2). Therefore, increasing distance from field edges into crop fields greatly reduces flower visitation and the number of visiting species (Ricketts et al., 2008; Garibaldi et al., 2011a). On average, bee visitation rates and richness are reduced by half at distances about 670 and 1500m, respectively, from natural vegetation (Ricketts et al., 2008). As a consequence, not only does average crop yield often decrease with distance to field margins or natural vegetation (albeit at lower rates than pollinator abundance and richness), but it also becomes less predictable (Garibaldi et al., 2011a). A long-term survey conducted in Scandinavia also revealed trends in the composition and diversity of bumble bee communities and crop yield. Bumble bees are important pollinators in temperate regions of the world, particularly of the Northern Hemisphere. In Sweden, a drastic decrease in bumble bee diversity has occurred during the last century and present assemblages are dominated by two short-tongued species, *Bombus terrestris* and *B. lapidarius* (see section 3.2.3). Associated with this shift in bumble-bee community composition, Bommarco et al. (2012) found declines in average seed yield of red clover (*Trifolium pratense*) in recent years and a long-term trend of yield that was twice as variable after 90 years. Also, cultivation of some traditional crops, like mustard (*Brassica rapa*), has been decreasing in northern India and Nepal because of declining yields. In this case, local farmers blame high levels of pesticide application to crops, and their impact on bees, as the main cause of reduced yields (Vaissière et al., 2011).

Based on this evidence, large-scale agriculture expansion should be expected to be associated with decreasing crop productivity. An analysis of the 54 major crops cultivated in France over the past two decades found that potential benefits of agricultural intensification were offset by increasing pollination deficits, as the mean and predictability of the yield of pollinator-dependent crops decreased with increasing intensification (Deguines et al., 2014). However, at the global scale Ghazoul and Koh (2010) did not find any consistent relationship between changes in yield in pollinator-dependent crops and agricultural intensification. Also, the long-term FAO dataset provides no evidence so far of a decrease or deceleration in the growth in the average yield of pollinator-dependent crops compared to non-dependent crops during the last five decades (Figure 3.13; Aizen et al., 2008). A more detailed analysis confirmed no deceleration of yield growth with increasing pollinator dependency (Garibaldi et al., 2011b), although introduction of managed pollinators might be masking to some extent any effect of pollinator decline on crop yield (Potts et al., 2010). However, the absence of evidence of decreasing or decelerating agriculture yield over time among pollination-dependent crops worldwide (Aizen et al., 2008) must not be taken as proof of an absence of risk of global agriculture to an on-going or future large-scale pollinator decline.

Further analyses of the FAO dataset revealed that increasing pollinator dependency is associated with lower and more variable rates of yield growth (Garibaldi et al., 2011b). These findings are consistent with the hypothesis that animal pollination limits the productivity of many crops worldwide, and stress the vulnerability in the productivity of many crops to pollination disruption.

Although introduced pollinators can substitute partially for the pollination provided by declining wild pollinators, as potentially invasive organisms they can become superabundant, overexploiting both wild and crop flowers, and thus reducing reproductive output and agricultural yield. For instance, the extremely high densities reached by European *B. terrestris* in the Patagonia region of South America (Morales et al., 2013) can be associated with a marked decrease in raspberry fruit quality via increasing style damage (Sáez et al., 2014). Also, because the honey bee, *Apis mellifera*, harvests large amounts of pollen, the balance between pollination and pollen theft could shift with its abundance (Hargreaves et al., 2009). However, benefits might still be higher than the costs, as in the case of coffee yields in the Americas, which have been claimed to have increased after honey bees became highly invasive following the arrival of the Africanized race of honey bees (Roubik, 2002). In any event, because of their manifold effects on native faunas and floras and uncertain impacts on agriculture, pollinator introductions should be discouraged in places where they are not native and have not been introduced in the past.



31. Figure 3.13. Temporal (1961-2006) trends in mean crop yield.
The depicted means (+ or – 1 standard error) in relative yield were estimated from the subset of 10 pollinator-dependent and 10 non-dependent crops widely cultivated in both the Developed and Developing World (modified from Aizen et al., 2008).

3.9 Indigenous knowledge

3.9.1 Trends in stingless bee keeping and wild honey bee colonies

Stingless bees, in the tribe Meliponini, are one of the groups of social bees that live in colonies, constructing hives that include production and storage of honey (Roubik, 1989). Stingless bees are widely distributed in the tropics and sub-tropics and have been widely managed/exploited in central and south America and Africa.

Knowledge of the rewards contained within stingless bee hives appears to be fairly ancient. The ancient Mayans highly valued honey and wax from stingless bees and recognized a special god, Ah Musen Cab, associated with stingless bee keeping. In addition, many hunter-gatherer peoples, including the Hadzabe of Tanzania (Peterson, 2013; Marlowe et al., 2014) and the Abayanda of Western Uganda (Byarugaba, 2004) have folk taxonomic systems recognising distinct species of stingless bees and the different qualities of their honey. Wild harvesting of stingless bee honey is also widely practised in Africa today. The bulk of stingless bee diversity is found in the Neotropics, with over 400 species described from Brazil alone where > 30 spp. are important for honey production. In the African dryland, savannah and forest habitats they can be among the most abundant bees seen at flowers (Martins, 2004).

Despite the huge potential offered by stingless beekeeping in rural areas in developing countries (Macharia et al., 2010; Jaffé et al. 2015), the trend appears to be a decline of stingless bees as well as loss of the knowledge of their husbandry and management by traditional stingless beekeepers. Amongst the Mayan people of the Yucatan peninsula, *Melipona beecheii* is managed in traditional log hives. Beekeepers using this bee, from the Maya zone in Quintana Roo state, Mexico, testify to a 93% decrease in hives during the past quarter century (Villanueva-Gutiérrez et al., 2005). The loss appears to be driven primarily by beekeepers adopting Africanized honey bees for management in hives as they produce higher volumes of honey.

In Kenya, stingless beekeepers in Kakamega Forest recalled times when stingless bees and their products were a common part of the forest-edge households' livelihood and diet. However, the decline in abundance of stingless bees, as forests have been cleared, has resulted in fewer keepers of stingless bees. Loss of stingless bees in Kenya appears to be driven by both loss of habitat as well as wild-harvesting of colonies (Martins, 2014). As more areas of tropical forest are lost, this trend is expected to continue both for stingless bees and honey bees, as widely echoed by the forest-dwelling Ogiek and other hunter-gatherer peoples in East Africa who have had to adapt cultural practices such as payment of dowry, which was traditionally done with several large bags of honey, to a token amount of honey today due to the decline in availability of wild colonies for harvest. This is attributed to destruction of forests, overharvesting, logging and charcoal production (Samorai Lengoisa, 2015).

Notably, in several areas where honey bees have been introduced, competition with native stingless bee species has been observed by local communities (Cebolla Badie, 2005), and in different cultural contexts, different trends in use of introduced honey bees is evident (Ramos-Elorduy Blásquez, 2009). Agricultural intensification can also change the availability of wild honey, and this trend has been documented in Ethiopia (Verdeaux, 2011) and introduction of managed honey bees also affects local honey bee varieties and their managed that were better adapted to local ecological conditions and cultures, as an example from the Cevennes National Park in France demonstrates (Elie, 2015). Introductions of honey bees into New Zealand have had a positive impact on local cultures, where the Maori have adopted the use of honey and developed a strong appreciation for bees, including noticing their recent declines and ecological roles (Doherty and Tumarae-Teka, 2015).

Stingless beekeepers in a number of communities in Brazil are increasing the numbers of their colonies through projects facilitated by local non-governmental organizations and community-based organizations working in collaboration with beekeepers (Cortopassi-Laurino et al., 2006), even as deforestation affects these bees (Brown and Albrecht, 2001). There is a growing production of stingless bee honey in Brazil and this is one of the main sources of income for some communities. For example, in the Xingu Amazon region three different species of stingless bees are managed: jatai (*Tetragonisa angustula*), tiuba (*Melipona compressipes*) and marmelada (*Frieseomelitta* sp.).
Wild honey bee species in Asia are also facing declines, with some evidence that logging is destroying the ‘bee trees’ used by colonies of *Apis laboriosa* and *Apis dorsata* for nesting in Bhutan, Nepal and India (Verma, 1991; Pain, 2009; Vit, 2013). In parts of Indonesia local communities have devised innovative ways of enticing migratory swarms of *A. dorsata* using rafters placed in strategic locations for these bees to construct combs, and carefully manage these in relation to flowering trees in the forest, but see an overall decline due to the loss of forest cover to oil-palm plantations (Madhu Duniya, 2011, Césard and Heri, 2015). In both wild stingless bee and honey bee colonies (all *Apis* spp.) there is a need for more thorough baseline assessments to establish whether declines are on-going, are reversible and what the drivers of these are (see Chapters 2 and 6). This is also an opportunity for inter-disciplinary collaborations between scientists as well as holders of indigenous and local knowledge.

3.10 Knowledge gaps and recommendations

An obvious conclusion of our survey of the state of knowledge of status and trends in pollinators is that surprisingly little is known about them, with the exception of honey bees (*Apis mellifera*) and some bumble bees (*Bombus* species) and for a few well-studied regions of the world, particularly NW-Europe and North America. Given that these are only a tiny fraction of the diversity of pollinator species on the planet, it is difficult to draw conclusions, other than broad generalizations, with much confidence. Although the growing interest in pollinators and research on them and the ecosystem services they provide allow us to go somewhat beyond the similar conclusions of a study of the status of North American pollinators (National Research Council, 2007), it is obvious that much remains to be learned. For an overview of key questions in this field also see Mayer et al. (2011), who list questions drawn up by the scientific experts in the field. Note, however, that they list mainly scientific questions. These questions alone can rarely provide a complete answer to questions involving societal stakeholders such as farmers or (traditional) managers of bees or natural areas (see Biesmeijer et al., 2011).

To assess better the status of pollinators, standardized pollinator monitoring schemes need to be implemented. Monitoring of honey bees (recently set-up as part of the CoLoSS network and now broadly adopted) now annually provides precise estimates of winter colony mortality for many countries. This provides policy-makers with essential information to design mitigation strategies. Monitoring of other pollinator groups, particularly bees and flies that dominate pollination in many ecosystems, is more difficult, but not impossible. Only in this way can policies be targeted to those groups and regions where acute problems actually are occurring. Monitoring should target both natural ecosystems (where many threatened pollinators and pollinator-dependent plants occur) and agro-ecosystems (where pollinators are needed for crop pollination). Note that some pollinator groups are severely understudied, e.g., beetles, wasps, and moths (see Cascante et al., 2002; Donaldson et al., 2002; Johnson et al., 2004).

Occurrence of pollination deficits for crops and wild plants and its cascading effects are largely unknown. More studies are needed to (1) assess in which crops and under which management and landscape conditions pollination deficits occur; (2) identify when and where in natural systems wild plants suffer from pollination deficits; (3) whether pollination deficits lead to yield gaps (in crops), lower reproduction and population decline (in wild plants). Monitoring for pollination deficits would produce important information on status and (after some years) trends on which policy-makers could base incentives and mitigation measures.

Our knowledge of plant-pollinator networks is often too limited to predict impact of climate change and other drivers on interaction networks and their ecosystems. Currently information is mostly collected on visitation (e.g., hummingbird A visits flower X), whereas pollination (e.g., hummingbird A deposits 125 pollen grains per visit to flower X) or reward intake (e.g., hummingbird A collects 1mg of sugar per visit to flower X) are the ecologically relevant parameters. Such information is particularly relevant in light of the “rewiring” that takes place in flower visitation webs as species composition changes in response to disturbances, or over time.

Scientific knowledge, albeit incomplete, does not always reach farmers, habitat managers or policymakers. Scientists need to be more active in making their knowledge accessible. Awareness of policymakers, farmers and the general public can only increase when information on pollination is included in the right way and through the channels used by each stakeholder group. For example, inclusion of pollination information and promotion of best pollination practices in agricultural extension could improve crop yields and pollinator-friendly crop management.

Traditional and local knowledge on pollinators, their products and pollination practices is underused in policy and science. Such information needs to be collected before it disappears and can be important in guiding communities towards sustainable futures. For example, knowledge on traditional management of stingless bees in many tropical regions may be applied in crop pollination and small-scale farming systems.

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Chapter 4: Economic valuation of pollinator gains and losses

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Executive Summary

Pollinators provide a wide range of benefits to humans, such as securing a reliable and diverse seed and fruit supply, underpinning wider biodiversity and ecosystem function, producing honey and other outputs from beekeeping, and supporting cultural values. **These benefits can be expressed in economic terms to quantify the consequences of gains and losses in pollinator abundance and diversity to human wellbeing (*well established*)** (4.1, 4.2, 4.9)**.**

Current markets and economic indicators (e.g., Gross Domestic Product) fail to capture the full range of benefits from pollinator abundance and diversity (*well established*) (4.1.1.), and the full costs of supporting managed pollinators *(unresolved)* (4.1.1). Given that many decisions about land use are based on markets and economic indicators, such failures can result in the loss of pollinator-mediated benefits and sub-optimal land management decisions from a social perspective (*well established*). Indeed, declines in pollinator abundance and diversity have altered the benefits they provide to humans (*established but incomplete)* (4.1, 4.2).

Economic valuation of such pollinator-derived benefits provides information to undertake corrective actions on these market and economic indicator failures *(unresolved).* Each time we make a decision affecting natural or semi-natural habitats there is an implicit (i.e. not informed) valuation of them, involving trade-offs with other land-use decisions. Therefore, humans are always valuing nature's benefits, either directly or implicitly. Economic valuation is a process in which these values are made explicit by using well-informed methodologies and justified criteria. Explicit values provide information to land managers (e.g., farmers), related industrial sectors (e.g., pesticides, supply providers), consumers, general public, and policy makers to modify land use choices or other public policies with greater consideration of pollinator biodiversity and sustainability (4.1.1, 4.2, 4.6).

The economic consequences of pollinator gains and losses are multidimensional, affecting the production and distribution of scarce goods and services, including production factors (e.g., human, financial and natural assets) (*unresolved*). According to the IPBES conceptual framework, value is defined as: *"In keeping with the general anthropocentric notion of "nature's benefits to people", one might consider a benefit to be ecosystems' contribution to some aspect of people's good quality of life, where a benefit is a perceived thing or experience of value"*. The impacts of pollinator gains and losses can be valued in both non-monetary and monetary terms. Non-monetary indices, such as crop production and nutritional quality enhanced by pollination services, can be of great interest (4.2.6). Within monetary terms, economic methods can measure both market values, when goods or services traded in economic markets (e.g., crop production) (4.2.2, 4.2.3, 4.2.4, 4.2.5), and non-market values, when relating to benefits not directly traded on markets (e.g., supporting aesthetic wild flower diversity) (4.2.6).

Economic valuation can measure use values, such as crop production from insect pollination, and non-use values, such as the values people place on the existence of pollinators. Valuation can be aggregate, examining the combined value of all pollinators within a region, or marginal, examining the change in value given a certain (non-total) gain or loss of pollinators. Marginal values are relevant for decision making because partial increases and decreases in pollinator abundance and diversity are more likely than complete loss, and because decisions concern marginal changes (4.1).

The annual market value of additional crop production directly linked with pollination services is estimated at $235bn-$577bn (in 2015 US$) worldwide (Table 8, Section 4.4.3) (*established but incomplete*). In addition, in the absence of animal pollination, changes in global crop supplies could increase prices to consumers and reduce profits to producers, resulting in a potential annual net loss of economic welfare of $160 billion-$191 billion globally to crop consumers and producers and a further $207 billion-$497 billion to producers and consumers in other, non-crop markets (e.g., non-crop agriculture, forestry and food processing) (4.7.4, Table 10, Section 7).

In addition to crop production, pollinators provide a full range of non-monetary benefits to the economy, particularly to the assets that form the basis of rural economies (*established but incomplete*). For example, human (e.g., employments in beekeeping), social (e.g., beekeepers associations), physical (e.g., honey bee colonies), financial (e.g., honey sales) and natural assets (e.g., wider biodiversity resulting from pollinator-friendly practices). The sum and balance of these assets are the foundation for future development and sustainable rural livelihoods (FAQ section, 4.2, 4.5). Therefore, evaluating how pollinator-friendly *versus* unfriendly practices (or landscapes) change these assets would be a robust approach to valuing pollinator changes in both monetary and non-monetary terms. This approach allows quantification of the synergies and trade-offs (for example, between financial and natural assets) associated with pollinator enhancement (4.2.6).

Most studies of the economic impacts of pollinator gains and losses only estimate the monetary benefits in existing markets rather than the actual impact they have on peoples' wellbeing (*well established*). These estimates are dependent upon the methods utilized, and can change dynamically across spatial and temporal scales (*well established*) (4.3). For example, the benefits of pollination services to apple production was found to vary between $791 and $25,201 per hectare (2015 US$) for different agroecological systems using different methods (4.7.4, Table 10, Section 7).

Estimation accuracy of the economic value of pollinator gains and losses are limited by existing biological and economic data, as well as the need for methodological development (*established but incomplete*). For example, although there is broad understanding of the relative extent to which yields of most crops benefit from pollination, there are a number of uncertainties surrounding these such as the shape of relations between crop yield and pollination, how they vary for different cultivars of the same crop, and the interaction between pollination and agricultural inputs (4.5).

Unstable pollinator assemblages can result in substantial economic risks while highly diverse, resilient assemblages can provide stable long-term services (*established but incomplete*). To date, although a number of methods exist, no studies have quantified the economic value of this stability and few have considered the potential economic risks and uncertainties affected (4.4).

The spatial and temporal scales of ecological processes that affect the health of pollinator assemblages and their benefits, and the scales of social, economic, and administrative processes (involved in land-use decisions, market regulations, etc.) are seldom well aligned (*established but incomplete*). An important challenge is to match the ecological scale with the institutional scale of the problem to be solved (*unresolved*). For example, socio-economic value at larger scales may be of interest for policymakers, whereas profit analyses at smaller scales may be of interest for farmers. The temporal scale is also important, because ascribed values are endogenous to changes in the number and diversity of pollinators and other system (e.g., network) properties. Therefore, static values provide only limited, and perhaps misleading information for decision makers. Furthermore, within any given time period, the use of constant (e.g., average) values is also potentially misleading as it disguises the spatial variation in services and hence values (4.2.6, 4.3, 4.6).

Impacts of pollinator loss will be different among regional economies, being higher for economies based on pollinator-dependent crops (whether grown nationally or imported) (*established but incomplete*). For example, many of the world’s most important cash crops are pollinator-dependent. These constitute leading export products in developing countries (e.g. coffee and cocoa) and developed countries (e.g. almonds), providing income and employment for millions of people. In general, the importance of animal pollination services varies between 5-15% of total regional crop market output depending on the area, market price, and pollinator dependence of the affected crops, with the greatest contributions in East Asia (4.7).

Although the economic consequences of pollinator gains and losses can be significant across the world, most evidence is based on global market data or case studies in the developed world with very few detailed studies in the developing world (*well established*). This regional bias may therefore fail to capture the impacts of pollinator shifts on the people whose livelihoods and diets are most vulnerable to pollinator losses (4.7.3).

The joint use of monetary and non-monetary valuations (integrated valuation) of pollinator gains and losses can be used to better inform decision making on land use (*unresolved*). Valuation of pollinator shifts can help in the decision making process through cost-benefit analyses, risk analyses, socioeconomic studies, etc. This information can be used in certifications, environmental schemes, green GDP, and regulatory frameworks (4.6.3).

Introduction and outline

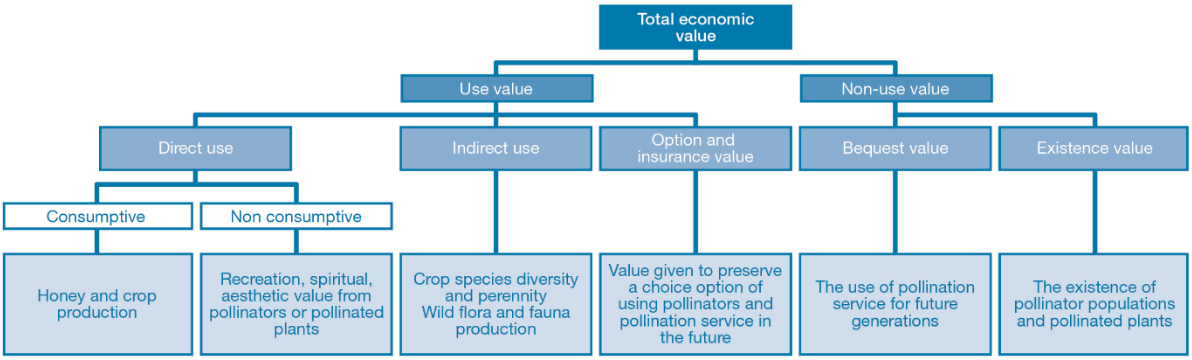
Pollinators are a key component of global biodiversity, providing vital ecosystem services to crops and wild plants (Klein et al., 2007; Potts et al., 2010; for more details, see Chapters 2 and 3). However, there is evidence of recent decline in both wild and managed pollinators and parallel decline in the plants that rely upon them (Potts et al., 2010; Biesmeijer et al., 2006). Declining pollinators can result in the loss of pollination services, which can have important negative ecological and economic impact that could significantly affect the maintenance of wild plant diversity, wider ecosystem stability, crop production, food security and human welfare (Potts et al., 2010).

The importance of animal pollinators in the functioning of most terrestrial ecosystems has been extensively described and analysed in a broad range of scientific literature (see Chapter 3). The importance of pollinators and pollination services can often be evaluated in economic terms in order to link decisions made with economic consequences (Daily et al., 1997; Daily et al., 2000). The economic assessment of pollinators and pollination services is measured by their total economic value (TEV; summarized in **Error! Reference source not found.**). Economically, the total value of an ecosystem service is the sum of the utilitarian reasons a society has to maintain it. This is typically divided into (i) use values, the values of the benefits that people gain from the functioning of the ecosystem (e.g., the pollination of crops); and (ii) non-use values, the values that people attribute to the existence of an ecosystem service, regardless of its actual use (existence value, e.g., the existence of pollinators) or the value they place on the potential to use the ecosystem service in the future (bequest value e.g., species that could pollinate crops in the future). Pollinators and pollination have a use value because the final product of their service can be used directly by humans, such as with crops or honey (a consumptive use), as well as the leisure and aesthetics created by the presence of pollinated wild plants within the landscape (a non-consumptive use value). Pollination can also provide indirect use values through supporting the reproduction and genetic diversity of wild and cultivated plants that benefit humans. Finally, the use value of pollinators and pollination also contains an option value (the value given to preserve a choice option of pollinators and pollination-dependent products in the future) and the insurance value (the capacity of pollinator communities to reduce the current and future risks associated with using pollination services; Baumgärtner and Strunz, 2014).

However, not all these values are directly related to markets (only the consumptive uses that are marketed). Consequently, the impacts of management on pollination services could be under-estimated when making decisions, potentially resulting in inefficient or unsustainable use of resources. Economic valuation provides two forms of essential information to stakeholders. Firstly, it highlights the economic contribution of pollinators to the various benefits provided to the agricultural sector and society. Thus, it tells the decision maker how much net benefit arises from different interventions, which in turn allows for the optimal design of such interventions. Secondly, economic valuation can assess the impact of variations in pollinator population on the economic welfare of different groups of people, such as farmers or consumers. By considering this information, decision makers, from both the public and private sectors, are able to make better-informed decisions about the impacts of proposed investments, public spending or management changes. This chapter aims to review the conceptual framework and the various methods of economic valuation of pollinators and the effective use of these valuations. There are also other value systems, including spiritual, cultural and indigenous and local knowledge values, which can inform decision-making, these are reviewed in Chapter 5.

In this chapter, pollination services are considered an ecosystem service, i.e., “the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life” (Daily, 1997). The evidence is clear for wild pollinators that are provided by natural ecosystem as forests or soils, but some ambiguity remains when considering managed pollinators as they can be considered as livestock, far from nature. However, they are used to provide services in agricultural systems that, while heavily managed, remain a functioning ecosystem (or agro-ecosystem, see Swinton et al., 2006; Swinton et al., 2007; Zhang et al., 2007). Thus described, pollination services from managed pollinators are ecosystem services offered by the agro-ecosystem. Unlike many well-quantified ecosystem services, pollination services are provided by mobile organisms that can move in uneven patterns across their foraging range, making them more difficult to assess accurately (Kremen et al., 2007). Furthermore, pollination services are an intermediate service, a service that is not beneficial in itself but instead underpins other benefits, such as crop production and landscape aesthetics, by helping produce pollinator-dependent crops for human food and nutrition security, along with the reproduction of certain plants (Fisher et al., 2009; Mace et al., 2012). The value of intermediate services is assessed not by looking at their direct consequence (pollination) but by their impacts on the final goods that are produced (food, honey, etc.). These final goods have a market price which gives some reasonable indication of their use value (note that prices may under-estimate values). However, pollinators are also final ecosystem services in themselves because of the value associated with their existence. Although this complicates the challenge of accurately valuing pollination services more substantial, these abstract benefits can still be valued economically. Consequently, the methods of valuing the impact of pollinator and pollination gains and losses can range from very simple to very complex at several levels.

The chapter starts with an outline of some frequently asked questions on economic valuation of nature and ecosystem services, with emphasis on pollinator gains and losses. Section 1 then presents the rationale behind economic valuation of pollinators and pollination. Section 2 critically reviews the range of methodologies that have been applied to quantify the benefits of pollination services. The strengths and weaknesses of each method are also discussed, in terms of their ecological and economic validity as well as the capacity to extrapolate the values to different spatial scales and data requirements are outlined for each one. Valuation may vary relatively according to the ecological or biological functioning of the ecosystems that support pollinators, the spatial and temporal specificity of the pollinating animals, and the value given by the consumers or beneficiaries of the final good obtained by this service (Farber et al., 2002; Fisher et al., 2009). Section 3 focuses on temporal and spatial scale effects on the economic valuation, including tools for integrating these factors into valuation. Economic valuation tends to assume that the consequences of pollination service loss are precisely known. However, decision-making is confronted with stochastic relations between events, giving rise to a number of factors that can significantly affect the economic value of pollinator gains and losses. Section 4 considers the effects of economic risk and uncertainty inherent to pollination services (e.g., fluctuations in service delivery or market prices) and pollinator community resilience, including methods to quantify and value these factors. Section 5 reviews knowledge gaps related to the economic valuation of pollinators, covering agronomic, ecological and economic knowledge that could be used to improve value estimates. Section 6 reviews the applied use of these economic valuations for decision-making, reviewing the stakeholders concerned with these valuations and, for each of them, how they should interpret the values and use them. Finally, Section 7 analyses case studies that used the methodologies presented in this chapter. The chapter ends with a synthesis of all these sections 8.



32. Figure 1 – Total economic value of pollinators and pollination service
(adapted from Pascual et al. 2010).

Frequently asked questions

In this section, we briefly outline some common frequently asked questions on economic valuation of nature and ecosystem services, with emphasis on pollinator gains and losses. We briefly explain the approach adopted in this chapter, and direct the reader to specific sections where this is discussed in detail in this chapter. We hope this section clarifies the benefits and the limits of economic valuation.

1) Are economic values the same as prices?

Distinction must be drawn between prices and values. Prices are the monetary exchange rate of a good on a market, or information that institutions (including markets) link with things in order to manage their use. In contrast to this, economic values express the importance people place on things, more precisely, they are a quantitative expression of the impact a service has on the overall economic wellbeing of people. Each time we make a decision affecting natural or semi-natural habitats there is an (implicit and possibly explicit) valuation of the consequences of this choice, involving trade-offs with other land-use decisions. Therefore, humans are, in many circumstances, implicitly valuing ecosystems through the decisions they make. Economic valuation is a process in which these values are made explicit by using well-informed methodologies and justified criteria. The neoclassical economic theory of value can be regarded as a theory of what should be a perfect price system in order to transmit to economic agents the most relevant information on the relative utility and scarcity of all goods and services. However, in the real world, prices do not usually indicate the values. Ideally, economic valuation studies should estimate values; yet, several methods tend to estimate prices or price variations, which are used as indicators of value (e.g., market price of renting honeybee colonies can be used as a proxy of the economic value of honeybees). See Section 1 for a more detailed explanation.

2) Does economic value mean monetary value?

Though the question is often addressed in these terms, it has to be reformulated because “monetary value” has no clear meaning. If the question is “should the economic values necessarily be expressed in monetary terms?” the answer is “not necessarily”, but for practical reasons and communication purpose, it is generally the case. Economic values can be expressed in any currency. Nevertheless, monetary units have practical advantages, for example, as a common unit across highly diverse costs and benefits and it is the same unit that other investments (including in non-environmental policy) are assessed in. Therefore, monetary units are generally used in valuations, although this tends to reinforce the ambiguity between values and prices. In monetary terms, economic valuation methods include market prices, when the benefits relate to existing markets (e.g., crop production), and non-market values, when relating to benefits not directly traded on markets (e.g., supporting aesthetic wild flower diversity). Non-monetary indicators can also be of great importance, for example, given that demands for agricultural products are constantly increasing from a growing and more affluent population, it is important to maintain the regenerative nature of agroecosystems, such that food production and diversity, and livelihood are improved for farmers. These important considerations are indeed difficult to express in monetary terms. See Sections 1 and 2.4 for further discussion.

3) Does the valuation of nature and ecosystem services imply privatization or commodification?

Economic evaluations are usually motivated by goals such as decision support, policies design or raising awareness among public decision makers of the importance of certain issues. The intention is not privatizing or commodifying public assets, which is often considered both impractical and unethical, but to recognise their values and include them explicitly in public or social decision-making. For example, the value of a river as a provider of clean water for a town does not imply a market for buying and selling rivers. Similarly, the value of a meadow as a provider of insect pollination for nearby crops does not imply a market for buying and selling meadows. It recognises a common, natural asset that should be protected for the benefit of the overall welfare of those affected. Valuation allows the importance of such an asset to be compared with the interest for society of alternative actions or policies that degrade it. Therefore, using techniques to estimate the value of a resource to society can help its members to better understand the scope and scale of the benefits received from the resource. Furthermore, economic values and other valuation systems (see Chapter 5) are not mutually exclusive and can be combined using multi-criteria analyses. See Section 1 for further discussion.

4) Does economic value include non-use values?

Non-use values have been progressively introduced in economic valuation of natural assets in order to get more significant indicators of the total importance of the multiples reasons explaining why people value nature’s services. Economic valuation thus includes methods to quantify both use values (e.g., crop production due to insect pollination) and non-use values (e.g., the value people place on the existence of pollinators). Indeed, valuation theory places a great emphasis in capturing both of these types of economic value. See Section 1 for further discussion.

5) How much uncertainty is associated with economic values?

The uncertainty is an important limitation affecting the precision of economic valuation methods related to crop production. For example, the underlying empirical data linking pollination to yield are sparse and do not adequately represent variation among crop varieties, years, or places, particularly for the widely grown crops. Unfortunately, valuations have often been widely communicated without explaining this uncertainty (whether or not it is in the discussion text of the scientific papers). The fact that the estimation of values share uncertainty, as is true of most estimates in any scientific field, does not mean that the process and use of valuation is inherently flawed. If the valuation process is not made explicit, the value given to natural assets or ecosystem services may be zero, a value that we can be certain is wrong. It is important that values should be communicated to policy makers and the public with corresponding estimates of uncertainty, for example, by providing ranges of values instead of a unique value. We also identify in this chapter several biological knowledge gaps that directly affect valuation uncertainty. Thus, though variations among valuations may be the effect of technical failure, they may also reflect the fact that the valuation of the same service in different circumstances has no a priori reason to be the same. Moreover, these differences can simply reflect the natural heterogeneity in benefits, which in turn inform these values in decision-making. The underlying ‘true’ value that we are trying to measure is likely to fluctuate itself quite considerably because of changes in food demand and supply, the development of technology and changes in populations and their socio-economic characteristics, among others. See Sections 3 and 4 for further discussion.

6) Does valuation precludes conservation because the use values of natural systems are usually lower than alternative land uses?

In many instances, a particular use value of natural ecosystems can be lower than alternative land uses. For example, the opportunity cost of replacing more forest area with coffee plantation can be higher than the pollination services provided by the forest habitat to the coffee plantations. In plain language, it may be possible for a farmer to make more profit by expanding coffee area than from the higher yields (tons ha^{-1}) that result from pollination services from forest next to plantation, thus creating incentives to destroy the forest. Although a particular use value of nature can be lower than alternative land uses, the estimation of this value does not inherently promote the destruction of nature. On the contrary, valuation may illustrate that the long-term consequences of pollination services lost may be greater than the value of new coffee production by reducing benefits to other plantation patches. Furthermore, the economic value of pollination services is additive to the values of other ecosystem services that forests provide in greater quantities than plantations, like clean water and fresh air to humanity (i.e., use values beyond crop pollination), and that conserving nature has a value for society even without perspective of use (i.e., non-use value is high). In this way, by estimating the value of pollination among other ecosystem services we add reasons to the conservation of nature in addition to the traditional, long-standing non-use values. As stated before, an advantage of economic valuation is to make the benefits and the decision-making process more explicit in regards to nature. For example, a particular forest may have low private use values (e.g., timber value) but high public use values (e.g., recreation). Social decision makers might therefore protect this forest even if its non-use values were low (e.g., no wildlife species of conservation interest). Therefore, a key issue is; ‘valuation for who’? The potential value of a field to the farmer is different to the potential value of that field to society. The market reflects the preferences of private individuals. Economic valuation allows us to look at values in the round, both private and public, and shows that the two are rarely identical. See Sections 3 and 5 for further discussion.

Section 1 – Nature and significance of the economic valuation of pollination

1.1 On the meaning of economic valuation

The concept of value is used to describe how agents (typically individuals or, more broadly, societies) assign or express their interest in things; the "things" are objects, ideas, persons or anything else. Among multiple frameworks, the economic concept of value aims to measure and capture these values in largely quantitative terms; the current significance that is explained within this section. For an extensive analysis of economic valuation, non-economist readers are referred to microeconomic or environmental economics textbooks such as Just et al. (2008), Hanley et al. (2013) or Perman et al. (2012).

1.1.1 Understanding the meaning of economic value: utility and scarcity

Economics has been defined as "the science, which studies human behaviour as a relationship between ends and scarce means which have alternative uses" (Robbins, 1932). As such, economic value reflects the utility and scarcity of "things".

Utility refers to the satisfaction that agents obtain from the consumption of goods or services (a simple distinction is that services are not depleted by use, while goods can be). It is usually accepted that agents' utility is subjective and depends on their preferences. The social welfare is the sum of the utility gains and losses of each agent in society. The utilitarian perspective advocates choosing options that offer the greatest social utility or welfare. However, consumers do not derive utility directly from pollinators, but they can gain utility from consuming the products of the pollination process, such as fruits or aesthetically valuable flowers (Fisher et al., 2009).

Scarcity is not necessarily a measure of physical amounts, but of the tension resulting from the lack of supply of usable resources relative to the wants of the people (demand). Scarcity is at the core of the allocation issues. The scarcity of pollinators can lead to a decrease of pollination services and therefore a reduction of the utility of consumers of these benefits. Properly informed, economic valuation of pollinators provides relevant indicators of the relative utility and scarcity of the diverse resources offered through the process of pollination that may contribute to human welfare.

1.1.2- Values, costs and prices

Advantages and limitations of bringing the diversity of preferences into a single-dimension analysis. Economic valuations typically measure values in monetary figures. However, this is often criticized as being too simplistic as it brings the diversity of wants and needs into a one-dimension indicator. When related to nature, these wants and needs are difficult to substantiate and do not really help decision makers to understand the actual functioning of human societies in their relation to ecosystems as, because of methodological limitations, economic valuations alone cannot fully capture the richness and diversity of relations between societies and nature. This is a particular issue when the results are poorly reported and do not allow to fully capture or express the variability and diversity of values among individuals. However, the purpose of the valuation is to enlighten decision-makers on the utility/scarcity issues resulting from the choices they can make. Expressing benefits and costs in a way common to standard economic activity allows, aside of other measures, for more informed decision-making than would otherwise be possible. Expressing the intensity of the tensions on ecosystem services with a monetary indicator allows comparing them with the prices that can be observed on the markets.

Prices, costs and values*: *how do they differ? Economists use three complementary but distinct concepts to express the impacts of economic activity in monetary units: prices, costs and values. Prices are the amounts that buyers must pay to sellers when there is a market i.e., the mechanism by which buyers and sellers interact to determine the price and quantity of a good or service. When the market is competitive, prices may vary in order to balance supply and demand. Costs express what agents must give up to get (or produce) the items they want, i.e., the efforts they would bear in terms of monetary cost, but also of time, inconvenience or income foregone (often referred to as opportunity costs). The use of ecosystem services could lead to a situation with no cost if there are no private cost (the cost incurred by the suppliers or the price paid by the consumers if any), or negative "externalities" (see Section 1.1.3.). Values reflect the interest of agents for goods and services, knowing that their preferences for these objects are influenced by both their needs and culture, and the information they have. Although they are often used interchangeably with values, the benefits are, in reality, the positive impacts produced by pollinators and pollination services. Economic valuation of pollination and other ecosystem services aspires to quantify the welfare gains from benefits[5].

Marginal values. Economic value is often derived from the maximum amount a consumer is willing to pay for a good or service in a market economy. For goods and services for which there is no market, these welfare values must be estimated by appropriate methods (see Section 2). The values useful to inform public policy choices are the values of goods and services units gained or lost resulting from the different choice options. These are what economists call marginal values. In the

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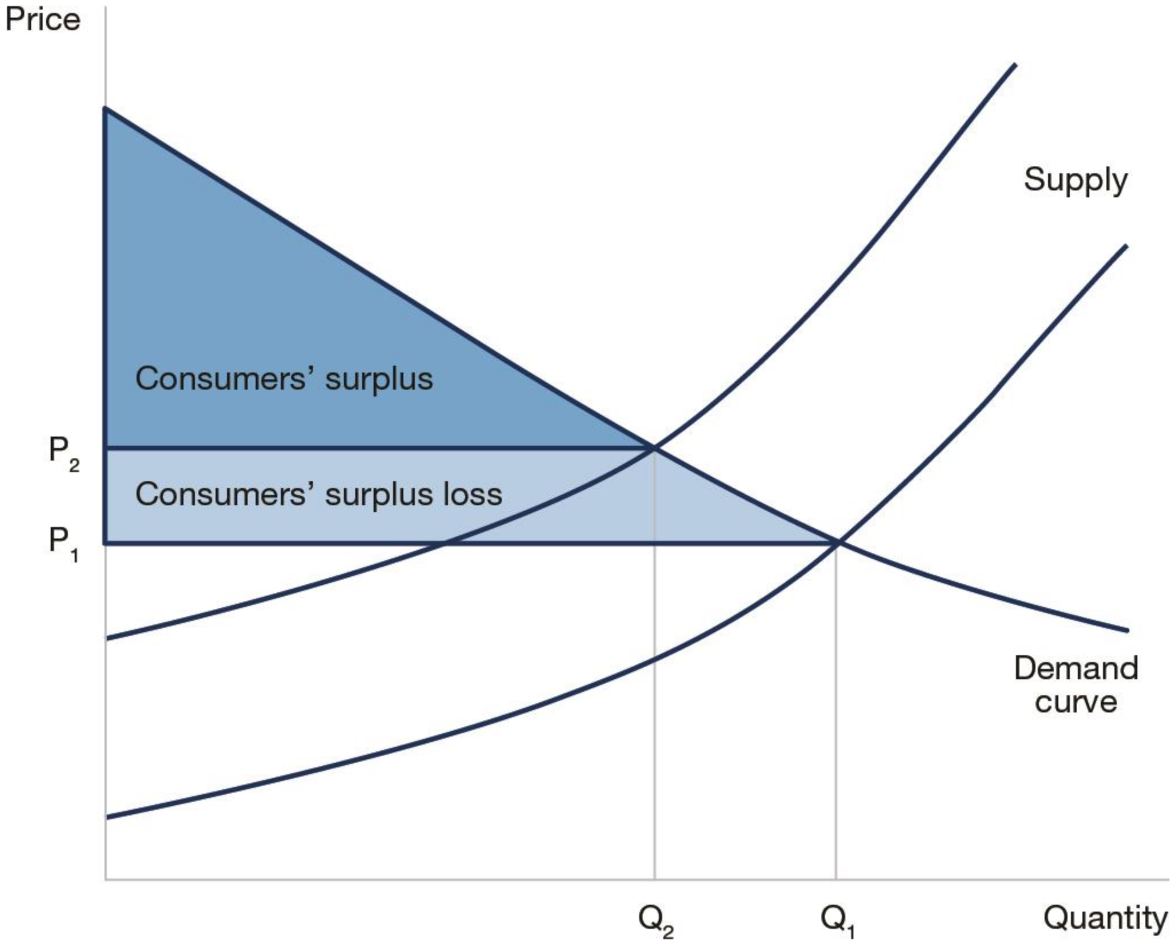
[5] In the "cascade model" of the CICES (Haines-Young and Potschin, 2010), benefits are defined as the share actually used of the entire ecosystem services potential.

context of ecosystem services, marginal values are even more relevant to informing decision making when complete collapse of services is unlikely (Costanza et al., 2014).

Net economic value and consumer surplus. Most economic valuations refer to "willingness to pay" (WTP) as a measure of the value of goods and services, regardless of whether WTP are obtained from direct statements or derived from any observable information. WTP is used as a measure of utility because it represents an individual's subjective view of what a thing is worth to them, given their budget constrains (as are market prices). In this way, it differs from utility, which may be much greater than an agent's budget. As it is not possible to ask every individual what they are willing to pay for a benefit, WTP is instead estimated from surveyed sample or observed behaviours using economic statistics (Econometrics). From this, it is possible to derive consumer surplus (CS) - the difference between what consumer would be willing to pay (WTP) to get a good or service and the cost they actually bear (market price or opportunity cost). Symmetrically, producer surplus (PS) is the difference between the market price and the production costs, representing the welfare gains to the producers of the good or service.



33. Figure 2– A simple scheme of the consumers' surplus.

In 33, the demand curve is built by ranking the WTP for each unit of the service from the highest to the lowest and the supply curve ranks units by increasing production costs. The intersection (Q*, P*) indicates the hypothetical market equilibrium (if there is a market), the equilibrium being where market prices are determined. The blue area covers the difference between the WTP and the market price for all the units that will be effectively produced and consumed. It represents the CS or net value of the service for the final consumers.

When there is no cost, CS is directly equal to the sum of WTP. Such cases are extremely rare in the real world, except where there are no alternative uses of that resource. This would be the case for ecosystem services if they were available at no cost, including no opportunity cost resulting either from legal constraint imposed to agents interacting with processes behind the ecosystem service or either from no alternative uses of these services.

1.1.3 – The externalities issue

An externality is a cost (negative externality) or a benefit (positive externality) that affects a party who did not choose to incur that cost or benefit, and does not get or pay compensation for it. A positive externality may be pollination when as a by-product of honey production. A negative externality could be the loss of crop pollination resulting from declining insect pollinators due to pesticide use.

The existence of externalities is directly dependent on the structure of the property rights (there is no externality if the managed pollinators belong to the farmers that grow the crops) and on the legal or economic status of pollinators or pollination services (*private goods, public goods, common goods, club goods*, see ***Error! Reference source not found.***, Fisher et al., 2009). The criteria of classification are two-fold: whether the consumption of a good by one person precludes its consumption by another person (rivalness) and whether or not one must pay for a good in order to use it (excludability). Honeybees can be considered as a private good or service when they are exchanged in a pollination market. Indeed, their services are privately owned (rival) and marketable (excludable). However, this classification assumes that the honeybees have no possibility to pollinate other crops in another field or wild plants. In this case, their services would become a common good because they are non-excludable (once they are provided everybody use them) but rival. Wild pollinators are considered as a public good because their services are non-rival (the fact that an agent uses them does not prevent other agents to use them) and non-excludable.

14. Table 1 – Characteristics of good and services from pollinators
adapted from Fisher et al. (2009).

	Excludable	Non excludable
Rival	**Private good.** Pollinators are private good when they are owned and / or managed by beekeepers; pollination may be a private good when it can be controlled in a specific area.	**Common good.** Pollination is provided by pollinators to all crops and wild flora in an area that depend only on the pollinator species (say honeybees). If the abundance or diversity is limited, there is rivalry among crops or between crops and wild flora.
Non rival	**Club good**. Pollinators could be a club if a group of farmers and beekeepers were organized to manage them, but scientific literature does not provide an example of such an organization.	**Public good**. Pollinators are public good when society can benefit from their service freely on the sole criterion of spatial proximity either to crops or to wild flora that create social amenities.

The economic status of pollination service is not quite clear because it may vary according to several circumstances and institutional context (see Cheung, 1973). When wild pollinators provide the service, it can be considered a public good. When honeybees that have not been rented provide services, they can be seen as a positive externality of honey production or as a reciprocal externality between beekeepers and farmers (Meade, 1952). When there is a market for hives rental (e.g. in the United States), pollination becomes a marketed service whose economic efficiency can be discussed (Cheung, 1973; Rucker et al., 2012). The difference between a market and non-market situation may have significant implications for the long-term management of the service. As long as there is no price signal from the market, or other signals from e.g., public policies, the agents (those whose choices and behaviours influence the dynamics or conservation of pollinators) will not be affected by the consequences of their choices and behaviours. This may potentially result in unstable or unsustainable long-term management practices.

1.1.4 – Monetary contribution versus economic value of the impact (or consequences) of an ecosystem service

A distinction should be made between the monetary value of the contribution to society of an ecosystem service and the economic impact of the loss of this service on the society. Taking the example of Figure 2, we could assume that the contribution of the ecosystem service to society is the gain in production between Q_1 and Q_2. In this way, the monetary value of the contribution would be the price, P_1, multiply by the net production due to the ecosystem service. The economic impact or consequence of the ecosystem service loss measures the impact on the price and quantities at the equilibrium of such a decline. The economic value of the decline would be measured by consumer and producer surplus losses. A more detailed discussion of the distinction between monetary contribution and economic valuation of pollination services can be found in Gallai et al. (2009a).

1.1.5 – The cost-benefit analysis framework

Economic valuations are usually part of a larger process of economic analysis. There are in fact two main frameworks: cost-benefit analysis and cost-efficiency analysis. Both framework use many of the same principals and data but have substantially different scope and objectives, making them useful in different situations.

Cost-benefit vs. cost-effectiveness analysis. Economic valuations refer primarily to the idea of calculating and comparing the costs and benefits, typically for policy-makers who have to make a decision among several choice options. Cost-benefit analysis aims at identifying the option with the highest net present value (NPV). NPV measures the balance of economic gains and losses linked to each option. In order to allow the comparison of cost and benefits that occur at different time, future gains and losses are down weighted using a discount rate (see Section 3.2.2.3.) according to the expected change in the value of money over time in order to obtain their present value. When calculated in a social context (as opposite to individual or private), and provided you have included and accurately valued all major benefits and costs and applied the appropriate discount rate, the highest NPV maximizes the social welfare. Cost-Benefit Analysis (CBA) is often used to identify this maximum: what are the levels of benefits gained from investing certain costs in an action. For example, Blaauw and Isaacs (2014) explicitly measured the benefits of pollination services from field margins sown with flowering plants to nearby blueberries relative to the costs of managing and maintaining these margins, finding that the total benefits outweighed the total costs after 3 years. It is therefore quite different from the cost-effectiveness analysis (CEA), which aims at identifying the most efficient way (lowest cost) to reach a particular goal: e.g., considering which mitigation measure would provide a minimum level of insect pollinators needed at the lowest relative cost.

CBA and distributive justice. A well-designed CBA should be able to recommend choice options that maximize social welfare. This optimal situation is sometimes called allocative efficiency because it is a situation where all goods are allocated to their most beneficial use. Nevertheless, this result may not be considered fair. The CBA may lead to solutions that are theoretically optimal but less preferable in terms of social justice since the positive and negative effects are distributed unevenly among agents. A policy with positive aggregated impact (say a ban of some pesticides that degrade the diversity of pollinators) may have a negative impact on certain agents that do not receive much or any of the benefits (e.g., farmers that grow wind pollinated crops that depend on this pesticide) (for overviews of these issues see Martinez-Allier, 2003; Pearce et al., 2006). Following seminal critics such as Rawls (2001), Sen (1999a, 1999b) or Fehr and Schmidts (1999), innovative analyses have introduced justice considerations. CBA can be carried out with different social decision making rules and taking into account issues such as the diminishing marginal utility of income (as required in the UK Treasury

Green Book guidelines) so as to incorporate issues of social distribution. The same comment may apply to CEA.

The sustainability criterion. Maximizing NPV is an efficiency-based criterion (the *most efficient alternative is the one that maximizes NPV*). As such the NPV can be positive for a project that is not sustainable (i.e., consistent with sustainability goals). Indeed, a development project can be sustainable, while its NPV is negative. The measure of sustainability is still an ongoing debate, however the classical sustainability criterion (Pezzey, 1989; Solow, 1993) assumes that consumption or welfare must be non-decreasing over time (the consumption of tomorrow should not be lower than the one of today). Since the consumption path is not necessarily representative of the welfare (Ascheim, 1994), classical conceptions of sustainability tend to focus on non-decreasing social welfare (Arrow et al., 2004). Following the concept of development as freedom (Sen, 1999a), recent perspectives tend to consider that a better sustainability criterion should be to maintain life opportunities (Howarth, 2007). The sustainability of pollination services should be evaluated regarding, not only its impact on consumption path or the welfare of a typical individual, but in order to maintain these life opportunities.

1.2. Linking economic values with pollination

The economic literature systematically links valuation with decision, public awareness and policy-making. Indeed, the conceptual framework of economic valuation is designed for collecting and organizing information toward hierarchizing and selecting choice options (for instance, agriculture policy or biodiversity strategy; Costanza et al., 2014). However, there apparently are broader motives behind economic valuations, and some authors have suggested that the link between valuations and decision-making was more related to general advocacy than to providing technical information (Laurans et al., 2013; Laurans and Mermet, 2014).

1.2.1- Understanding the importance of what is at stake

There is growing evidence of insect pollinator decline in many regions and its consequences (e.g., fruit and vegetable production decline in quantity and/or quality) are occurring, but building indicators of these changes is difficult and the result can be controversial or of limited social impact if expressed in a metric understood only by scientist and experts. The first interest of estimating the value of pollination service or the cost of pollinator decline is certainly to raise awareness on their importance for our societies, and to offer a clear and simple argument to help policy-makers to make choice about the opportunity to design and implement appropriate measures**.** Estimating the cost of pollinators' decline in economic terms allows the comparison of the result to other issues and, more importantly, to the cost of the remedies that can be proposed to this problem. In many cases, a precise study of the local variations in value indicators will be more helpful for decision making than global information.

1.2.2- Defining hierarchies, priorities and choices

Comparing the cost of declining pollinators to the cost of implementing alternative options in behaviours and solutions is clearly a difficult task. The main difficulty is usually to assess the cost of moving away from the current policies and behaviours. However, drawing a clear picture of alternative practice and organization can be a challenge as well. There is in fact little literature that directly offers estimates of such change (for an analysis based on cost of replacement, see Allsopp et al., 2008) and the few published results appear quite sensitive to the valuation method.

The design and assessment of cost-effective policies and action can be of real importance, but enlighten only a framework for a least-cost approach for some policy target (cost-efficiency) without demonstrating that it is the best social choice (the gain are not necessary larger than the costs). For example, the market for colony rental for almond orchards in California (Klein et al., 2012) might be the simplest way to meet the needs of large-scale monocropping landscapes. However, the pollinator shortage might also be solved if agricultural landscapes were to become more heterogeneous (Hussain and Miller, 2014), if producers switched to crops less dependent on pollinators or even developing artificial pollination techniques, but this is in most cases highly speculative. Economic valuation can assist in this process by identifying not only the most cost-effective solution but the fairest and most sustainable ones as well.

Section 2 – Methods for assessing the economic consequences of pollinator gains and losses

A number of methods have been used throughout the published literature to quantify the economic consequences of pollinator gains and (most often) losses. The following section reviews the principle details of each of these methods, focusing on what it measures (price or value and of what specifically), an overview of the methodology involved, it’s strengths and weaknesses, under what situations it is suitable to use and what data is required. Key examples of each method (some of which are reviewed in detail in Section 7) are provided for interested readers. Table 2 summarises these methods for ease of reference.

2.1. Price Aggregation

2.1.1. Aggregate crop price

What it Measures: The total market price of animal pollinated crop production.

Methodology: This method assumes that production of all animal pollinated crops would cease in the absence of pollination services and therefore equates the total sale price of all crops that benefit from animal pollination, with the value of pollination services themselves.

Strengths: As it assumes that crops are either uneffaced or completely lost, this method has very simple data requirements and is equally applicable at all spatial scales providing sufficient data is available. Assuming the complete loss of insect pollinated crop production may be realistic for some highly pollinator-dependent crops with high management costs.

Weaknesses: By assuming that crops are either entirely dependent upon pollination or not dependent at all, this method significantly overestimates the overall benefits of pollination services and does not estimate the marginal impacts of pollination services. Although production of some crops would probably cease, in many crops, these benefits are not large enough (Klein et al., 2007) that they could not potentially be produced profitably without animal pollination. Finally, the method does not consider producer's ability to substitute between crops or sources of pollination and the effect such losses may have on prices and consumer or producer welfare.

Data Required: Data on the price per unit and number of units sold for crops known to benefit from animal pollination.

Examples: Matheson and Schrader (1987); Costanza et al. (1997).

Suitable to use: As it greatly overestimates the impacts of pollination services and does capture economic value, this method is not suitable for use as an economic appraisal of pollinator gains or losses and is included only for historic reference.

2.1.2. Managed pollinator prices

What it Measures: The market price of managed pollination services.

Methodology: The sum market price for the use of these pollinators in crop production is taken as the total value of the pollination service they provide, which is assumed to have arrived at an accurate price via traditional market forces (Rucker et al., 2012). This can be based on a) recorded numbers of hives actually hired (Sandhu et al., 2008) or b) the total stock of managed pollinators. To date, this method has only been applied to honey bees, although it is equally applicable to any managed species bought or rented for use as a crop pollinator.

Strengths: This method reflects the market price for pollination services as an input and is thus compatible with standard economic theory and accounting. Differences in rental price for honeybees can capture variations in the relative value the market places on pollination services to crops, theoretically linked to the market price of the crop and the relative benefits of the service. Providing that regional variations in prices are captured, this method is equally applicable at any scale. Economic modelling can also be used to predict future values based on changes in factors affecting services (Rucker et al., 2012).

<u>Weaknesses:</u> While some larger markets such as the United States have well-developed markets for managed honeybees (Rucker et al., 2012), in many counties, markets for honey bee pollination services are very small resulting in little commercial beekeeping for pollination (e.g., Pocol et al., 2012; Carreck et al., 1997). Where markets do exist, existing evidence suggests that prices are largely independent of the benefits to the crop, influenced instead by factors such as management costs, limited honey yield (or none suitable for human consumption) from some crops, the availability of commercial honey bees and the sale prices of the crop (Rucker et al., 2012; Sumner and Boriss, 2006). Other managed pollinators are bought at fixed prices per unit, which are, similar to other agricultural inputs, uninfluenced by the benefit to the crop. As such, price fluctuations will not reflect changes in the benefits of the service but the market forces affecting the price of producing and supplying these pollinators. Most significantly, this method completely discounts the benefits of wild pollinators, which are often a more significant contributor of pollination services than e.g., managed honey bees (Garibaldi et al., 2013), services provided for free by local beekeepers (Carreck et al., 1997) or pollinators managed directly by producers. Finally, managed pollinator prices alone will not reflect the benefits of varying interactions between wild and managed species that often have different, complimentary foraging habits (Brittain et al., 2013; Greenleaf and Kremen, 2006 but see Garibaldi et al., 2013).

<u>Data required:</u> Rental prices of managed honeybees and/or purchase price of other managed pollinators; estimates of the number of pollinators per hectare required for optimal pollination.

<u>Examples:</u> Burgett et al. (2004); Sandhu et al. (2008).

<u>Suitable to use:</u> This method should only be employed where a market for managed pollination services exists at a large enough level to form a substantial proportion of pollination service provision. Due to the inability to capture wild pollination services, this method is primarily suitable in systems where all pollination is provided by managed insects – for example glasshouses. Spending on managed pollinators is however likely to be important to local decision-making (Section 6).

2.2 Production functions

Production functions are analytical or statistical models that represent the impact of a quantity of an input on the quantity of an output produced in relation to all other inputs used. Two forms of simplified production function have been widely used to estimate the economic consequences of pollination services: Yield Analysis and Dependence Ratios. These methods are only partial production functions, as they do not account for the impacts of other inputs on production. Full production functions (covered in Section 2.2.3) have not been applied to pollination services to date, however a growing number of studies have advocated their use. Ultimately, none of the methods detailed below capture the true value of pollination services, only the market price of production these services underpin. In particular, as they do not capture changes in prices resulting from changing production they are mostly suitable at smaller spatial scales where yield change is unlikely to affect market price. Therefore, all production function approaches have to be combined with surplus estimation in order to assess the welfare value of benefits, particularly at wider scales.

2.2.1. Yield analysis

What it Measures: The market price of additional crop production resulting from pollination services.

Methodology: Using agronomic experiments, this method compares the average output of sub-samples where pollinators have been excluded to other sub-samples left open to pollination with the difference acting as a measure of pollination service benefits. More recent studies have expanded this approach by considering the impacts of the observed change in output on producer costs (e.g., Winfree et al., 2011) and the potential market price of production lost from deficits in pollinations services (e.g., Garratt et al., 2014). In these studies, changes in producer output or profit resulting from pollination are used as a measure of value. If data on pollinator visitation rates and efficiency are available, it is possible to divide the market price of output per hectare among particular pollinator taxa to estimate their relative importance within the system (Winfree et al., 2011). The marginal benefits of different levels of managed pollination services on yield can be captured by varying the number used within the landscape (Delaplane et al., 2013) or by assessing the suitability of local habitat to provide pollination services (Ricketts and Lonsdorf, 2013).

Strengths: Comparing open-pollinated and pollinator-excluded sub-samples, allows for an accurate assessment of the benefits from pollination to particular crops under field conditions if all other factors are equal. These studies can also capture the variation in pollination services benefits to different cultivars of the same crop and the impacts that pollinator driven changes in production will have on marginal costs (e.g., the costs of labour for fruit picking) allowing for more detailed and accurate estimates of service benefits (see Garratt et al., 2014).

Weaknesses: Despite numerous studies using this method, yield analysis is not a standardized methodology within economic valuation literature. Although most studies are use relatively consistent methods for determining pollination service benefits, variations in methodology (e.g. Ricketts et al., 2004) may affect the accuracy of estimates even in the same crop (Garratt et al., 2014; but see Vaissière et al., 2011 and Delaplane et al., 2013 for standardized methods). For example, few studies account for the impacts of pollination services on crop quality, which may result in an underestimation of benefits of pollination (Garratt et al., 2014; Klatt et al., 2014). By contrast, as this method does not account for the marginal effects of other inputs or ecosystem services on crop productivity (e.g., pest regulation; Melathopoulos et al., 2014; Lundin et al., 2013) the benefits of pollination services may be overestimated. This is particularly significant in very highly dependent crops where as much as 100% of crop market output can be attributed to pollination using this method, effectively estimating that all other inputs having no benefit. In reality, other inputs will still influence yields, even in very highly dependent crops, by affecting e.g., the size and number of fruits produced.

Data required:

- Minimum: Agronomic estimations of crop yield in both a pollinator-excluded and open-pollinated system (following e.g., Vaissière et al., 2011; Delaplane et al., 2013), crop market price per unit.
- Optimal: As above plus agronomic estimations of crop specific quality and market parameters in both a pollinator-excluded and open-pollination system. Estimates of changing management and harvest costs arising from lower yields without pollinators.

Examples: Garratt et al. (2014); Klatt et al. (2014).

Suitable to use: As they capture pollination service benefits at a very precise scale, yield analyses are most useful illustrating the benefits of pollination services at local levels. Regional scale benefits can be estimated with this method if a number of sites, covering a diverse range of environmental conditions, are sampled. At larger scales, assessment at a very large number of sites to cover variations in environmental conditions would be required.

2.2.2. Dependence ratios

What it Measures: The market price of additional crop production resulting from pollination services.

Methodology: These studies use dependence ratios, theoretical metrics of the proportion of crop yield lost in the absence of pollination, to estimate the current contribution of pollination to crop production within a region. This proportion of crop production is multiplied by the producer price per tonne (or other unit of production) to estimate the total benefits of pollination services. The expected proportion of yield lost can also be multiplied by yield dependent producer costs (such as labour costs) to estimate producer benefits. Unlike yield analyses, which utilize primary data collected from the field, dependence ratios are based on secondary data such as personal communications with agronomists (e.g., Morse and Calderone, 2000) or from literature on agronomic experiments comparing yields with and without pollination services (e.g., Allsopp et al., 2008), often using the same methods as employed in yield analyses.

Strengths: By estimating the proportion of yield lost, dependence ratio studies theoretically capture the link between pollination services and yield, without the need for further primary data collection (Melathopoulos et al., 2015). Because of the large body of literature available (e.g., Klein et al., 2007), dependence ratio studies are relatively simple to undertake and can be readily applied across a range of crops at any regional, national or international scale (e.g., Lautenbach et al., 2012).

Weaknesses: As with yield analyses (above) dependence ratio studies neglect the impacts of other inputs on crop production potentially biasing estimates upwards. Most dependence ratio studies are based on subjective personal communications which lack an empirical backing (e.g., Morse and Calderone, 2000) or from reviews, particularly Klein et al. (2007) and Gallai et al. (2009a) which, although a synthesis of available knowledge, bases many of its estimates on a small number of often older studies (see Section 4.5.2.2). Consequently, the metrics are generalized for a whole crop, regardless of variations in benefits between cultivars or the effects that variations in environmental factors or inputs have on the level of benefits (Section 4.5). When applied over large areas where multiple cultivars and environmental conditions are present, this can result in substantial inaccuracies (Melathopoulos et al., 2015). As the dependence ratio metrics typically represent a complete loss of pollination services, they inherently assume either that pollination services within the region are presently at maximum and that the studies they are drawn from compare no pollination to maximum levels, neither of which may be accurate (e.g., Garratt et al., 2014). In most cases, no assessment is made of the marginal benefits of different pollinator populations or consumers and producer’s capacity to switch between crops (Hein, 2009).

Data required: Crop yield per hectare, crop market price per unit, measure of insect pollinator dependence ratio (e.g., Klein et al., 2007).

Examples: Leonhardt et al. (2013); Lautenbach et al. (2012); Brading et al. (2009).

Suitable to use: As the dependence ratios used are often rough approximation of pollinator dependence, this method is mostly suited to illustrate the benefits of pollination services to crops larger scales. Due to their inability to distinguish differences in benefits between locations, cultivars and management and their implicit assumption that services are at a maximum level the method is less suitable for making more informed management decisions but can act as an initial estimate.

2.2.3. Production function models

What it Measures: The market price of additional crop production resulting from marginal changes in pollination services in relation to other factors influencing crop production.

Methodology: Production functions measure the role of pollination as part of a broader suite of inputs (e.g., fertilizers, pesticides and labour) and environmental factors (e.g., water) allowing for an estimation benefits relative to other factors (Bateman et al., 2011; Hanley et al., 2015). Production functions can take a number of forms depending on the relationships between the variables involved: e.g., additive functions assume that inputs can perfectly substitute for one another, Cobb-Douglas function assumes that inputs cannot be substituted at all. All of these forms assume that inputs have diminishing marginal returns – i.e., after a certain point and all things being the same, the benefits of additional units of input gets progressively smaller and may eventually become negative. By incorporating the costs of inputs (e.g., the costs of hiring managed pollinators or the opportunity costs of sustaining wild pollinators), it is possible to determine economically optimal combinations of inputs that maximize output relative to cost.

By incorporating the costs of each input, these crop production functions can accurately relate pollinator gains and losses to benefits under different management strategies. The resultant effects on output can be incorporated into partial or general equilibrium models (see Section 2.4) of surplus loss. Separate pollination production functions can also be developed to estimate the levels of pollination services provided by a pollinator community, depending on the efficiency of the species within the community and any additive, multiplicative or negative effects arising from their activities (e.g., Brittain et al., 2013) and interactions (Greenleaf and Kremen, 2006). The sum of these relationships and the crop and variety specific thresholds of pollen grains required will determine the overall service delivery of the community (Winfree et al., 2011). By focusing on functional groups of pollinators, rather than individual species, these results can also be readily transferred across regions to account for community variation. Finally, pollinator production functions can link the production of an output or a pollinator community to resources surrounding the crop (e.g., forest fragments around fields), allowing for accurate estimation of potential service delivery (Ricketts and Lonsdorf, 2013).

To date, only Lonsdorf et al. (2009) have developed a production function for pollinators, using expert opinion on habitat suitability for different pollinator groups to estimate the availability of pollinators within the landscape. However, this model does not translate the effects into economic benefits. Ricketts and Lonsdorf (2013) further develop this by linking aspects of surrounding land use with the benefits of pollination services to crops, which, although not explicitly pollinator production functions, can inform the basis of such analysis in the future. Jonsson et al. (2014) demonstrate the full applicability of the method by using field data to develop a production function analysis of the benefits of aphid pest control via natural enemies in Swedish barley fields.

Strengths: Production functions for crop yields allow the benefits of pollination services to be accurately estimated from any region with respect to local environmental and agricultural systems, assuming similar levels of pollination service. This avoids issues of over-attributing benefits to pollination services common to yield analysis and dependence ratio studies and captures substitution patterns between inputs (Hanley et al., 2015). In combination, crop and pollination service production functions allow for the most accurate estimation of the marginal benefits of pollination services across most regions where the crop is grown, providing sufficient data on local pollinator communities and agri-environmental conditions are available. Pollinator production functions linking the landscape to pollinator populations also allow estimation of the monetary value of pollinator natural capital (Section 2.6) within a landscape or even at larger scales. By directly linking pollinator populations to services and outputs, multiple production functions can be used to model the marginal effects of pressures (e.g., habitat loss) and mitigations (e.g., habitat recreation) on the economic productivity of a crop and thresholds at which shifts in pollinator communities result in collapses of service provision.

Weaknesses: Production function models are complex to estimate, requiring extensive agronomic and ecological research in order to quantify the impacts of each parameter on a given crop. A wide range of communities have to be assessed to account for the varied impacts of community composition and interactions if the effects estimated are to be transferred beyond the study sites or economic production functions are to be used to identify efficient combinations of pollination and other inputs. Although substitution patterns among inputs and ecosystem services can be modelled, further experimental data would need to be added to identify pollination service thresholds in case minimum levels of services are required for viable output.

Data Requirements: Ecological data on the impact of pollination services on crop quality and quantity relative to other inputs. Data on producer input costs and crop sale prices. Ecological data on the pollination service efficiency of different pollinators (pollen deposited and rate of visitation) relative to landscape parameters and community composition. For extrapolation: local data on pollinator community composition, environmental conditions and agricultural inputs.

Suitable to use: As they draw a strong focus on local pollinator communities, production function models are most suitable when assessing the local scale impacts of pollination services and changes in management but can be generalized for wider use if sufficient ecological data is available.

Examples: None to date but see Ricketts and Lonsdorf (2013).

2.3. Replacement costs

What it Measures: The estimated market price of artificial or supplemental pollination services.

Methodology: Typically, this is the cost of mechanical pollination via a human applicator (Allsopp et al., 2008) but can also be the costs of hiring managed pollinators to replace a known proportion of total services provided by wild pollination services (Winfree et al., 2011). Artificial pollination is often undertaken via hand pollination, using small paintbrushes to apply pollen to flowers, although a variety of mechanical methods have been developed, such as vibration wands to pollinate tomatoes (Pinillos and Cuevas, 2008). This method requires that the replacement method is i) the lowest cost replacement available ii) at least as effective as animal pollination and iii) that producers would be willing to pay these costs rather than simply switching crop (Söderqvist and Soutukorva, 2009).

Strengths: Unlike other methods, the replacement costs method does not overestimate the impacts of pollination services, as the cost estimate is independent of yield benefits (Allsopp et al., 2008). As long as appropriate labour and material capital required is known, the estimated costs per hectare can be transferred to other regions by adjusting the input costs used. Managed pollinators can also foreseeably provide pollination services to many wild plants either deliberately or as an additional side effect of pollinating crops and as such, the price of these insects can be an effective replacement cost for non-market benefits.

Weaknesses: Different replacement techniques may be ineffective for certain crops. For instance, hand pollination is not effective at replacing insect pollination in raspberries (Kempler et al., 2002) and managed pollinators are differently effective on certain crops (Delaplane and Mayer, 2000). Even where methods are effective, their viability may depend heavily upon the local availability and costs of labour. For example, hand pollination, was used on some insect pollinated fruit crops in areas of the Sichuan Province, China, affected by severe pollinator loss until rising wages made this increasingly unviable for producers, resulting in pollinated crops being widely replaced with wind-pollinated species (Partap and Ya, 2012). Therefore, it is doubtful that producers in countries with high wages would adopt these practices at all. While technological advances could produce lower cost alternatives (e.g., Sakomoto et al., 2009), limited information regarding pollination service management makes the market viability of such alternatives difficult to assess. Finally, replacement costs do not reflect the economic value of pollination services, only the market price of the replacement method. Surplus valuation models (Section 2.4.) can estimate the impact of these changing costs on producer and consumer welfare if the replacement is likely to be adopted by most affected growers.

Data required:

- Minimum: estimates of material costs and labour requirements, minimum/typical wages.
- Optimal: estimates of replacement efficiency relative to original services, indication of levels of producer willingness to pay for replacement.

Examples: Allsopp et al. (2008); Winfree et al. (2011).

Suitable to use: This method is only suitable for decision-making where the replacement method is both demonstrably effective and likely to be adopted by affected growers (e.g., they have expressed a willingness to pay to adopt it). In the case of pollination services, this is only likely to be replacement of wild pollinators by managed pollinators (e.g., Winfree et al., 2011). Otherwise, as it does not quantify the either benefits or economic value of pollination services, only the potential costs to replace it, this method alone is not suitable for public decision-making.

2.4. Surplus valuation models

While the methods reviewed previously have measured the price of various pollination service benefits to markets, economic welfare valuation methods use statistical models to estimate the impacts of changes in production on the economic welfare of producers and consumers. Welfare valuation methods can be complex and a variety of different econometric models can be used; however, for this assessment, only the methodologies as a whole are discussed.

These models can take two forms: partial or general equilibrium. Partial equilibrium models only consider what the impacts of changing supply and demand of a product will have on the market for that product. General equilibrium models however capture the impacts on other markets by considering producers' ability to substitute between inputs and consumers' ability to substitute between products.

What it measures: The economic value of pollination services to a single market (partial equilibrium models) or several interlinked markets (general equilibrium models).

Methodology: Surplus valuation models begin with the estimation of supply and demand curves for a given product using standard economic models. From these, further economic models (e.g., Gordon and Davis, 2003; Gallai et al., 2009a) are used to estimate the effects a shift in supply resulting from a change in pollination services will have on prices and the subsequent impacts upon economic welfare via consumer and producer surplus (see Section 1). As pollination service loss causes crop supply to fall relative to demands, crop prices will rise, reducing consumer welfare and making the remaining produce less competitive, relative to other produce, when sold on wider markets. This price change is quantified by the multiplying proportion of crop production lost by the price elasticity of supply (if supply changes) or demand (if demand changes): a metric of the percentage changes in price in relation to a 1% change in supply or demand, assuming all other factors influencing price remain constant. These elasticity parameters can be approximated based on past studies (Gallai et al., 2009a), estimated using time series statistical analyses (Southwick and Southwick, 1992) or by estimating arc elasticity, an average of the change in production divided by the change in price over a large number of time periods (Winfree et al., 2011).

General Equilibrium models expand this by using more complex models (e.g., Bauer and Wing, 2014) that incorporate additional elasticity parameters that capture (a) producers' ability to substitute between pollination and other marginal inputs (e.g., Marini et al., 2015) and (b) consumers' ability to substitute between different crops and different sources of the same crop (Kevan and Phillips, 2001). Consequently, general equilibrium models capture the impacts of pollinator service losses on both the affected crop market and other related markets. Bauer and Wing (2014) propose a model that includes eight substitution elasticities, including substitution between different inputs and between domestic and imported varieties of each crop. However due to limited data availability, most of these are broad estimates included for exploratory purposes.

Strengths: Unlike the methods reviewed previously, surplus valuation models estimate the true welfare value of pollination services by quantifying how much available income consumers and producers would lose or gain following a drop in pollinated crop availability. If both consumer and producer surplus metrics are modelled, these models allow for relatively accurate estimation of both marginal and total welfare changes in response to total pollinator changes in the crop market (Gallai et al., 2009a). By using multiple elasticity parameters to simultaneously model a broad range of market reactions, General equilibrium models can produce more conservative and realistic estimates of pollination service value within a single crop market, with producers potentially profiting from price rises caused by service losses in other region while consumers always suffer a welfare loss (Bauer and Wing, 2014). By modelling these values in other markets, General equilibrium models can also highlight the wider impacts of service losses and identify vulnerable secondary sectors. If applied to different locations, these models can highlight areas where losses of pollinators would cause the most significant impacts on prices and, by extension, welfare.

Weaknesses: Accurate estimation of crop price elasticity relies on significant volumes of long-term data, which may not be available in a consistent form (Southwick and Southwick, 1992). As the scale of yield losses drives surplus changes, inaccuracies in these estimates (see Section 2.2.) can result in inaccurate estimations of value. While producer surplus estimates are applicable at all scales, consumer surplus is generally more appropriate at larger scales as, imports will often compensate for small scale losses, resulting in little or no price change unless the region is a major global producer of the crop (Kevan and Phillips, 2001).

By not accounting for producer and consumer substitutions, partial equilibrium models may overestimate the impacts of pollination services on a single crop market. To date, due to the complexity of estimating both supply and demand curves simultaneously, most studies using partial equilibrium models have only estimated consumer surplus (but see Gordon and Davis, 2003). This assumes that supply has an infinite elasticity, i.e., that producers can switch freely between crops and make no profit from their productive activities regardless of price (Southwick and Southwick, 1992; Gallai et al., 2009a). In reality, most producers trading in a national or globalized market will try to generate profit (Hein, 2009) and it may be difficult or impossible for producers to switch between high-price perennial crops.

General equilibrium models require extensive ecological analyses and economic analysis from a range of different markets, in order to determine the full range of substitutions involved. This may be very difficult for minor crop markets where degrees of substitutions are unclear or for crops where global production has recently expanded significantly due to expanded market opportunities (such as biodiesel feed crops; Banse et al., 2011). The effects of multiple markets on the modelled elasticities can also make it difficult to identify which variables in the model have a strong effect on the resultant estimates of welfare change (Bauer and Wing, 2014).

Data Required:

- Minimum: Crop yield per hectare, crop market price per unit, measures of insect pollinator dependence, estimates of crop price elasticity of demand or elasticity of supply (these can be estimated with long-term data on the total market consumption of the crop and the price per unit of crop over the same time period).
- Optimal: Estimates of both crop price elasticity of demand and of supply, estimates of the proportion of total consumption arising from national production (as opposed to imports), final consumer price per unit, price elasticity of demand for end consumers.
- For GEM only: Estimates of elasticity of substitution: between local and imported supply of a crop, between the production of crops grown in the same system, between the consumption of crops consumed within the same market and between crop inputs.

Suitable to use: Surplus valuation models are suitable for measuring the benefits of pollination services to consumers only where a sizable portion of a national or international crop market is likely to be affected by a change in regional or national production unless the crop is part of a specialty market with few suitable growing sites. They are suitable to estimate the value of pollination services to producers at all scales. Partial equilibrium models of producer surplus are more widely applicable for highly pollinator-dependent crops with high capital investments and few viable substitutes for the crop itself. Due to their comprehensive assessment of markets, General Equilibrium Models are more suitable for evaluating the impacts of national or international scale policy and scenarios but may be limited by their high data requirements.

Examples: Gordon and Davis (2003); Gallai et al. (2009a); Bauer and Wing (2014).

2.5. Stated preferences

Previous methods for assessing the economic benefits of pollination services focus on the benefits of pollination services to markets, a number of methods exist for estimating the value of non-market benefits from ecosystem services (see Section 1). These methods fall into two broad categories: revealed preferences, which use existing market data to extrapolate the value of benefits derived from the ecosystem service, and stated preferences, which use surveys to elicit respondent willingness to pay for ecosystem goods and services within a hypothetical market. No revealed preference methods are considered suitable for use in valuing pollination services (de Groot et al., 2002). Stated preference techniques however are potentially useful for valuing the existence of pollinators themselves and the non-market benefits that they have marginal influence on. Unlike previous methods however, this does not capture the effect of pollinators on production.

What it Measures: The marginal existence value(s) of pollinator populations and/or non-market benefits of pollination services (e.g., the diversity of pollinator-dependent wildflowers).

Methodology: Stated preference methods are particular survey or experimental based methods that typically use questionnaires to create a hypothetical market for bundles of ecosystem goods or services, which are not traded on existing markets. Respondent preferences for different bundles within these hypothetical markets can then be estimated using discrete choice models (Bateman et al., 2011). Prices are attached to each variable to enable researchers to estimate the economic value of each bundle to different respondents. These prices framed to capture either respondent willingness to pay (WTP) to either gain an increase or avoid a loss in the goods or services or respondent willingness to accept (WTA) payments to allow that a degradation or forego a gain in the good or services.

There are several forms of stated preference methods with the two most widely used being: contingent valuation and choice experiments. Contingent valuation methods offer respondents a complete bundle of goods with an attached price and a zero cost alternative representing a degraded or current state. Choice Experiments follow similar principles, except respondents are given multiple alternatives to the zero cost option. Each alternative has different amounts of the various goods within the bundle. Through repeated observations of such choices, typically across many respondents and using different attached prices, discreet choice modelling methods can estimate the probability of respondents within the sample selecting a given bundle, depending on its price, and a typical respondent WTP or WTA value.

Stated preference techniques can be applied to estimate the existence value of pollinators by eliciting respondent WTP for the maintenance of pollinator populations (e.g., Mwebaze et al., 2010) or marginal changes in wider pollinator abundance or species diversity. Estimates of the impacts of marginal changes in of pollination services to various non-market benefits (e.g., the diversity of aesthetic wildflowers) require a further analytical step, such as dependence ratios (Breeze et al., 2015) or production functions, to estimate the contribution of pollination to these benefits.

Strengths: Stated preference methods can be used to assess the economic value of potentially any non-market benefits arising from pollination services, regardless of the existence of markets for these services. Stated preference surveys can also estimate the WTP/WTA of different groups of respondents based on their demographics (e.g., age, income, proximity to the site of proposed change etc.), allowing a more accurate extrapolation of the values estimated beyond the survey area.

Weaknesses: Like many questionnaire-based methods, stated preference surveys are often particularly costly to undertake due to the substantial pretesting required to present the scenario in an easily understood manner and the large, representative samples required for statistically robust analysis. Responses to stated preference questionnaires can also be affected by number of factors, which may cause respondents to, deliberately or unintentionally, misreport their preferences, biasing estimations of their WTP/WTA. For instance, respondents may ignore the cost of options because the payment is a hypothetical situation, expressing a greater WTP than they actually hold (e.g., Henscher et al., 2010). Respondents may also have difficulty forming preferences for unfamiliar goods such as ecosystem services, resulting in them expressing inconsistent, often extreme preferences (Christie and Gibbons, 2011). Statistical analyses (e.g., Henscher et al., 2010; Christie and Gibbons, 2011) can reduce the impacts of these and numerous other biases but extremely careful question and scenario formulation is required to identify the occurrence of these biases.

Data required: Estimates of respondent willingness to pay for preventing a loss/maintaining existing levels of pollinators/pollination services or estimates of willingness to accept a loss in pollinators/pollination service benefits, ecological estimates of the impact of pollination services on these benefits. Empirical information on the impacts of proposed scenario on pollinator populations or other non-market benefits affected by pollinators is necessary to allow respondents to make informed choices.

Examples: Mwebaze et al. (2010), Diffendorfer et al. (2014), Breeze et al. (2015).

Suitable to use: This method is suitable for assessing the marginal values of either changing pollinator populations or other, non-market ecosystem service benefits. However due to the numerous biases and uncertainties that can occur in respondent preference expression, they should only be undertaken following rigorous testing to ensure that the questionnaire can be answered accurately by respondents and require a large, representative sample of the population affected by proposed changes.

15. Table 2 – Summary of methods to assess the economic consequences of pollinator gains and losses.

Method	Strengths	Weaknesses

Crop Price	Sum market price of insect pollinated crops	- Minimal data requirements	- Does not reflect the benefits of pollination services
Managed Pollinator Prices	Sum market price of managed pollinators hired or purchased for pollination services	- Reflects the benefits of pollination in a manner comparable to other inputs - Differences in prices can reflect varying benefits	- Ignores wild pollination services - Many countries have small or no pollination markets - Prices are influenced by market forces more than benefits
Yield Analysis	Market price of output of pollinated crops vs. crop without access to pollination services based on field studies	- Directly captures benefits of pollination services - Captures more precise variations in benefit between cultivars - Can capture marginal benefits	- Only appropriate for very local scales - Requires extensive planning to capture all benefits and any pollination deficit. - Does not account for the relative effects of other inputs or ecosystem services - Only estimates producer benefits
Dependence Ratios	Total market price of crop output multiplied by a crop-specific dependence ratio (metric of the proportion of yield lost without pollination)	- Captures the varied benefits of pollination across crops - Equally applicable at all scales - Minimal data requirements	- Only estimates producer benefits - Dependence ratios may over generalize between cultivars - Does not account for the relative effects of other inputs or ecosystem services - Assumes services are currently at maximum levels
Production Functions	Models of the effects of pollinators and pollination services on total crop output	- Can accurately assess the value of pollination service stocks. - Can be used to model the effects of pressures on services - Captures the benefits of pollination relative to other inputs and ecosystem services - Can be accurately extrapolated to other locations and scales	- Requires extensive ecological data - Models can be complex - Only estimates producer benefits
Replacement Costs	The cost of replacing pollination services technologically or with managed pollinators	- Not linked to crop prices - Applies at all scales - Does not over-attribute benefits to pollination services	- Replacements may not be effective - Assumes producer willingness and ability to pay - Not linked to benefits - Tied to input and labour prices
Partial Equilibrium Models	Estimates the welfare value of price change on available income to producers and consumers of a single crop market	- Can assess consumer and producer benefits - Captures marginal benefits - Can be used to assess impacts of service loss beyond the focal region	- Very complex to estimate, especially across regions - Does not account for substitution between crops or crop inputs - Subject to the quality of data on pollination benefits - Does not account for the relative effects of other inputs - Assumes services are currently at maximum levels
Generalized Equilibrium Models	Estimates the welfare value of price changes on producers and consumer both within the crop market and across other, linked markets	- Values benefits to producers and consumers - Captures effects across and within markets - Can be applied at any scale	- Extremely complex - Many substitution effects are not yet defined - Subject to the quality of data on pollination benefits - Assumes services are currently at maximum levels

Stated Preferences	Economic survey instruments designed to estimate respondent's welfare from the maintenance or improvement of non-market benefits such as the existence of pollinators	- Values non-market benefits - Not tied to market prices or factors - Can be used to analyse public opinion	- Difficult to develop in a manner easily understood by respondents, especially if they are unfamiliar with the ecosystem service being valued. - Need to ensure a representative sample and accurate responses - Requires complex modelling to analyse - Expensive to test and implement - Monetary valuation is not always applicable

2.6. Measuring Pollinator Natural Capital

2.6.1. Overview

Although monetary valuation methods can provide a useful tool in facilitating decision-making (see Section 6), they are primarily focused on capturing the impacts of change on ecosystem service flows. Another key factor of economic systems are the capital assets that underpin economic activity which are generally considered separately from the flows they generate. There are five widely recognised forms of capital: human capital (the skills and labour within the market), social capital (institutions such as businesses or schools), manufactured capital (physical items such as tools, buildings etc.), financial capital (credit, equity, etc.) and natural capital (natural resources and ecosystem services) (Nelson et al., 2010), each of which comprises a number of assets. Capital assets represent measurable, quantifiable stocks that can produce various flows of goods and services. Pollinators are generally considered natural (wild pollinators) or manufactured (managed pollinators) capital asset that produce pollination services, a flow. Changes in capital assets fundamentally affect what flows of goods and services are available to an economy and therefore the economic activities available. This subsection reviews the links between pollinators and various capital assets that produce and sustain the economic benefits of pollination services.

2.6.2. Measuring capital

In neoclassical economics, capital assets are often components of accounting frameworks, such as Gross Domestic Product. In recent years, other frameworks have been developed to integrate natural capital assets into these frameworks using "Green GDP" measures (See Chapter 6). The main international standard for Green GDP is the UN's System of National Accounts and its associated System of Environmental Economic Accounts (SEEA) (UN, 2012). These are expressed as the monetary price of all flows arising from each stock of capital assets, including future flows via discounting (see Section 3), using market prices where available but otherwise estimating value through non-market measures (e.g., replacement costs – Edens and Hein, 2013). Typically, neoclassical economics assumes a high degree of substitution between capital assets and aims to preserve and increase the net balance of all capital collectively (van den Bergh, 2001).

Within the SEEA there are a number of challenges affecting the asset valuation of pollination services – foremost, it is important to disambiguate the benefits of pollination relative to other ecosystem service flows produced from the same assets to avoid double counting (Boyd and Banzhaf, 2007). This is particularly important when considering honeybees, which can be used as both a source of honey production and pollination within the same year but will often not because of the low nectar yields of many crops (Rucker et al., 2012). Secondly, the SEEA framework assumes that assets are controlled by an institute and are marketable. Although managed pollinators are an owned asset and patches of habitat can be owned, pollination services are almost always a public asset as access to the animals cannot be restricted (aside from enclosed crops) and their foraging habits are very difficult to control (e.g., Stern et al., 2001). Finally, the SEEA framework also assumes that assets are marketable, which is not true for wild pollination services. These issues can be partially overcome by considering ecosystems not directly controlled by private actors as a separate productive sector within the market that produces its own outputs (Edens and Hein, 2013), although care should still be taken to avoid double counting.

Monetary valuation of assets can be complimented with non-monetary quantifications of the biophysical stocks that underpin ecosystem services to provide a more holistic assessment of the impacts of capital management and support planning for sustainable, long-term management (Dickie et al., 2014). This approach is particularly advantageous because it is not tied, directly or indirectly to market prices and can be used to monitor the status and trends of those assets that are economically valuable to production. Stocks of a multiple assets of a particular capital can also be measured as an index; assessing stocks of assets in a single period relative to the same assets in a reference period (with a default value of one) (Dong et al., 2012; Nelson et al., 2010). To date, no study has expressly included pollinators as an asset in these indexes.

2.6.3. Pollinator assets

A variable number of capital assets are often required to produce pollination services and hive products. For wild pollinators, this can be as simple as having sufficient suitable habitat to support viable populations and available land, inputs and labour to produce pollinator-dependent crops. For managed pollinators, there are additional requirements in terms of human capital to breed and manage the pollinators, manufactured capital to house and transport the pollinators, social capital to maintain the knowledge necessary to breed and use them effectively and natural capital in terms of wild pollinators that form the basis of breeding stock and genetic diversity.

Quantities of available managed pollinator assets are simply the number of available managed pollinators available to a region (e.g., Breeze et al., 2014). Estimating wild pollinator assets can be more complex as their numbers are almost impossible to measure without dedicated, systematic monitoring data (e.g., Lebuhn et al., 2013). Such monitoring is presently only undertaken in an ad hoc manner in a few countries and remains focused on species presence-absence (Carvalheiro et al., 2013). Larger scale analyses models such as InVEST (Lonsdorf et al., 2009) can be used to estimate pollinator populations and pollination service potential based on habitat suitability and proximity to pollinated crops (e.g., Polce et al., 2013; Schulp et al., 2014). Although rigorous, InVEST is only capable of estimating habitat suitability, not populations of pollinators, and assumes that there is a linear relationship between habitat quality and pollinator abundance in fields. A more expansive production function approach (see 2.2.3) linking quantitative metrics of habitat quality from primary ecological research with observed abundances of different pollinators could substantially improve estimates.

Because these assets will only supply services to relatively small areas, methods to assess economic value (Section 2.4) are not generally appropriate, as the impacts on crop price from any individual asset are likely to be small resulting in little to no welfare loss. At a basic level, yield analysis can be used in conjunction with regression analysis to estimate the benefits of pollinator capital from habitats at different distances to the crop (e.g., Olschewski et al., 2006). However, detailed production function models (Section 2.2.3) are ideal as they can produce estimates that more accurately represent the quality of services produced from particular habitat patches (e.g., Ricketts and Lonsdorf, 2013). Furthermore, they can also examine the substitution patterns between pollination and other capital inputs. However, the highly specific nature of these models makes it unlikely that they can be widely employed at present, necessitating a focus on using biophysical units of pollinator capital.

Unlike other measures of pollination value, quantifications of pollinator stocks should account for potential as well as realized pollination services as assets may not always be able to provide services. For example, if arable farmers within the landscape around a source of pollinator capital (Figure 3) regularly rotate their production between pollinated and non-pollinated crops, the assets will still have value as stocks of pollination even in years where no pollinated crops are grown as they still have the potential to contribute to crop production.

2.6.4. Pollinators influence on other assets

In addition to the flow of pollination service benefits, pollinators can also contribute to the production and maintenance of other capital assets (Table 3). Foremost by contributing to the propagation of plants that provide other ecosystem services (Isbell et al., 2011; Ollerton et al., 2011), pollination has a direct influence on the quantity and integrity of a range of other natural capital assets. These plants can in turn affect wider biodiversity (e.g., insect pollinated hawthorn berries which are inedible to humans but which provide winter feed for many birds; Jacobs et al., 2009). By influencing crop productivity, pollination services can also influence the flow of available nutrients within the human diets. This can have an impact on the asset of human health (Nelson et al., 2010) by causing additional disability and death (Smith et al., 2015), which in turn affects the availability of labour within the market. The link between pollinators and human health capital is discussed in terms of disability-adjusted life years below. In many local communities, unique beekeeping knowledge is a form of social capital, helping to support diversified farming incomes and providing a source of honey and other hive products (e.g., Park and Youn, 2012; see Chapter 5 for several case studies of applied indigenous and local beekeeping knowledge). Finally, by affecting profits from the sale of pollinator-dependent crops, pollinators can potentially affect financial assets such as debt or equity among producers (which will always be measured in monetary terms). This effect has not been observed directly due to pollination services but factors such as drought that affect crop yield have been linked with substantial losses of farmer equity (Lawes and Kingwell, 2012).

2.7 Pollinators contribution to nutritional security

As reviewed in Chapter 1, animal pollinated crops are often significant sources of key nutrients in the human diet, such as vitamins A and C, calcium, fluoride and lycopene (Eilers et al., 2011). Globally, a total loss of insect pollinators would potentially cause sharp increases in the number of people suffering from vitamin A (41M-262M) and Folate (134M-225M) deficiency, particularly in Africa and the Eastern Mediterranean (Smith et al., 2015). This could potentially result in up to 1.38M-1.48M deaths from malnutrition and communicable diseases and a further 25.8M-29.1M lost disability adjusted life years (a metric measuring years of healthy, non-disabled life lost) from factors such as heart disease and strokes due to limited dietary intake of fruits and vegetables (Smith et al., 2015). Although trade and supplements could compensate for these losses at a national level, many low-income regions of the developing world with high levels of vitamin A deficiency, such as southern Africa and Southeast Asia, are strongly reliant upon animal pollinated crops to provide these nutrients (Chaplin-Kramer et al., 2014). For example, based on information from food diaries, loss of pollination services in Uganda would cause an estimated 54% increase in the rate of vitamin A deficiency in rural parts of the country (Ellis et al., 2015).

2.8 Valuing pollination services in barter economies

In many less-developed countries, portions of the population do not trade goods for money but for other goods and services, limiting the relevance of monetary valuation (Christie et al., 2012). This can be overcome by using the various production function (2.2.2) or stated preference (2.2.5) methods described above, but expressing the benefits in terms of equivalent goods or time allocation within the market rather than monetary terms (e.g., Rowcroft et al., 2006). To date, no study has examined the value of pollination services to these barter economies, despite some studies valuing pollination services in many areas where such markets exist (Partap et al., 2012; Kasina et al., 2009).

16. Table 3 – Assets that influence and are influenced by pollinator gains and losses.

Capital	Asset	Measure	Potential impacts of pollinator gains and losses
Manufactured Capital	Managed pollinator stocks	Number of honeybee colonies, bumblebee colonies or absolute numbers of other managed pollinators.	Reduced availability of economically valuable pollination services, particularly if wild pollination services are also unavailable (Breeze et al., 2014; Southwick and Southwick. 1992).
Financial Capital	Equity and debt	Monetary measures of equity and debt associated with beekeepers and producers of insect pollinated crops.	Impacts on profits can affect available financial capital for future investment and expansion, influencing their welfare over the long term (not yet observed for pollination services but see e.g., Lawes and Kingwell, 2012).
Natural Capital	Wild Pollinators	Estimates of wild pollinator population or likely populations based on suitability using e.g., InVEST models (Lonsdorf et al., 2009).	Reduced availability of economically valuable pollination services, particularly if managed pollination services are also unavailable (Garibaldi et al., 2013).
	Biodiversity	Area and population of plants affected by pollination.	Reduced levels of pollination can potentially affect plant species diversity (Ollerton et al., 2011) and wider biodiversity which relies on pollinated plants (e.g., Jacobs et al., 2009).
Human Capital	Labour (for providing services)	Available number of beekeepers and other professionals able to provide managed pollination services.	Increasing losses of managed honeybees may push beekeepers out of business if expenses from replacing lost colonies become too severe. This in turn may affect the number of beekeepers available to supply pollination services and produce hive products, even if those that do remain have a large number of colonies each (Potts et al., 2010).
	Labour (benefitting from services)	Available labour within the workforce lost through malnutrition associated with a lack of pollinator-dependent crops.	Losses of pollination services may cause a decline in the availability of nutrients in the food chain, increasing disease and mortality (Smith et al., 2015); in turn potentially affecting the availability of labour within the work force.
Social Capital	Beekeeping knowledge	Number of local beekeepers with indigenous and local beekeeping knowledge.	Pollinator losses may cause a decrease in the number of beekeepers and with this the knowledge and skills required to effectively manage honeybees to provide pollination services and produce hive products (e.g., Park and Youn, 2012).

Section 3 – Valuation across temporal and spatial scales

3.1. The importance of scale for pollination valuation

Ecosystem services, such as those resulting from pollination, are essentially the consequences of ecological processes that depend on a combination of small structures (e.g., a flower or a leaf that can live from hours to months; Kremen et al., 2007) and large arrangements (e.g., community assemblage and landscape complexity emerging along decades or centuries; Liss et al., 2013). Indeed, there are hierarchical scales in Ecology that remain independent of human decisions, despite ecologists' efforts to define and delimit scale categories (**Error! Reference source not found.**; spatial scale). Institutional scales, on the other hand, are products of human social organization. For this reason, the scales of ecological processes that affect pollination effectiveness (and thus fruit set and crop yield; see Chapter 3) and those of social and economic processes (involved in decisions and management) are seldom compatible (MEA, 2005; Vermaat et al., 2005; Hein et al., 2006; Satake et al., 2008; Abildtrup et al., 2013). Studies into pollination valuation should incorporate elements from both ecological and institutional processes (e.g., the geographic distribution of pollinator species and national subsidies for crops), with proper scale categories that allow the collection and analysis of the data necessary to quantify the economic benefits of pollination services.

The Millennium Ecosystem Assessment (MEA, 2005) recommends that assessments of ecosystem services should be conducted at multiple temporal and spatial scales. However, delimiting scale categories to value pollination (as for any other ecosystem service) is complicated because some terms are often vague and used arbitrarily and in a relative way (i.e., linguistic uncertainty, see Table 1 of the Chapter 6). Studies on pollination should define what constitutes their specific spatial or temporal scale of interest (Kremen et al., 2007; Hein, 2009; Genersch et al., 2010; Bartomeus et al., 2011; Kennedy et al., 2013), for example, it is likely that a regional economic process in Costa Rica does not have the same geographic extension as in Brazil. In addition, the definition of scale is frequently influenced by political issues, such as municipal, provincial or national boundaries, or transitory policies from successive governments with contrasting ideological positions. As such, multi- and cross-scale approaches are necessary to account for all the factors involved in pollination valuation.

Here, we adopt the MEA's definition of scale: the extent or duration of observation, analysis, or process. According to Limburg et al. (2002) and to the MEA (2005), the scales of economic systems are determined by the area and time horizon over which goods and services are traded, extracted, transported or disposed of. The temporal and spatial scales (the scale "domain") of analyses can affect valuation of ecosystem services, including pollination (MEA, 2005), because the nature of the economic value generated by pollinators (see Section 2.4) varies with the physical dimensions (space and time). For example, according to Hein (2009), in a small spatial scale (i.e., local) pollination supports farmer income, whereas in a large spatial scale (i.e., national) it is fundamental to ensure food supply. Thus, the institutions involved in decision-making that affect land management and markets change across scales; at a local scale, decisions such as type of crop and pesticide use can be made by an association of farmers, whereas national scale decisions (e.g., pesticide regulations) are usually taken by government agencies and financial organizations.

Scale mismatches in pollination valuation can occur in three basic ways: Firstly, the scales of ecological, social and economic processes that affect crop yield and production costs often differ. Secondly, the scale of the provision of the pollination service (i.e., local, see definition in Table 4) is different from the scale of decision-making by farmers (i.e., farm) and agencies involved in land and economic policies management (i.e., national; Satake et al., 2008). Finally, the chosen scale for valuation is different from the characteristic scale of the processes that affect pollination effectiveness and product prices (a methodological problem; MEA, 2005). Scale mismatches can affect the accuracy of valuation estimates and, more crucially, the distribution of benefits from management actions.

Thus, it is crucial to delimit clear scale categories for pollination valuation. Many approaches were proposed for ecological processes and ecosystem services in general (Turner, 1988; MEA, 2005; Vermaat et al., 2005; Hein et al., 2006; Feld et al., 2009; Díaz et al., 2011; Serna-Chavez et al., 2014), but no standard categorization of scales has been proposed for pollination valuation so far. It makes sense to work with scale categories that represent the extension (spatial scale) and duration (temporal scale) of processes for which the necessary data for valuation can be collected or compiled. Variables such as crop yield and price are frequently aggregated in government censuses by farm, county, politically defined sub-state or sub-national regions, provinces/states and countries, and conducted monthly or annually. The proposed scale categories for the present assessment are defined in Table 5.

17. Table 4 – The matches and mismatches between ecological and institutional (economic) spatial scales
(modified from Hein et al. 2006, originally adapted from Leemans 2000).
We adopted a particular scale for pollination valuation, and its compatibility with ecological and institutional scales varies across categories.

Ecological scale (km^2)	Institutional scale	Match	Pollination valuation scale	Compatibility
Global (> 50,000,000)	International	Yes	Global	Both
Biogeographic region (1,000,000-50,000,000)	Continental/International	No. Lack of consensus on boundaries of biogeographic regions[1] and continents. A given continent can contain more than one region and vice-versa	Continental	Institutional
Biome (10,000 – 1,000,000)	National/International	No. Biomes frequently are much bigger or smaller than the country's area	National	Institutional
Landscape (10,000 – 1,000,000)	State/Provincial	No. Lack of consensus on landscape boundaries. Catchment area is frequently used[2] and is sometimes much smaller than state/province area	Regional	None
Ecosystem (1 – 10,000)	County/Municipal	No. Lack of consensus on terrestrial ecosystem boundaries. Usually smaller than county/municipality area	Farm/Local	None
Plot (< 1)	Family	Yes	Field	Both
Plant	Individual	Yes	Not used	---

[1] See Udvardy (1975), Cox (2001), Holt et al. (2013).
[2] See Vermaat et al. (2005).

18. Table 5 – Definition of temporal and spatial scales proposed for pollination service valuation.

Temporal scales:	
Seasonal:	changes observed within one year, from periods of weeks to months, according to climate changes, pollinator phenology, the specific timing of crop production, fiscal calendar and economic events;
Annual:	changes along consecutive years, analysed with classical economic indicators that are obtained every year via institutional census and databases;
Decadal:	changes compared every ten years, using classical economic indicators, reflecting recent past and future trends that are influenced by biodiversity decline, climatic variations and economic and political crises;
Century:	changes observed or projected for more than 100 years, reflecting long-term, slow processes such as climate change and massive biodiversity loss via local or global extinctions.
Spatial scales:	
Field:	a sub-division of a farm for which data on pollinator dependency (plant's pollinator threshold, fraction of flowers pollinated by each pollinator species) are compiled;
Farm:	one productive unit composed of several fields for which data on yield and production costs are compiled;
Regional:	aggregation of farms within a well-defined region;
National:	area defined by a country's boundaries for political reasons, where the government collects data from farms in regular basis;
Continental	area defined by continents (large land masses) that contain several countries, delimited by convention or political reasons;
Global:	the geographic realm includes many countries from different continents worldwide.

3.2 – Pollinator valuation across the temporal scale

3.2.1- Rationale

The temporal scale has important strategic implications that can vary between stakeholders. For example, farmers are often more interested in longer term average yields (over several years) than short-term (1 year) maximization of yield, thus considering longer time scales is essential when linking valuation to decision-making. Another example is related to the farmer decision-making in crop choice: farmers can easily switch between different annual species; however, due to the time lag between planting and productivity, switching from annual to perennial species or between perennial crops involves a major long-term commitment. Farmers' ability to switch between crops depends also on the level of investments needed by the managing choices (e.g., irrigation costs limit the ability to switch to another crop). Thus, the temporal scale is important to consider because the meaning of the economic valuation is fundamentally scale-dependent as well as the political implications of management decisions. In indigenous beekeeping, while the majority of beekeepers take a short-term view to exploiting their seemingly abundant resources, some innovative groups and networks of local entrepreneurs have secure long-term products, processes and market sustainability laying the foundations for sustainable livelihoods and conservation (Ingram et al., 2011). In the next sub-section, we present the factors that need to be taken into account when considering the different temporal scales.

3.2.2. Temporal factors affecting pollination valuation

3.2.2.1. Price dynamics

Many economic valuation studies estimate pollinators' contribution to crop production (see Section 7). In several methods used for evaluating pollination benefits, two main variables are used: the crop price (Section 2.2 and 2.4.) and the price of beehives (Section 2.1.2 and 2.3.).

The variability of the crop price across time is driven by variation in both demand and supply of the crop. However, these two components of the agricultural market are prone to change at different intervals, some crops will change every few years while others will change several times a year, due to many factors (Drummond and Goodwin, 2014). Factors influencing demands include the price of substitute goods of pollinated crops, price of complementary goods, the consumers' income, the consumer's tastes and preferences for different crop (dependent or not on pollinators), the expectations of a pollinator decline and the demography of consumer population. Factors influencing the supply include the price of inputs, the price at a preceding period, the substitutes and their characteristics (e.g., their prices), the technology, the taxes and subsidies, the expectation about future events and the number of businesses. The complexity arising from the interaction of all these factors highlights the difficulties of predicting future crop market prices, affecting longer-term valuation estimates.

The price of hiring beehives for pollination is similarly determined through equilibrium between supply of beehive from beekeepers and the demand from farmers. Professional beekeepers[6] will also aim to maximize their benefits. However, this benefit depends on the two main goods or services that this activity contributes to produce: pollination service and honey. Thus, Rucker et al. (2012) and Muth et al. (2003) demonstrated the competition between pollination service and honey market; when the price of honey was high, beekeepers preferred to produce more honey and abandoned the pollination service. The consequences for the crops market are measurable, because the decrease of the supply of pollination service has a negative impact on the yield of crops and, consequently, the price of crops will increase. Therefore, the evolution of the beehive price is also highly dependent on other markets, making predictions similarly difficult.

It is noteworthy, that the pollination service market by bees or bumblebees seems very well developed in North America (Bond et al. 2014, Rucker et al. 2012, Burgett 2011, Sumner and Boriss 2006). However, there are very few studies analyzing this market all over the world. Breeze et al. (2011, 2014) and Aizen and Harder (2009) analyse the potential availability of honeybees in the UK, Europe and across the world respectively, in comparing the supply and the demand for beehives. These studies demonstrate the potential for expanded pollination service markets around the world but there is no proof of the existence of such well-developed market as in North America.

Because the markets are mutually influenced, policy interventions on one market will have consequences on the other. This is highlighted by Muth et al. (2003) who demonstrate that subsidies paid by the US government to beekeepers to protect them from competition with cheaper Chinese honey resulted in increased crop prices and decreased social welfare due to a loss of consumer surplus from US crop consumers.

3.2.2.2 Production effect

Economic valuation should consider the time period over which the effects of an action occur because variations in pollinator availability will change over different temporal scales as populations become more or less resilient (see Section 4). For example, when considering the seasonal scale, valuation focuses on the impact of pollinators' gain or loss on the price of the pollinator service before and after the pollination period for the beekeepers (Rucker et al. 2012). The annual scale would take into account of i) the adaptation cost of beekeepers, (e.g., buying new beehives or losses in honey production - Muth et al. 2003); ii) the farmer gains or losses due to pollination in agricultural

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[6] Hobby beekeepers are not considered in detail here because of the limited available literature on the subject.

production (Winfree et al. 2011); and iii) national indicators of annual contribution of pollinators to crop production (e.g. Gallai et al. 2009). The decadal scale is a way to incorporate the impact of the preceding year on the result of actual year, for example how the previous year's prices affect the production of annual crops in the focal year. Economic valuation should measure not only the impact of yearly pollinator variation but also the evolution of this impact (see examples in Lautenbach et al., 2012; Breeze et al., 2014 and Leonhardt et al., 2013). At a longer scale (i.e., century), economic valuation can be used to measure the sustainability of the relation between pollinators and our society. This involves both the role of bees in agricultural production and their importance on the wild nature. Within this scale, there are likely to be immense long-term fluxes in policy and agricultural technology, for example the massive shift to high intensity agriculture in post-WW2 Europe into the common agricultural policy.

3.2.2.3 Discount rate

The temporal scale has some implications on the approach used for the valuation in the case of a cost-benefit analysis (CBA). As expressed in Sections 1 and 6, CBA compared the flows of future private benefits to future social costs of doing a project. The rule of decision is that when the net present value (NPV) is positive, the action is more likely to be implemented. The procedure used for the actualization of future values to present time and conditions is called discounting.

A long debate exists around the value of the discount rate and, more precisely, on the discount of future "utility" or "welfare". It is defined by Nordhaus (2007) as: *(the) measures (of) the relative importance in societal decisions of the welfare of future generations relative to that of the current generation.* It includes also the valuation of the present generation for the benefits she will receive in their future. A discount rate of zero would mean that the value gives to future benefit of using pollination service are identical than the present one. A positive discount rate means that people give more value to consumption by their own generation compared to the future one while a negative discount rate indicates a focus more on the value for future generations.

The value of this rate has a significant importance to the interpretation of the NPV because, in the rare instances when it is negative, the weight attached to the welfare of individuals increases with time. However, discount rates are more commonly positive (Nordhaus, 2007, Chapter 7 of the TEEB, 2010). However, as mentioned by Neumayer (2007), the focus on discounting rates misses the whole issue that future degradation may result in the permanent loss of natural capital. Indeed, Nordhaus (2007) suggests that human welfare still expands under positive discount rates but does not expand optimally and may be unfair or unsustainable. With respect to pollinators, this may suggest that lower discount rate that reflect the importance of pollinator conservation for future generations may have negative impacts if they result in high levels of extinction or if pollinator abundance and diversity losses would happen in a long time. Consequently, the value given to this loss and the consequences of such loss to the future generation may be too low to affect the behaviour of the present generation. According to Neumayer (2007), in such cases it may be better to argue on the grounds of preserving natural capital before irreversible loss takes place. This argument was adapted to the specific case of pollination by Olschewski and Klein (2011). Another solution would be to use a discount rate that falls with time (Weitzman, 1993). However, evidence to argue that present generations have a strong or weak interest in preserving the future benefits of pollination service is needed to develop such revised or scaling discount rates.

3.2.2.4. Availability of long-term data sets

Good estimates of pollination value to consumer and producer welfare depend on the availability of several biological and economic data (see Section 2.4.). These databases are seldom consistent for long periods. There is also a strong interaction between temporal and spatial scales at this case, with better temporal resolution (i.e., data collected at shorter time intervals) at medium scales (national). Geographic bias is strong, with great variation in the availability of long-term national and sub-national data between countries (Lautenbach et al., 2012). At the global and national scales, most estimates used crop production, cultivated area, prices and beehive number, among others, provided by the Food and Agriculture Organization (FAO) of the United Nations over the last five decades (e.g., FAOSTAT, 2007; http://www.fao.org). For some variables, data is not available for all consecutive years for all countries, demanding statistical procedures to estimate values for specific periods (Leonhardt et al., 2013) or assuming that introduced biases are consistent in time and space (Lautenbach et al., 2012). At the sub-national level (i.e., within-country variations), the level of detail on data collection and availability in FAO databases differs substantially among countries. For example, the USA provides spatially structured data on yield whereas Germany reports yield data in highly aggregated formats (Lautenbach et al., 2012). In addition, FAO data on production prices are subdivided in two datasets, from 1966 to 1990 and from 1991-2009, which are not directly comparable (Leonhardt et al., 2013).

Long-term biological data is also difficult to obtain, since it involves many different species of pollinators and variables that are prone to temporal and spatial variations. Usually, variables such as the amount of pollen deposited by each pollinator species and the fraction of flowers each of them fully pollinate are quantified without temporal replicates. In a recent review, Melathopoulos et al. (2015) indicated the high level of uncertainty about the pollination dependency coefficients for the 10 crops with the highest aggregate benefits of pollination services. Such biological data are not available in public databases aggregating multiple countries or regions but are usually scattered on published documents regarding each specific crop at local scale (see Bommarco et al., 2012). In a recent review, Vanbergen et al. (2012) presented a list of major gaps in knowledge and research priorities to demonstrate how pollination functions differ across species and crops. Many of their recommendations include obtaining temporally replicated biological data that are important for valuation, with systematic monitoring of pollinator diversity, abundance and efficiency. This is especially necessary for those crop types with very limited knowledge and high economic importance. A summary of the most important data limitations and needs for valuing pollination services at different scales is given in Table 6 (see also: Sections 2 and 5.3).

19. Table 6 - Main data needs for accurate economic valuation of pollination services across scales.

Scale	Limitation	Data needs
Local/National	Non-market or non-monetary food consumption	-Production for own consumption or direct trade for goods and services; -Harvesting of wild fruits and honey
Local/National	Production and consumption in the secondary market	-Quantity and sale prices on the secondary markets
Local/National	Price responses to changes in supply of particular crops	-Information on consumer preferences; -Crop substitution elasticities
Local/National	Management of pollinators	-Number of beekeepers and beehives for own production and rental; -Type and extension of crops that use managed pollinators
Local/National	Seasonal variations in production and prices	-Intra-annual data on production and prices
National/Global	Standardized databases (National-among regions/states/provinces; Global –among countries)	-Standard procedures for data collection (i.e., minimum crop area considered for inclusion, area/volume units, cultivars)
National/Global	Distortion in market prices due to taxes or subsidies	-Official information on subsidies and taxes
Local/National/Global	Precise estimation of pollinator dependency is not available for several crops	-Pollination biology for different crops and cultivars replicated through time and space
Local/National/Global	Decrease in agricultural value in the case of pollination failure	-Frequency of different types of decisions of farmers and consumers responding to changes in supply
Local/National/Global	Pollination impacts on fruit quality	-Quantification pollination effects on fruit visual appearance, palatability or nutritional composition

3.2.3 Tools

3.2.3.1. Time series analysis

The term “time series” is generally used to refer to a non-random temporal sequence of values of a variable, ordered at successive and regular time intervals (Tsay, 2002; Montgomery et al., 2008). Time series analysis implies that data points taken over time may have an internal structure (such as seasonal variation) that should be considered (Montgomery et al., 2008). Thus, this approach is well suited for valuing pollination services across temporal scales, because several factors influencing pollination benefits can be addressed and forecasted. This would include ecological aspects, such as plant and pollinator phenological patterns and future trends, pollinator abundance and diversity changes, and economic variable, such as yield, production costs and prices.

There are several different types of time series analyses and models (see Tsay, 2002; Montgomery et al., 2008 for a full compendium), but most studies regarding pollination services usually adopt regression methods (**Error! Reference source not found.**). More complex time series analyses, such as stochastic simulations and complex forecasting models constitute a powerful tool to determine the impacts of pollinator loss under different land use scenarios (Keitt, 2009) but no studies have yet applied these techniques to pollination services (Section 7). Forecasting methods are frequently used in econometrics, finance and meteorology, but their use in ecological analyses is increasing (Clark et al., 2001). Availability of new data sets and the development of sophisticated computation and statistical methods, such as hierarchical models (Clark et al., 2001), offer new venues to work together with decision-makers to use forecasting techniques in pollination service assessments.

3.2.3.2. Scenarios

A way of understanding the future is to create scenarios of possible futures. The aim of scenarios is not to predict the future evolution of our society but to discuss the impact of pollinators under different possible futures of our society (MEA, 2005). More precisely, a scenario is a storyline that describes the evolution of the world from now to a possible situation (Garry et al., 2003). Scenarios are constructed to provide insight into drivers of change, reveal the implications of current trajectories, and illuminate options for action. They should compare at least two possible futures. Scenario analysis typically takes two forms: quantitative modelling (mathematical simulation models or dynamic program models) and qualitative narrating (deliberative approaches used to explore possible futures and describe how society could be situated in these futures - MEA, 2005). Qualitative deliberation can be undertaken between experts, consultants, researchers and stakeholders.

More recent scenarios often combine the qualitative and quantitative approaches; e.g., the SRES scenarios (Special Report: Emissions Scenarios; Nakicenovic et al. 2000), MEA scenarios (MEA, 2005) or ALARM scenarios (Assessing Large scale risks for biodiversity with tested methods; Spangenberg et al. 2012, Settele et al. 2012) at the global scale. Similarly, the UK NEA scenarios (Haines-Young et al. 2014) use this approach at the national scale. The SRES scenarios project the future evolution of greenhouse gases following the evolution of several driving forces, such as demographic change, social and economic development, and the rate and direction of technological change. However, these scenarios do not take into account the interaction between ecosystem services and our human society. These issues were introduced by the MEA and ALARM project.

The MEA defines four scenarios: Global Orchestration, Order from Strength, Adapting Mosaic and Techno garden (MEA, 2005). In the Techno garden and Adapting Mosaic scenarios, ecosystem services are recognized as important for society and need to be maintain and developed, whereas in the Global Orchestration and Order from Strength scenarios, they are replaced when it is possible or made robust enough to be self-maintained. Pollination services were explicitly addressed within these scenarios: Global Orchestration, Order from Strength and Techno garden projected a loss of pollination services because of species losses, use of biocides, climate change, pollinator diseases and landscape fragmentation. In the Adapting Mosaic scenario, pollination services remain stable due to regional ecosystem management programs.

However, these scenario options do not consider the economic value of these changes. By contrast, Gallai et al. (2009b) utilised existing estimates to project these values in the ALARM scenarios. Three scenarios are defined by the ALARM project (a Europe wide project on biodiversity): BAMBU, GRAS and SEDG. BAMBU (Business As Might Be Usual) refers to the expected continuation of the current land use practices. The GRAS (GRowth Applied Strategy) scenario is a kind of liberal scenario where the borders between countries are considered open to free market and the weight of restrictive policies is lower than BAMBU scenario. The SEDG (Sustainable European Development Goal) scenario focuses on the reduction of greenhouse gases and, more generally, on climate change. Using the land use change within each scenario, Gallai et al. (2009b) evaluated the changes in the economic value of insect pollinators to the Spanish and German agricultural sectors in 2020. They demonstrated that the economic contribution of insect pollinators would increase in Germany within GRAS and BAMBU scenarios, while it would remain the same within the SEDG scenario. On the other hand, the economic value would decrease in all scenarios in Spain.

The scenarios presented above are general (national or global scales) and difficult to apply to a specific region. Another study (Priess et al., 2007) used basic regression models combined with metrics derived from field data to analyse the impact of deforestation on pollination services (in terms of revenue per hectare of coffee) in north-eastern border of the Lore Lindu National Park (Indonesia). This study used four scenarios of twenty years each (from 2001 to 2021): business as might be usual (BaU), agricultural progress (AgPro), high migration (HiMig) and forest encroachment (ForEnc). Their analysis indicated that producers in the region would experience losses of between 0.3% (AgPro) and 13.8% (ForEnc) of their total revenue over a 20-year period.

These general scenarios have difficulties in quantifying the changes in both wild and managed bees across a range of possible futures and evaluating the economic consequences. The InVEST model is an interesting tool that could be integrated to the scenarios (Sharp et al. 2014). The model is based on a land use and land cover (LULC) map of natural and managed lands. Crossing different ecological and agronomic variables and land management strategies, the model predicts the evolution of wild and managed bees from a local to national level.

In brief, scenarios are a tool that aim to help guide the stakeholders for decision making in giving them the possible future state of the abundance and diversity of pollinators and the benefits of their services. However, they do not provide information on the actions to take, the instrument to use or other that stakeholder should entertain in order to undergo in one specific scenario that seems better than the others do.

3.3. Pollination valuation across spatial scales

3.3.1 Rationale

Economic analysis proposes three frameworks of analysis: macroeconomics, microeconomics and mesoeconomics. Macroeconomics is the study of the entire economy including employment, inflation, international trade, and monetary issues. It may be used to value pollinators at the national and global scales. Microeconomics deals with the economic behaviour of individuals, either producers or consumers. It may be used to value pollinators at the field, farm, and regional scales. Mesoeconomics is an intermediary point of view between micro and macro level - defined as the sum of utility of agents and firms at a local and regional level.

The distinction between microeconomics, mesoeconomics and macroeconomics is important to clarify because the analysis would change radically. Indeed, the valuation at the field, farm or even regional scale would consider two types of impacts from pollination services on crop supply: the marginal impact of these pollinators into crop production (ideally using a production function model - Section 2.2.3.) and the consequences for the marginal cost of the farmer (e.g. Winfree et al., 2011). The effect of a marginal change in pollinator populations can be directly observed in the crop market, however unless a region is a major producer of a crop, the impact is likely to be small (Section 2.4.). These analyses are limited to the crop market, whereas sometimes the stakeholder would need a more complex analysis, which considers national or global scale analyses, (i.e., macroeconomics).

At a national scale, economic analysis can consider the interaction of different markets through a multimarket analysis or a general equilibrium model (e.g., Bauer and Wing, 2014 – see Section 2.4.). These allow modelling of the impacts of pollinator loss on other sectors that do not depend on pollinators in the analysis, i.e., the ability to substitute pollinators (Bauer and Wing, 2010). Thus, the spatial scale is important to consider because the type of economic approach fundamentally depends on it. In the next subsections, we present the factors that need to be taken into account when considering the different spatial scales.

3.3.2. Spatial factors affecting pollination valuation

3.3.2.1. Loss of data quality at large scales

A frequent shortcoming of spatial analyses is that the resolution (i.e., the interval between observations) (MEA 2005) of the data decreases as the scale increases (Turner et al., 1989). One of the causes of such loss is the fractal nature of spatial information (Vermaat et al., 2005), which increases the length of borderlines when they are mapped at finer scales (Costanza and Maxwell, 1994). The same occurs for the area of a given valuable natural habitat (Vermaat et al., 2005). For example, Konurska et al. (2002) used satellite data with different spatial resolutions (NOAA-1 km and Landsat-30m), finding that the aggregated value of ecosystem services for the entire USA increased approximately three times with increasing resolution. Thus, the same problem may occur for valuation of pollination across scales using Geographic Information System (GIS) procedures.

GIS use involves obtaining and processing satellite imagery, which can be expensive and time-consuming at large scales, although these limitations are decreasing as Earth Observation data becomes more widely available. Frequently, it is impossible to distinguish very similar land cover categories using GIS, for example while most satellite images can be detecting cropland areas, they are not suited to determine crop type (Monfreda et al., 2008; see Schulp and Alkemade, 2011 for a review on the limitations of global land cover maps to assess pollination services). In this case, ground-truth validation is necessary, involving fieldwork to determine land cover, which can be logistically impeditive at national and global scales. Finally, the spatially explicit information available for valuation is usually obtained from censuses and aggregated at municipality, state or national levels by national bureaus of statistics, a procedure that *per se* causes some loss of information (Vermaat et al., 2005). Furthermore, increasing the spatial scale means using data collected by different researchers or agencies using distinct protocols, which frequently are not directly comparable (Lautenbach et al., 2012, Leonhardt et al., 2013). By contrast, GIS data are gathered by pixel or cell. Inserting such reported administrative data (crop type, production area, yields) into mapped units frequently involves several calculation steps and many assumptions (Monfreda et al., 2008) that may decrease estimate accuracy at large scales.

Some studies used GIS to calculate pollination service value at the local (including landscape) scale (Lonsdorf et al., 2009, Ricketts and Lonsdorf, 2013), but the most comprehensive attempt to map pollination benefits at the global scale was conducted by Lautenbach et al. (2012). These authors used the geographic distribution of crop areas and crop yields made by Monfreda et al. (2008) with latitude-longitude grid cells of 5 minutes x 5 minutes made possible by the use of the use of satellite. Despite the fine resolution (approximately 10 km x 10 km at the equator), this approach has some limitations, because the distribution of yield statistics into raster cells (i.e., a grid containing values that represent information) eliminates some crops for such cells (Lautenbach et al., 2012). Thus, accurate estimates of pollination benefits at national and global scales can be strongly influenced by evolving low-cost satellite technology to distinguish different crop types, and countries' adoption of standardized frameworks to collect crop data (e.g., Vaissière et al., 2011; Ne'eman et al., 2010).

An alternative to the lack of detailed data for pollination valuation at larger scales is the use of benefit or value transfer-based mapping (Troy and Wilson, 2006; Eigenbrod et al., 2010). This procedure consists of determining the value of the pollination service for a given crop type at a local scale, and using this as a proxy to estimate the value of the same crop type at other locations or at the regional or national scale. However, this procedure has several limitations related to the lack of correspondence between locations (Troy and Wilson, 2006; Plummer, 2009; Eigenbrod et al., 2010), leading to generalization errors that can only be overcome with improved spatial data and increasing the number of local replicates used for calculating the value of pollination services. A review of spatially explicit tools for pollination service valuation is available in Chapter 6 (see also a summary in **Error! Reference source not found.**), and details on geographic differences on pollinator availability, efficiency and dependency are given in Chapter 3.

3.3.2.2. Landscape design

The general effects of landscape design (spatial heterogeneity, connectivity, isolation, and proportion of natural habitats) on pollination by managed and wild species are addressed in Chapters 2, 3 and 6. Several studies have demonstrated positive effects of the pollinator habitats maintenance on agricultural yield (Ricketts et al., 2008; Garibaldi et al., 2011; Ferreira et al., 2013; Kennedy et al., 2013). However, sparing natural vegetation in a given farm incurs an opportunity cost from not using that area for crop production or other economic activities. Thus, management decisions regarding land use can be greatly improved by cost-benefit analyses of trade-offs between different ecosystem services (Farber et al., 2006; Nelson et al., 2009; de Groot et al., 2010). Most information on trade-offs between economic gains from forest conversion and pollination services comes from case studies on coffee production, usually at the local and regional scale (Priess et al., 2007; Ricketts et al., 2008; Olschewski et al., 2006; Olschewski and Klein, 2011). For example, Olschewski et al. (2006) compared the net welfare of increased coffee production by maintaining nearby forests versus converting such forests to alternative crops in Ecuador and Indonesia. In both regions, crop revenues exceeded coffee pollination values, generating incentives to convert forests, even if owners would be compensated for pollination services. However, it is important to highlight that i) pollination is only one of the many ecosystem services provided by natural vegetation; and ii) that less impacting management systems (e.g., agroforestry, rustic practices) are good candidates to reconcile ecological, economic and cultural values (Priess et al., 2007; Olschewski and Klein, 2011; Vergara and Badano, 2008; see also Chapter 5).

Environment friendly production systems (shaded coffee and cacao and other agroecological practices; Mas and Dietsch, 2004; Priess et al., 2007; Kremen et al., 2012) can be economically viable at the producer level if "green certificates" (e.g., organic and fair-trade) enhance landowners net revenues (Gobbi, 2000; Perfecto et al., 2005). However, cost-benefit analyses for coffee and other production indicated that only an elevated consumer's willingness to pay high prices for green products could generate the necessary economic incentives for forest preservation (Benítez et al., 2006; Olschewski et al., 2006; Bateman et al., 2015). Thus, direct payments for ecosystem services, accounting not only for pollination but also for carbon sequestration, soil conservation, water quality and biological control, among others, are probably necessary to sustain biodiversity-friendly production systems. This seems to be true for a high-price commodity with a global market such as coffee, but studies on other crop types are still lacking.

The implementation of payments for ecosystem services generated by biodiversity-friendly landscape planning has been controversial and difficult for many reasons (Landell-Mills and Porras, 2002; de Groot et al., 2010, Kinzig et al., 2011; Lockie 2013). The economic impacts of wild pollination are still not fully incorporated into market schemes (especially the stock market), and natural vegetation is usually evaluated only it's benefits to for carbon storage and timber production (De Konig et al., 2005; Satake et al., 2008; Phelps et al., 2010). Thus, mechanisms for income generation are still lacking (Olschewski and Klein, 2011). Another problem is the mismatches between the scale at which the pollination service is provided (e.g., regional) and the scale of landowner management decision (i.e., farm); and between the scale of pollination provision and the global scale of carbon storage, which can create inequalities among landowners with and without forest areas (Satake et al., 2008). Payments for ecosystems services are often criticized on the ground that they commodify nature (Liverman 2004; McAfee and Shapiro, 2010; Gómez-Baggethum and Perez, 2011; Adams, 2014). Several authors have expressed concerns that this could have severe social-environmental consequences particularly, reducing protection efforts for species/habitats with little to no economic importance, eliminating of not-for-profit conservation values and abandoning traditional management practices (Wunder, 2006; Kleijn et al., 2015; Wilcove and Ghazoul, 2015). Some alternatives to direct payments for ecosystem services that promote a non-utilitarian view of nature, such as land use planning, environmental education and community-based approaches are presented in details in Chapter 6.

20. Table 7 - Summary of factors that affect valuation methods across scales and the tools to apprehend such effects.

	Factors affecting valuation across scales	Tools to apprehend scale effects	Examples
Temporal scale **Rationale: different demands across institutional levels (e.g., farmers x government)**	- Price dynamics - Production effect - Discount rate - Availability of long term data sets	- Time series analysis	- Regression methods[1] - Stochastic simulations[2] - Forecasting models[3]
		- Scenarios	- SRES[4] - MEA[5] - ALARM[6] - UK NEA[7]
Spatial scale **Rationale: micro vs. macroeconomics valuation**	- Loss of data quality at large scales - Landscape design	- GIS techniques	- Maps[8] - Landscape metrics (fragmentation, connectivity)[9]
		- Spatially-explicit frameworks	- Polyscape[10] - InVEST[11] - ARIES[12] - Envision[13] - Markovian models[14] - Niche modeling[15]

[1] Gordo and Sanz, 2006; Aizen et al., 2008; 2009; Aizen and Harder, 2009; Lautenbach et al., 2012; Bartomeus et al., 2013; Leonhardt et al., 2013[2]; Keitt, 2009[3] ; Clark et al., 2001[4] ; Nakicenovic et al., 2000[5]; MEA, 2005[6] ; Spangenberg, 2007 ; Gallai et al., 2009b; Spangenberg et al., 2012; Settele et al., 2012[7]; Haines-Young et al., 2014[8]; Schulp and Alkemade, 2011; Lonsdorf et al., 2009; Lautenbach et al., 2012; Kennedy et al., 2013; Ricketts and Lonsdorf, 2013[9]; Ricketts et al., 2004; Garibaldi et al., 2011; Ferreira et al., 2013; Kennedy et al., 2013[10] ; Jackson et al., 2013[11]; Lonsdorf et al., 2009; Nelson et al., 2009; Tallis et al., 2011; Ricketts and Lonsdorf, 2013; Zulian et al., 2013[12] ; Bagstad et al., 2011; Jackson et al., 2013[13]; Bolte et al., 2007; Hulse et al., 2008[14]; Satake et al., 2008[15]; Settele et al., 2008; Giannini et al., 2013; Polce et al., 2014.

Section 4 – Valuing pollination service stability

4.1. Overview

Economic analysis and valuation aim at comparing options to develop quantitative indicators of the impacts of decisions and policy-making. Typically, economic valuation tends to assume that the consequences of pollination service loss are precisely known. In reality, things are usually more complicated and decision-making is confronted with stochastic relations between events. This gives rise to the concepts of uncertainty, risk, vulnerability and resilience (collectively referred to, for the sake of brevity, in this assessment as Stability), all of which can significantly affect the economic value of pollinator gains and losses in decision-making.

- Uncertainty is defined by the UN approved ISO 31000 framework as "the state, even partial, of deficiency of information related to, understanding or knowledge of, an event, its consequence, or likelihood." (ISO, 2009). Numerous forms of uncertainty (see Chapter 6) affect pollinators and pollination services but within economic valuation, uncertainty usually arises from stochastic factors, those that derive from the natural variability within a system. For example, increasing distance from habitat has been linked with increasing variation in the level of pollination services provided to crops (Garibaldi et al., 2011).
- Risk is defined as the "effect of uncertainty on objectives", typically measured as a composite of the magnitude of impacts and the probability of them occurring (ISO, 2009). Economic theory usually assumes that people are either risk-averse (avoid risks), risk-neutral (indifferent to risk) or risk-loving (seeking risk) in different situations. Economic analyses often assume that agents are risk-averse and will therefore typically make decisions that have lower risks than other decisions (i.e., are either less likely to occur and are less likely to be negative) than other decisions. Changes to pollinator populations can increase the risk of inadequate pollination service delivery if key species decline. Managed pollinators can reduce these risks but over-reliance can impose other risks to growers should production costs rise (Rucker et al., 2012). By increasing the flow of genetic materials within plant populations, pollination can also increase resistance to disease, reducing the risks of yield loss from disease outbreaks. For example, Mexican production of bat pollinated Agave cacti, farmed as the basis for tequila production, has suffered substantial losses from outbreaks of vascular wilt (*Fusarium oxysporum*) due to a reliance upon cloned varieties with little resistance to the fungus (Ávila–Miranda et al., 2010).
- Vulnerability measures the degree to which a system is susceptible to and is unable to cope with adverse effects (McCarthy et al., 2001). Vulnerability is a function of three elements: exposure, sensitivity and adaptive capacity (Turner et al., 2003). In the case of pollination, the exposure can be represented by the dependency of a plant upon pollination to reproduce or, for crops, the change

in crop yields or economic outputs affected by changes in pollinator populations. The sensitivity is indicated by the shape of the relationship between pollination and benefit (linear, concave or convex yield loss). The adaptive capacity of the cropping system can be approximated by the capacity of alternative techniques to substitute animal pollinators (e.g., substituting managed pollinators for wild species or increasing other inputs).

- Resilience (in the context of social-ecological systems[7]) refers to the capacity of a system to return to its original state after being disturbed and the magnitude of change it can sustain before it changes to a radically different state (e.g., Berkes et al., 2003; Folke, 2006). In the case of pollinator communities, resilient communities are those that can continue to provide a reliable level or services even in the case of temporary or permanent loss of major pollinators. Communities that are more resilient will recover from temporary declines in key species (e.g., temporary population declines due to extreme weather) than less resilient communities (which may permanently cease to provide any services).

4.2. Incorporating stability into standard valuation methods

Although variation in pollination services can result in uncertain benefits (e.g., Bauer and Wing, 2014), to date, most valuation studies have not considered issues of service variability within the benefits of pollination services (Melathopoulos et al., 2015), often only providing a single estimate of benefits rather than a range of possible values (see Section 7). Uncertainty has been incorporated into some existing dependence ratio and surplus analysis studies by assessing the impacts that variations in certain factors, such as dependence ratios (Lautenbach et al., 2012), price elasticities (Gallai et al., 2009a) or substitution parameters (Bauer and Wing, 2014) can have on estimates of value. In yield analysis, uncertainty can be incorporated by estimating value subject to inter-site or inter-annual variance in the benefits observed. The production function method can directly capture the effects of variation in several aspects of pollinator communities on service delivery, identifying how community variations may cause the output to vary.

Risks from potential honeybee losses have been incorporated into some dependence ratio (Section 2.2.) and surplus analysis (Section 2.4) studies (e.g., Cook et al., 2007; Southwick and Southwick, 1992) using hypothetical or expert derived weights that reflect the capacity of wild pollinators to replace honeybee losses. In these studies, the risk value of honeybee loss is the value of production that cannot be compensated for by other pollinators. However, these weights are subject to many of the assumptions of dependence ratios themselves and often stem from the assumption that honeybees are presently the majority pollinator, which may not be the case (Garibaldi et al., 2013). Within stated preference studies, risk can be applied to non-market benefits by including an attribute representing the probability that the benefits will not be delivered as described. Vulnerability of producer benefits

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[7] The concept of resilience has also been used for many decades in material sciences or in psychology.

can be quantified by estimating the proportion of the total economic value of the agricultural sector (Gallai et al., 2009a) or agricultural GDP lost in the event of pollinator community collapse (e.g., Lautenbach et al., 2012).

4.3. Additional methods for assessing the economic value of stability

A number of methods from the wider ecological economics literature are also suitable to specifically assess the economic value of stability and resilience in benefits from pollinators, the most relevant of which are reviewed below. These values are generally considered distinct from the direct use value of service benefits themselves but can draw upon methods to estimate use values, becoming an additive factor in assessing TEV by quantifying the uncertainty in management decisions that will affect pollinators and services (Armsworth and Roughgarden, 2003). For each method this subsection reviews: what it measures (uncertainty, risk resilience or vulnerability), an overview of the methodology, including its strengths and weaknesses, links with the main methods for valuing the impacts of changes in pollinator populations (Section 2) and the data required. Table 8 summarises the methods and their strengths and weaknesses.

4.3.1. Portfolio models

What it Measures: Uncertainty (the degree of uncertainty of service provision) and Risk (the costs of maintaining communities that provide different levels of service stability).

Methodology: Portfolio models use various econometric models to estimate an economically optimal collection of assets, including their associated costs, which minimize the variability of the output and with it the risks to producers (Admiraal et al., 2013). This method has not yet been applied to pollinator populations but has been adapted to assess the effects of soil natural capital on crop production (Cong et al., 2014a). This methodology could be used develop optimal portfolios of pollination service assets, such as managed pollinators or specific habitat types to support particularly effective wild pollinators, that have low risk of service collapse. Alternatively, this method can be used to determine portfolios of the suitable foraging resources for honey production within a year. Portfolios may vary depending on the risk aversion of the agent expected to make the change (Cong et al., 2014a) and costs (e.g., the opportunity costs of habitat management) should factor into portfolio analysis as portfolios based on benefits alone may differ strongly compared to cost-benefit portfolios (Ando and Mallory, 2012).

Strengths: Portfolio models can be projected across longer time scales in order to minimize long-term risks (Cong et al., 2014a). Portfolio models also allow for varying degrees of producer risk aversion to be incorporated (Cong et al., 2014a), allowing research to present a range of options for management to producers (see Chapter 6). These can in turn be incorporated into map based optimization models as constraints (e.g., Cong et al., 2014b) to determine the optimal distribution of assets within a landscape e.g., where management measures should be placed on a farm). Model constraints may also be applied to prevent a portfolio over-emphasizing wild or managed pollinators, as the large-scale population collapses of one could be difficult to compensate with the other (Garibaldi et al., 2013). More hypothetically, portfolio models can build on production function methodologies to better optimize spatial placement of pollinator assets relative to other assets.

Weaknesses: Pollinator populations can vary strongly between years and landscapes, causing fluctuations in risk on an annual basis. Capturing these fluctuations, and the associated risk to producers, requires complex modelling that should account for other inputs (e.g., Production function models). Furthermore, no portfolio analysis model has actively considered how producer risk-aversion may change over years, making it difficult to estimate optimal portfolios over longer time periods. Portfolio models typically assume that assets do not interact with one another (Koellner and Schmitz, 2006) however, this is rarely true for pollination services where different assets (pollinators) can interact to affect service provision (e.g., Greenleaf and Kremen, 2006) and long-term risks via pathogen spill over from managed to wild pollinators (e.g., Meeus et al., 2011). Although costs can be determined for managed pollinators, it can be more difficult or even impossible to estimate the costs of wild pollinators at a group or species level. Finally, as land use, land management and producer risk aversion can vary strongly; portfolio models are rarely appropriate for larger scale analyses.

Links to primary valuation methods Portfolio models would be most effectively used an extension of the production function method (Section 2.2.3). By identifying links between assets (e.g., pollinators within a community, pollination as one of a number of inputs into crop production) and outputs (the economic value of pollination services), it is possible to determine the combination of assets that produces the lowest variation in outputs (Koellner and Schmitz, 2006). As service spill over will be affected by habitat configuration, this method should be combined with ecological models (e.g., Cong et al., 2014b) to determine how the configuration of interventions could affect variance in service delivery. Where links between the pollinator community and pollination services are not explicitly established, yield analysis (Section 2.2.1) or dependence ratios (Section 2.2.2) will be required to quantify the economic benefits and variance of each portfolio. At a minimum, yield analyses can be used to infer the benefits of individual habitat patches, but this will be subject to greater uncertainty. Portfolio models could also use information from plant-pollinator network analysis and stated preference surveys (Section 2.5.) to identify possible co-benefits from the portfolio. For example, stated preference surveys into the value of aesthetic wildflowers could be sued to weight the selection of flowers and the placement of flower rich habitats within a landscape to optimize both pollination services to crops and the aesthetic value of the habitat.

Data required:

- Essential: Production function data on the effects of different habitats on pollinator communities and/or the impacts of individual pollinators within a community on pollination services. Information on the costs beneficiaries incur when using an asset.
- Desirable: Measures of producer risk-aversion, projected availability of assets (e.g., habitats or managed pollinators).

4.3.2. Sustainable livelihood framework analysis

What it Measures: Vulnerability (local capacity to adapt to significant losses of pollinators).

Methodology: Sustainable livelihood framework analysis uses biophysical measures of various capital assets (Section 2.6.) to determine how vulnerable a region is to a particular change (e.g., a marginal loss of pollinators) by evaluating whether the available capital within the region would be able to fully substitute for any capital affected by that change (Tang et al., 2013). This method has not yet been applied to pollination service losses but has been used to assess the impacts of climate change on rural communities (Hahn et al., 2009; Nelson et al., 2010). Alternatively, biophysical metrics of pollination service stocks could be built into an assessment of regional vulnerability to global pressures including climate change. The assets studied are selected based on how likely they are to be affected by the change in question and their effectiveness as substitutes for other capital assets in providing a service (Nelson et al., 2010). Biophysical measures are often derived from existing data sources such as the national statistics or from primary survey data (e.g., Hahn et al., 2009). As capital assets will often have substantially different units of measure (e.g., number of pollinators, area of forest etc.), an index is created for each asset, which is then usually compiled with other assets of the same capital type (e.g., natural capital). All capital indexes are then combined into a composite index that captures the total access to capital, the capacity of capitals to substitute for each other and the relative access to each capital; for instance, an area with high financial capital but little access to other capital would score lower on the index. The lowest-scoring regions are therefore the most vulnerable to the change (Nelson et al., 2010). Indexes typically weight all capitals equally (e.g., Hahn et al., 2009; Bryan et al., 2015) but some can use more specific weights based on statistical modelling (Nelson et al., 2010) or assigned directly by participants (e.g., Below et al., 2012).

Strengths: By incorporating non-monetary measures of capital, this method is particularly suitable for use in areas where monetary markets for pollination service benefits are minor, incomplete or absent (e.g., communities that do not trade crops for money). This also allows for the identification of key aspects of vulnerability to a region, such as the relative availability of particular capital that could become important under an alternative scenario. It can also be readily applied at any scale from households (e.g., Below et al., 2012) to regions (e.g., Hahn et al., 2009) and under a variety scenarios as long as the effects (positive and negative) on different capitals and the trade-offs between them can be accurately estimated (Nelson et al., 2010). It may also be appropriate to justify action where benefits are unknown but policy actions (and therefore preferences) are.

Weaknesses: This method does not inherently capture the benefits provided the focal capital, only the level of stocks that generate it and therefore does not fit into the typical cost-benefit paradigm (Section 1). As such, it should be coupled with assessments of the local benefits that are provided by the asset in its present state (e.g., Section 2) in order to determine appropriate responses. This method primarily functions by compiling different assets into one or several other indexes which may mask relationships and trade-offs between different capitals; improving an index of natural capital by planting large areas of forest on uncultivated land may improve overall natural capital at the expense of wider biodiversity. The methods used to weight the index used in sustainable livelihoods analysis often introduce assumptions about the relative substitution between capitals with e.g., equal weighting assuming that all capitals are perfectly substitutable (Hinkel, 2011). Often the link between the capitals and the adaptive capacity of the affected region is abstract, taking little account of how the capital is actually used (Below et al., 2012). Furthermore, sustainable livelihoods analysis inherently assumes that all capital can reduce vulnerability to a change and is substitutable. However, in certain highly pollinator-dependent crops, fruit set cannot be initiated without animal pollination (Klein et al., 2007) and labour costs may prohibit the use of artificial pollination. Although substitutions between different forms of pollination service assets are possible, these are often imperfect (Garibaldi et al., 2013) and may not be effective in the case of technological replacements (e.g., Kempler et al., 2002). Although technological innovation may increase the capacity for capital to substitute for other capitals, the occurrence and adoption of this innovation is almost impossible to predict. Even where substitution is viable, estimating the quality of substitution between forms of capital is extremely complex and impossible to accurately quantify without strong data (Nelson et al., 2010).

Links to primary valuation methods Quantitative biophysical measures of managed or wild pollinator assets (see Section 2.6) can be included in framework without any modification as part of natural and manufactured capital indices. However, care should be taken to separate hives managed by professional and amateur beekeepers as changes in pollinator capital have different trade-offs to wider capitals. For example, price shifts for managed pollinators for instance may affect the financial capital of professional beekeepers (Sumner and Boriss, 2006) but not amateurs that do not typically receive payment for pollination services.

Data Required: Measures of all relevant assets and their distribution within a region at a spatially explicit scale.

4.3.3. Resilience stock

What it Measures: The monetary value of resilience (the capacity of the pollinator community to withstand and recover from pressures that affect its capacity to provide benefits).

Methodology: This method assesses the long-term trade-offs and benefits from different managements on service availability by considering resilience (Section 4.1.) as a separate asset that can be affected by pressures and mitigations (Maler et al., 2009). The impacts of a pressure or mitigation on resilience can be measured as a change in the marginal shadow values of the service (Bateman et al., 2011). Shadow values represent the long-term benefits of ecosystem services from natural capital to society, including their potential future values. The shadow value of an ecosystem service can be estimated by applying a discount rate (see Section 3.2.2.3) to estimates of the future value of the ecosystem services generated by the capital asset; e.g., the value of pollination services now and in the future assuming similar land use. The resilience of pollination services to crops and wildflowers will be influenced by the abundance and diversity of key functional pollinators (Winfree and Kremen, 2009). Higher abundances of key species and a higher diversity of potential service providers will increase resilience by increasing the community's capacity to adapt to change (e.g., Brittain et al., 2013). Thresholds for resilience, the point at which an asset would be unable to return to its original state if a pressure were to degrade its functioning, will therefore be the point at which a pollinator community is unable provide services following a reduction in a key species or group. These thresholds are presently unknown, although ecological network analyses may provide a starting point for future evaluation.

Strengths: The economic value of resilience as a stock inherently captures the value of insurance; the mitigating effect of resilience upon producer wellbeing, which can be estimated separately utilizing specialized models (Baumgartner and Strunz, 2014). As a capital asset it can be readily incorporated where monetary markets for crops are absent, with the shadow value simply becoming the projected stock of the resilience asset.

Weaknesses: This method is highly influenced by the discount rate applied to create the shadow value. In the case of pollination services, this will depend on both the projected future benefits and, for crop pollination, the discounted price of the crop in future periods. These prices are likely to be very difficult to project and discount rates can be very difficult to estimate (Section 3.2.2.3). By applying this method to a single ecosystem service, this method may over-state the impacts of pollinator gains and losses in isolation. In reality, ecosystem services and inputs may compensate for one another (e.g., pollination services increasing yield in certain oilseed rape, *Brassica napus*, varieties in the absence of fertilizer – Marini et al., 2015), necessitating a complex, whole systems approach that considers multiple services in a single resilience stock. Insurance values are inherently linked to user preferences for risk aversion, such as the maximum amount of pollinator-dependent yield loss a producer is willing to accept before switching crops (e.g., Gordon and Davis, 2003), which should be estimated separately to extrapolate insurance value (Baumgartner and Strunz, 2014). Most critically, the threshold levels of pollinator diversity and abundance needed to provide economically viable levels of pollination services remain unknown due to a lack of large-scale community monitoring of pollinators or pollination services.

Links to primary valuation methods: The shadow value of pollination services will have to be derived from either a production function (Section 2.2.) or, ideally, surplus valuation methods (Section 2.4.). Production functions can inform the marginal effects of changes in the pollination service community, including the relative contribution of different species, identifying thresholds for the system studied and the value of benefits potentially lost by a composition change. Finally, stated preference methods will be required to assess the non-use value of pollinator resilience stocks.

Data required: Threshold levels of pollinator abundance and diversity required to provide pollination services to a particular plant, estimates of the present value of pollination services, projections of future benefits and a suitable discount rate.

21. Table 8 – Summary of methods and their strengths and weaknesses for assessing the economic value of uncertainty, risk, vulnerability and resilience.

	Method	Strengths	Weaknesses
Portfolio models	Statistical models are used to construct an optimal portfolio of assets (pollinators or habitats) that minimize variance in expected benefits	- Account for varying degrees of producer risk aversion - Readily incorporated into long term management and spatial planning	- Often highly complex to estimate - Requires substantial and in depth ecological and economic data, ideally from production function analyses to capture changing risks - Assumes that assets do not interact with one another
Sustainable Livelihoods Framework Analysis	A range of complementary capital assets are quantified and summed into an index to identify regional vulnerability to a change	- Does not require the presence of monetary markets - Applies at all spatial and temporal scales - Can be used without adaptation for any policy scenario	- Does not reflect the benefits of capital changes, only the costs and trade-offs on capital. - Needs to be combined with measures of benefit to facilitate decision making - Indexes of capital can mask trade-offs between different assets - Pollination cannot always be substituted for and many substitutes are imperfect - Weighting of the index can be difficult and introduce assumptions - Many indicators are only abstract representations of adaptability
Resilience Stock	Resilience is quantified as a stock that can be quantitatively degraded like other capital assets	- Does not require the presence of monetary markets - Captures the value of service insurance	- Monetization is highly dependent upon discount rates which are difficult to estimate accurately - Does not account for service substitution - Difficult to extrapolate from source site

Section 5 – Knowledge gaps

5.1 Overview

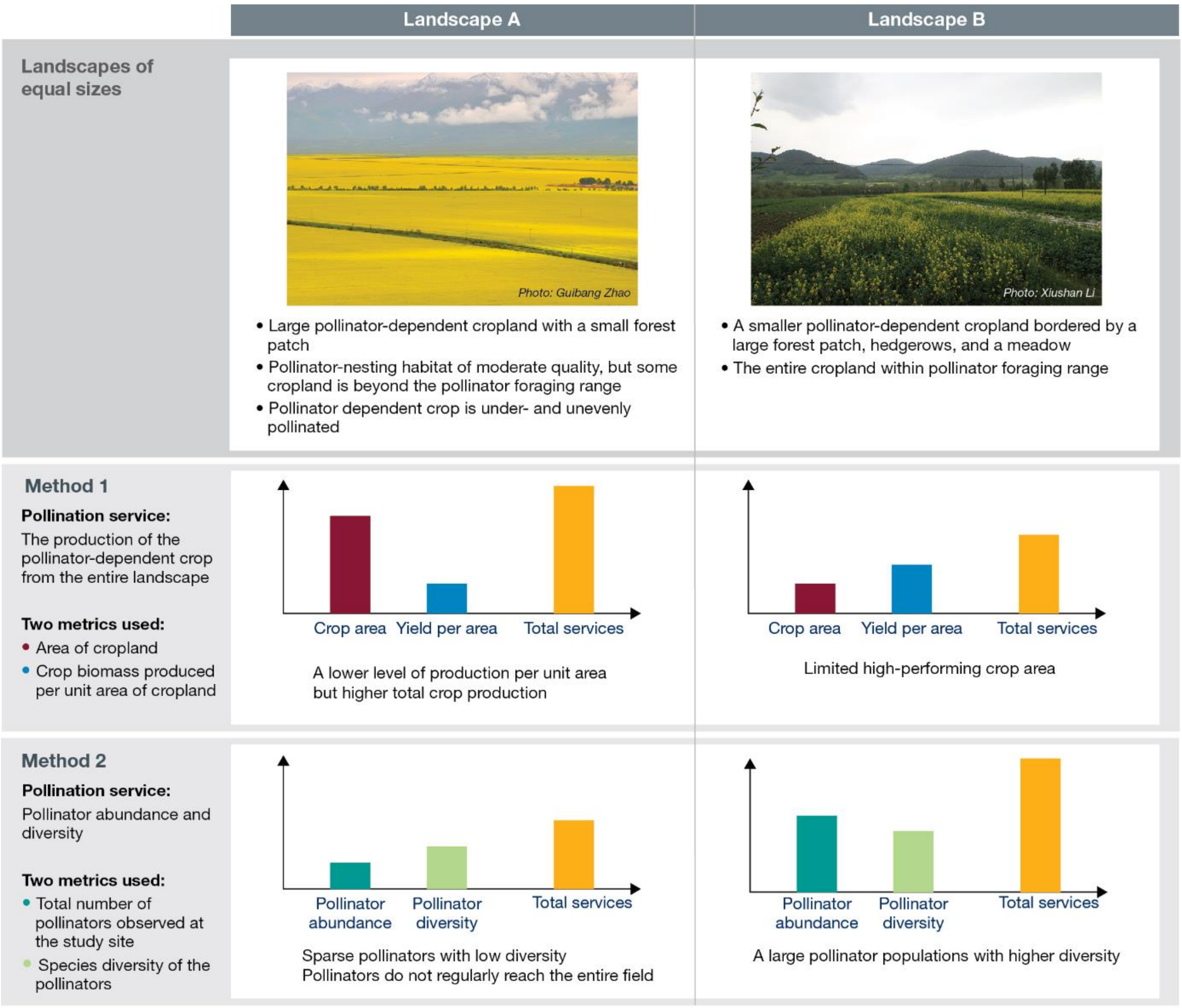
There is a consensus that biological knowledge gaps are an important limitation to economic analyses of the benefits of pollinations services (TEEB, 2009; Vanbergen et al., 2012; Dicks et al., 2013). The absence of biological information directly affects each of the methodologies and frameworks used or proposed to evaluate the impact of pollinators' declines (see Section 2). For example, there is only limited information about the effect of habitat fragmentation in pollination dynamics (Hadley and Betts, 2011) or landscape effects (Viana et al., 2012) and variability in the concept of pollination deficit (Liss et al., 2013). There are also biases in global sampling towards large-scale farming in temperate regions (Steward et al., 2014), bias in sampling examples (Archer et al., 2013) or the interface with climatic change (Prather et al., 2013). An urgent priority and research challenge will be to establish how multiple pressures affect pollinators and pollination under continuing environmental change and their subsequent economic impacts (Vanbergen et al., 2013). The relationship between crop management practices and the response of crop yield to pollination is complex and, in the vast majority of cases, completely unknown and for most regions of the world. For most wild pollinator taxa, we have no data as to whether there have actually been declines (Goulson et al., 2015). While the contribution of wild bees to crop production is significant, service delivery is restricted to a limited subset of all known bee species and conserving the biological diversity of bees therefore requires more than just ecosystem-service-based arguments (Kleijn et al., 2015).

Although biological knowledge gaps remain the primary factor limiting accurate valuation of pollination services, a number economic knowledge gaps fundamentally also limit the current scope of valuation studies. As such, the current knowledge base is likely to neglect certain beneficiaries and may over- or under-estimate the impacts of pollinator gains and losses. This section critically reviews a number of the key knowledge gaps affecting accurate estimation of the economic impacts of pollinator gains and losses, highlighting which methods are primarily affected (Section 2 and 4) and what the impacts of this incomplete information are likely to be.

5.2. Agronomic/ ecological knowledge gaps

5.2.1. How do we measure pollination services?

In a review regarding how pollination is measured in published works, Liss et al. (2013) found that pollination was most often defined by crop yield (41%), followed by pollinator abundance/diversity (31%), pollen transfer (21%), pollinator visitation (13%), and plant fitness (9%). Lack of robust, reliable and consistent indicators for pollination services could produce contradictory or inaccurate results by lack of understanding of the relationship between pollinator identity, abundance and diversity and service level (Liss et al., 2013).



34. Figure 3 – Comparison of different methods for evaluating pollination services (Liss et al., 2013)

Different ecosystem service definition and metric selections could hypothetically alter study conclusions about pollination service provision and confound comparisons among studies. Pollination services are estimated to be high in Landscape A when using a crop yield definition but low based on pollinator abundance and diversity, while the opposite is true in Landscape B. Production function models in these landscapes would over- or under-estimate pollination service benefits and may in turn drive sub-optimal decision making if farmers were to add or not add mitigation measures respectively (modified from Liss et al., 2013).

Methods affected: Production Functions (Section 2.2.3), Yield analysis (Section 2.2.1), Stated preferences (Section 2.5.).

Impacts: A robust metric of pollination services is essential to accurately estimate the pollination service provided by pollinator communities. Inaccurate measures can potentially cause over- or underestimation of benefits. In crops, this is particularly important in production function analyses, which should capture the effectiveness of different pollinators within a community in providing pollination services. An ideal measure would be to estimate the pollen deposition by each species up to a threshold required for fruit or seed set (Winfree et al., 2011). However, although standardized frameworks exist to measure this in the field, it is a very labour intensive process (Vaissiere et al., 2011; Delaplane et al., 2013). Assessments of how well pollination service metrics correlate with one another could therefore allow for simplification of fieldwork and greater comparability between studies. Different metrics may also be required for valuing different benefits; for crops the level of pollen deposition is key to ensuring optimal economic output (Winfree et al., 2011), however for aesthetic wildflowers, the rate of legitimate visits to aesthetically valuable species rather than other species may be more important.

5.2.2. What are the benefits of pollination service on the final crop output?

Much of the current understanding of pollination service benefits is based on studies that solely focus on changes in initial fruit/pod set rather than final producer profit (including costs) and are often assumed to be representative of all cultivars of a crop (Bos et al., 2007; Garratt et al., 2014). In reality, crop quality can be a significant component on the markets for a particular crop increasing the sale price (e.g., apples - Garratt et al., 2014) or the quantity of extractable materials (e.g., oilseed rape – Bommarco et al., 2012). In some crops a minimum quality threshold is often required for a crop to enter a specific market, for example, in the European Union strawberries must be of a particular shape and size to enter the primary produce market (Klatt et al., 2014), with others entering a lower quality secondary market for processing. Similarly, recent studies have demonstrated substantial variations in the benefits of pollination services to different cultivars of the same crop (e.g., Hudewenz et al., 2013; Garratt et al., 2014); however, for many crops the variations in these benefits remain unknown. As such, estimates of value extrapolated from a single cultivar may be misleading, particularly in crops with a high cultivar turnover (e.g., oilseed rape – Hudewenz et al., 2013) or where cultivars sell for different prices (e.g., Garratt et al., 2014). Many studies do not account for increases in costs resulting from additional pollination, such as greater picking or input costs (Winfree et al., 2011).

<u>Methods Affected:</u> Yield Analysis (Section 2.2.1.), Dependence Ratios (Section 2.2.2.), Surplus Models (Section 2.4.).

<u>Impacts:</u> Failure to capture the full extent and variation of benefits for a crop can result in under- or over-estimation of benefits, particularly if extrapolated over a range of cultivars (see Garratt et al., 2014). This will in turn affect the estimates of changes in crop production on prices, an important component of welfare analysis – for instance if crop quality decreases more than quantity then overall prices may fall even in cases of lower available supply.

5.2.3 Interactions between pollination services and land management or other ecosystem services

5.2.3.1 How do management practices affect the benefits of pollination services?

Although pollination services can have a strong influence on yields, yields will be strongly driven by local management of the crop, such as input, planting regimes etc. In most economic studies, the benefits of pollination are overestimated because the influence of other anthropogenic inputs (insecticides, fertilizers, etc.) are not accounted for (see Section 2). For instance, Marini et al. (2015) demonstrate that in certain oilseed rape cultivars, yields are enhanced to different extents by the amount of nitrogen applied to the soil but benefits to crop yield from insect pollination seemed to increase with decreased nitrogen levels.

Furthermore, local management can affect the delivery of pollination services. Recent reviews and meta-analyses suggest that the impacts of human land use on pollinators are generally negative (Kennedy et al., 2013). Kremen et al. (2012) concluded that agricultural intensification reduced the diversity and abundance of native bees such that pollination services they provided are below the necessary threshold to produce marketable products. To date there have been very few studies that have looked at the impacts of changing management on the economic benefits of pollination services (but see Blaauw and Isaacs, 2014).

Methods Affected: Dependence Ratios (Section 2.2.2.), Production Functions (Section 2.2.3.), Surplus Models (Section 2.4.).

Impacts: Failure to account for the impacts that management and inputs can have on the scale of benefits to crops (including additional costs) can result in over- or under-estimation of the benefits of pollination services to a crop. This is particularly significant when extrapolated across larger spatial scales that encompass areas with natural variations in productivity (e.g., through soil quality, climate etc.). Furthermore, the capacity to trade-off between pollination and other inputs is an important consideration in surplus modelling, particularly general equilibrium models (which consider how such substitutions could affect benefits) and production function analyses (which consider the benefits of pollination relative to other factors affecting yield) and could limit the accuracy of both approaches.

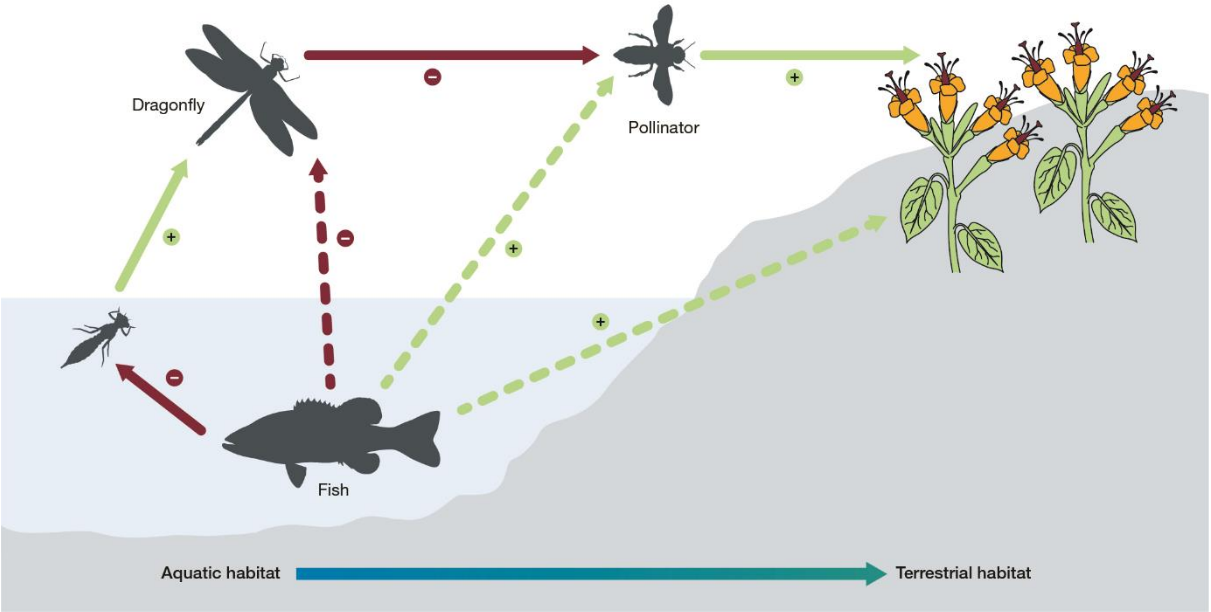
5.2.3.2 How do different ecosystem services affect the benefits of pollination services?

Most research implicitly uses as a simplifying assumption the notion that ecosystem services (in this case pollination) do not have significant and variable relationships with one another (Bennett et al., 2009). Decreasing level and stability of yield in insect-pollinated crops has so far solely been attributed to pollinator declines, without considering how other ecosystem services have changed in tandem (Lundin et al., 2013). Different factors, including pollution, can change these ecological relations; therefore, there is a need to alleviate humans’ impact on nature by a holistic approach that includes and prioritizes the loss of pollinators. To ensure continued ecosystem services, it will be important to maintain not only an abundance of key species but also species interactions and the diverse, healthy ecosystems that sustain them.

Furthermore, despite their apparent importance, interactions among ecosystem services, particularly those involving regulating services have generally been underappreciated; ecological management and monitoring have focused on provisioning or cultural services. While there has been substantial ecological research on some regulating services such as pollination and carbon sequestration, these services' role in ensuring the reliability of other ecosystem services has not been systematically assessed (Bennet et al., 2009). For example, Knight et al. (2005) demonstrate the impact water quality can have on pollinators via trophic cascades. Fish that require good water quality to maintain stable populations in turn predate upon dragonflies, the principal predators of pollinators within the system (Figure 4). Loss of water quality can therefore affect pollination services by reducing the fish population, reducing the predation on dragonflies and indirectly increasing predation on pollinators.

Methods Affected: Yield Analysis (Section 2.2.1.), Dependence Ratios (Section 2.2.2.), Production Functions (Section 2.2.3.), Surplus Models (Section 2.4.).

Impacts: Coordinated management for multiple ecosystem services can have positive synergistic effects, which can outweigh the summed benefits of managing ecosystem services that are spatially or temporally separate (Lundin et al., 2013). Failure to adequately capture these trade-offs will lead to an over-/under-attribute yield gains to pollination services. Research that quantifies the provision of multiple services, the trade-offs and synergies among them and also examines the ecosystem processes that link services will lead to a better understanding of how the relationships among ecosystem services can change over time and space (e.g., Marini et al., 2015). Such understanding may enable manipulation of systems to decrease trade-offs, enhance synergisms, and promote resilience and sustainable use of ecosystem services (Volk, 2013).



35. Figure 4 – Interaction web showing the pathway by which fish facilitate plant reproduction
Solid arrows indicate direct interactions; dashed arrows denote indirect interactions. The sign refers to the expected direction of the direct or indirect effect (modified from Knight et al., 2005).

5.2.3.3 How do pollination services affect the benefits of other ecosystem services?

Although pollination is a service that results from direct interactions between plants and animals, because of its reproductive value to plants it also has an important, indirect role in other vegetation-based services, such as water filtration, erosion control, carbon storage and sequestration (Montoya et al., 2012) and landscape aesthetics (Breeze et al., 2015). The total value of insect pollinators to crop production would be even higher if indirect benefits, such as enhanced soil fertility and soil conservation through the pollination of various nitrogen-fixing legumes and replenishing soil nutrients, were taken into account (Partap et al., 2014).

For example, the total value of insect pollinators to agriculture would be even higher if economic risks of both direct crop sectors and indirect non-crop sectors in the economy were taken into account (Bauer and Wing, 2014). Perhaps the most drastic effects would be in uncultivated areas where a large share of the soil-holding and soil-enriching plants would die out (Bohart, 1952).

<u>Methods Affected:</u> Yield Analysis (Section 2.2.1.), Dependence Ratios (Section 2.2.2.), Production Functions (Section 2.2.3.), Surplus Models (Section 2.4.), Stated Preferences (Section 2.5.)

Impacts: Like farm management practices, failure to account for the interaction between pollination and other ecosystem services can result in under- or over-estimation of the benefits of pollination services, especially for crops that are highly self-incompatible. In order to assess total economic value, it is important to quantify the various non-market benefits of pollination services. In order to do so, the contribution of pollination services to various benefits provided by other, intangible ecosystem services must be quantified to accurately extrapolate the value of pollination to these final services via stated preferences or production functions that capture appropriate feedbacks. Failure to do so will result in some benefits being ignored in valuation and trade-off decisions informed by them.

5.2.4. How do variations in wild pollinator communities affect service delivery?

Only recently have ecologists specifically addressed daily, seasonal, and annual temporal patterns in network structure of plant-pollinator interaction networks (Burkle and Alarcon, 2011). For example, Price et al. (2005) studied pollination by insects and humming birds to a montane herb (*Ipomopsis aggregate*) for 7 years, finding that pollination services are variable within and between years by several orders of magnitude even at the same sites. Whereas studies of short duration may detect covariance of floral variation and pollination success, additional sources of variation across sites and years may weaken, strengthen, or even reverse this effect (Burkle and Alarcon, 2011). Although plant–pollinator systems are highly dynamic, measures of their interaction networks are structurally stable across all time scales studied. This suggests that the mechanisms governing the assembly of pollination networks are likely independent of species composition, thereby preserving ecosystem function, across seasons, years or decades (Burkle and Alarcon, 2011). A better understanding of the links between pollination and population dynamics is needed to know when limits to seed input and seedling establishment affect population size and structure (Wilcock and Neiland, 2002).

Several case studies have noted that wild pollinators may positively enhance the effects of managed honeybees on crop yields through by increasing movement across flowers (Greenleaf and Kremen, 2006) or movement between different individuals of self-incompatible crops (Brittain et al., 2013). These studies demonstrate the economic importance of interspecific interactions for pollination services in some crops (but this may not be widespread; see Garibaldi et al., 2013) and suggest that protecting wild bee populations can help buffer the human food supply from honey bee shortages.

Both abundance and behavioural-mediated mechanisms can enhance the stability of pollination services in some crops (Greenleaf and Kremen, 2006; Brittain et al., 2013). Assemblages that contain a wide range of species with different ecological requirements could maintain pollination services as environmental conditions change over time because i) declines in abundance of some taxa can be offset by increases in others and ii) interspecific interactions can enhance net pollination services. This is the basis of the biological insurance hypothesis with respect to pollination as an ecosystem service (Rader et al., 2012). Lever et al. (2014) describes the capacity of pollinator populations to persist under harsh conditions. However, once a system's threshold is reached, pollinator populations may collapse simultaneously, raising questions about the resilience of pollination networks across different temporal and spatial scales (Petanidou et al., 2008).

Several studies (Javorek et al., 2002; Artz and Nault, 2011; Rader et al., 2012) have compared the pollination service effectiveness of honeybees and various wild pollinators (wild bees, flies), using not only the frequent visitors but also different measures of pollen transfer efficiency (amount of pollen deposited on stigmas per single visit and stigmatic contact). Rader et al. 2012 found that pollinator importance changed little irrespective of the spatial and temporal variations among taxa.

Methods Affected: Production Functions (Section 2.2.3.), Natural Capital quantification (Section 2.6.), Resilience stocks (Section 4.3.3).

Impacts: Understanding the contributions of different pollinators within a community and the effectiveness of their interactions (Greenleaf and Kremen, 2006) is essential to understanding the total economic benefits of a community, identifying areas with possible pollination deficits and planning management accordingly. The management requirements for e.g., ground-nesting bees may be very different to those of hummingbirds, moths etc. This can result in over- or under-estimating the value and resilience of wild pollinator natural capital within the landscape by incorrectly assuming that all pollinators provide equal benefits to a particular crop.

5.2.5. How effective are artificial pollination methods

While numerous technological replacements and supplements for insect pollination services have been developed (Pinillos and Cuevas, 2008) their effectiveness in providing pollination services compared to animals remains unknown for a large number of crops. Different technologies are likely to be differently effective for different crops; for example, hand pollination is effective in Cherimoya (Gonzalez et al., 2006) but not Raspberry (Kempler et al., 2002) and some have been developed in response to specific needs (e.g., vibration wands in tomatoes – Pinillos and Cuevas, 2008).

Methods affected: Replacement Costs (Section 2.3.).

Impacts: Replacement cost studies must assume that methods are equally effective to animal pollination, which may not be the case, over- or under-estimating the total costs involved. Furthermore, effectiveness may affect producers' willingness to uptake the replacement. If this is not known, it will not be possible to accurately estimate how realistic the replacement scenario is.

5.3. Economic knowledge gaps

5.3.1. Limited information regarding non-market or non-monetary food consumption

Studies into the economic benefits of pollination services have thus far exclusively focused on crops produced and traded on monetary markets. In reality, producers in many countries will consume a certain amount of their own produce in lieu of selling it on an open market or will exchange their produce directly for other goods and services. For example, in India though most of the crops and their value are covered, fruit and vegetable production statistics are inadequate. About 40% of the geographical area under agriculture is without the benefit of reliable statistics on crop acreage by crop season (Sengupta, 2007). Similarly, people across the world have access to wild fruits and many will grow a small amount of their own food in gardens or allotments. While it is possible to estimate the economic benefits of this produce by determining the equivalent value of the produce on the market and applying standard valuation methods, there are no large-scale estimates of the amount of produce used in this way.

Methods affected: Dependence Ratio (Section 2.2.2), Surplus Models (Section 2.4.).

Impacts: This knowledge gap limits understanding of the full extent of pollination service benefits to crop production by underestimating the total amount produced. In many developing countries, crops consumed at home or traded in non-monetary exchanges are likely to be a significant part of local consumption. The welfare benefits of non-market crops consumed by producers are likely to be very significant to local producers as the crops are consumed at effectively no cost.

5.3.2. Limited information regarding seasonal trade in produce

Most studies on the value of pollination services have only considered inter-annual variations in crop production. In reality, production and, by extension prices will fluctuate within the year as well (intra annual variation) for some crops. Although modern refrigeration can extend a crops storage life, making it available longer throughout the year (Klatt et al., 2014), spikes in availability are likely to occur for many crops. This will affect both short-term prices and total international trade within the year, with imports increasing to meet demands where supplies are lower and subsequently lowering the overall price (Kevan and Phillips, 2001). Although some seasonal price data is available (e.g., UK – Defra, 2014) the extent of seasonal variation in international production and trade of insect pollinated crops remains largely unknown.

Methods affected: Surplus Models.

Impacts: Lacking seasonal data, the effects of international trade on national prices over time are impossible to estimate. As such, estimates of the impact of pollination services on consumer or producer welfare remain incomplete. This is particularly significant when estimating the impacts on secondary consumers as supplies may be strongly linked to certain regions at particular times of the year, increasing the negative consequences of service losses in those regions.

5.3.3. Limited information regarding production and consumption on the secondary market

Presently, all estimates of the market value of pollination services have used data on the sale prices paid to producers. As such, any estimates of value derived from them only reflect the welfare benefits to primary consumers only. In many countries, these buyers will be wholesalers (e.g., supermarkets) who will in turn sell the produce at a higher price elsewhere; for instance, in the UK, sales at farm gate only reflect 42% of the final sale price (Defra, 2014). Thus, the welfare of these end consumers has not yet been assessed and may potentially be additive to the value to initial buyers, should price shocks be passed further down the supply chain. Furthermore, the preferences of end consumers will drive primary consumption and production of particular crops in order to meet demands. As long-term sales and prices set by these suppliers are considered commercially sensitive, it is very difficult for research to establish the structure of these secondary markets.

Methods affected: Surplus Models (Section 2.4.).

Impacts: The lack of sufficient information on the quantity and price of produce on the secondary market limits the capacity of existing methods to assess the impacts on end consumers, under-estimating the total benefits of pollination services by neglecting a large proportion of beneficiaries. Furthermore, information on consumer preferences is important to establishing crop substitution elasticities, limiting the capacity of research to estimate how prices respond to changes in the supply of a particular crop and the resultant impacts on producer and consumer welfare.

Section 6 – How economic gains and losses in pollination can be used to inform decision-making?

6.1. Overview

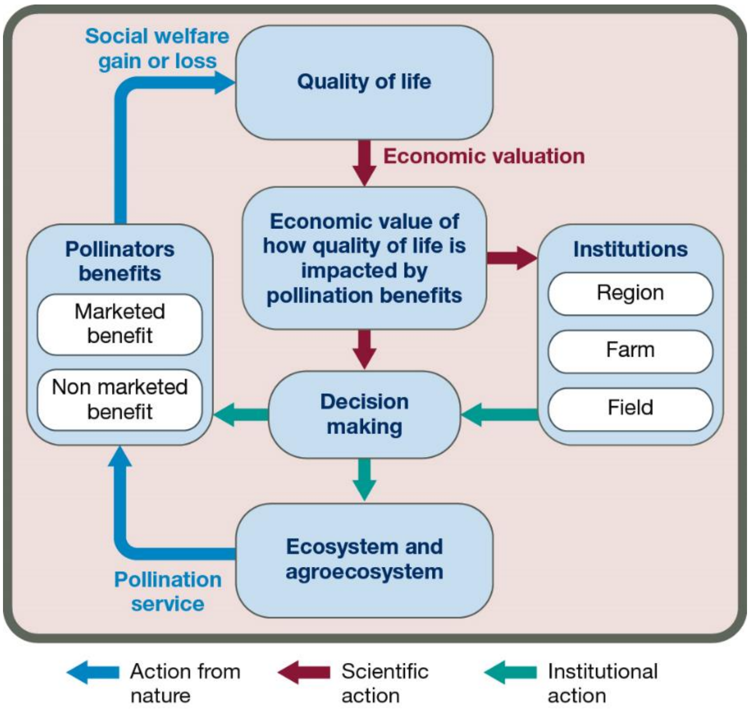
Institutions, governance systems and other indirect drivers are the ways in which people and societies organize themselves and their interactions with nature at different scales (Díaz et al., 2015). The decision process of protecting or not protecting pollinators is driven by the organization of the society. These benefits can be private (increased farmer profit due to pollination), or public as the amenities created by pollination on a landscape. **Error! Reference source not found.** illustrates how economic valuation (red arrows) can be used directly or indirectly for decision-making (green arrows) at different scales within the framework of IPBES. Economic valuation can be used by private and public institutions to estimate the importance of pollination services. By measuring the economic impact of changes on private or public benefits', valuation can feed directly into the decision making process.

6.2. Tools and methods for using economic valuation in decision-making

Economic valuation of pollination services can be used at scales ranging from individual farmers and cooperatives to national governments. Important tools and methods to inform decision-making that rely on economic valuation are, mainly, cost-benefit (and cost-effectiveness) analysis (Chapter 4, Section 1.1.4 and Chapter 6, Section 6.5.1.5), environmental accounting (Chapter 6, Section 5.8) and modelling pollination services (Chapter 6, Section 5.10). Some other tools integrate or incorporate economic valuation as vulnerability assessment (Chapter 6, Section 5.7), decision support tools (Chapter 6, Section 5.12), and Multi-Criteria Analysis (Chapter 6). Multi-Criteria Analyses (MCA) are a family of methods which combine multiple metrics into a series of criteria to simultaneously consider a range of impacts arising from activities and decisions (Sijtsma et al., 2013). MCA often include economic considerations (e.g., the rate of employment and profit) alongside environmental (e.g., habitat and air quality), political (e.g., political stability and participation) and socio-cultural (e.g., education and cultural identity) aspects (Estevez et al., 2013; Scriedu et al., 2014). Although MCA have been applied to management scenarios concerning the management of ecosystem services, including those important to food production (e.g., Fornata et al. 2014, Volchko et al., 2014), to date no study has directly assessed pollination services within this framework. MCA are particularly advantageous as they are capable of considering the full suite of values that the affected stakeholders possess, rather than solely focusing on an economics worldview, which may not always be appropriate (Scriedu et al., 2014). Both monetary and non-monetary assessments of the benefits of pollination services can be incorporated into MCA depending on the criteria identified by the affected stakeholders. For instance, if agricultural productivity were identified as an important economic criterion for stakeholders, then both the monetary value of pollination services to crops and the available stocks of pollinator assets to ensure current and future production would be ranked highly. However, in regions where agriculture is primarily subsistence based, it may be more appropriate to consider the non-monetary benefits of pollination to capital (Section 2.6, 2.8.).

In Chapter 6, Section 6.5 (Table 6.5.2), the experience, strengths and weaknesses of these tools and methods for informing decisions about pollinators and pollination are reviewed, alongside other tools and methods less reliant on valuation.

Economic valuation of pollination is a crucial element in designing payment for ecosystem services schemes (FAO, 2007; Chapter 6, Section 4.3.3), because the value of the service provided could constitute one basis for justifying the payment amounts. Another basis could be the opportunity cost to the producer.



36. Figure 5 – Schematic representation of how economic valuation is used by institutions and for decision making at different scales
as embedded in the IPBES framework (adapted from Díaz et al. 2015).

6.3. Use of economic valuation of pollination at different stakeholder levels

Once the use and non-use values for both, private and/or public benefits of pollination services (including economic consequences of pollinator decline) are known, appropriate responses can be developed at multiple levels. In agriculture, the main levels of governance are typically: farmer, producer/cooperative, industry and government (Daily et al., 2009; Kleijn et al., 2015). In Chapter 6, Table 5.3 describes the utility of different tools and methods for decision-making on pollinators at these different levels.

6.3.1. Use of valuation at farmer level

If farmers know the potential economic consequences of pollinator decline in their private benefits, they can choose alternative crops or varieties that do not result in either loss of income to them as private actors or to society as a whole. For example, hybrid varieties of oilseed crop have both higher values per unit produce and requirements for insect pollinators than the open-pollinated varieties. If there are declining trends on the availability of managed honeybee colonies in the area, then the farmer will be able to estimate loss of production from hybrid crop versus open-pollinated crop and make appropriate decision at farm level (Hudewenz et al., 2013; Marini et al., 2015). Economic valuation will be helpful in understanding or estimating tangible losses from any change in pollination service arising from changes in populations of pollinators and hence farmers can make decisions to grow particular types of varieties to cope with that situation. Alternatively, knowing the profitability losses of pollinators could be used to invest in measures to mitigate loss (such as flower strips) (Wratten et al., 2012; Garibaldi et al., 2014).

6.3.2. Use of valuation at producer level

If a group of farmers is involved in, for example, seed production, then they can measure the profit gain or loss due to pollinators change (using e.g., production function models; Section 2.2.3.) to guide their decision-making for appropriate production and marketing strategies. If there is a trend in the profit changes from linked pollinator gains and losses in the area, seed producers can make decisions to adjust their operations accordingly and establish a collaborative grower response. They can adopt certain strategies to bring additional managed pollinators or to change the type of crops that depends less on pollinators.

As described by Fisher et al. (2009), pollination services are provided omni-directionally and their benefits affect much of the surrounding landscape. When this service is offer by wild insects providing by a natural habitat, economic valuation can be used to incentivise a group of farmers who benefit from this service because their fields are in the surrounding landscape to maintain it.

6.3.3. Use of valuation at industry level

Industry’ scope is local, national and global. Industry that deals in sales and marketing of seed, oilseed crops, horticultural crops and other food products dependent on pollination can develop their strategies to respond to any change in pollinators’ populations. Industry can forecast the production figures, financial profits or losses and responses to shareholders based on profit valuation studies. For example, in cases were an industry is highly dependent on insect pollination, being able to illustrate the projected profit loss of a pollinator shortage (Allsopp et al., 2008) can allow this industry to more effectively lobby with government for pollinator friendly regulations or concessions (de Lange et al., 2013).

6.3.4. Use of valuation at government level

While the farmer, producer and industry levels are concerned with private values of pollination, governments (local, national or international) focuses on the effects on social welfare arising from pollinator gains and losses. Social welfare encompasses the firm profits but also the consumer welfare. Governments can use the economic valuation as a policy tool to respond to the changing needs of constituents mostly farmers in many parts of the world. Appropriate agricultural and food policies can be developed by using the information on valuation of pollination services (TEEB, 2010). For example, if there are significant changes in the population of pollinators, then governments can guide, through appropriate regulation or incentive, changes in the cropping patterns in the agricultural area. They can promote other crops with relevant inputs and market support to overcome any predictable losses due to the crops that are more dependent on pollinators (Garibaldi et al., 2014). Alternatively, government can support landowners more directly to maintain pollinator habitat through subsidies and/or regulations in cases where the pollinator-dependent crop is too valuable to society in terms of produce, export or employment provision, to replace. Pollinator maps showing varying level of abundance, habitat and key species can also be developed. These maps can be used along with economic valuation by decision makers (governments) for resource allocation to support agriculture. For instance, it could be helpful to know where the pollination potential is high, and simultaneously the crop production dependence on pollinators is high. It is also helpful for governments to have some monetary values to support some decisions. Government can also evaluate the non-marketed benefits of the pollination and use this economic valuation to estimate the interest or not in managing pollinator populations. The difficulties of such a valuation is that private and public interest are and measure the amount of the subsidy or taxes sufficiently high to incentivise landowners to change their practices.

6.4. Step-wise guide for using economic valuation for decision-making

Decision-making aims to protect or maintain the private and public benefit due to pollination service, and this for both wild and managed pollinators. Regardless of the scale used for economic valuation, there are a set of sequential steps to be taken to enable decision-making.

1. Determine the level of pollinator dependence of the plant as crop, crops grown or wild plant. This can be achieved with field studies (e.g. Yield analysis or production function models, Section 2.2.1 and 2.2.3) or through published resources such as Klein et al. (2007).
2. Determine the scale of production affected - the number of producers, the area of crops or the wild plants' landscape, the distribution within the region, etc. This is typically based on existing

national statistics. It is also necessary to determine the beneficiaries of pollination services at this stage in order to identify appropriate temporal or spatial scales of benefits measurement. If the benefit of the pollination service is marketed as a crop, the beneficiaries are typically farmers and consumers (including secondary consumers). Similarly, the contribution of pollination to overall agricultural production and the rural sector can be calculated. Ideally, valuation should be accompanied with consultation of these stakeholders to accurately incorporate their wants, needs and constraints and to identify any mis-matches between their objectives (Ratamaki et al., 2015).

3. The proportion contribution of wild versus managed pollination needs to be determined. This can be achieved through observational field studies (e.g. Winfree et al., 2011), cage studies of individual pollinator efficiency or through pollination production function models (Section 2.2.3).
4. The current availability of wild or managed pollinators now needs to be determined, ideally to act as a baseline. This can be achieved though current information on the numbers of managed pollinators within the country or using modelling approaches such as InVEST to predict wild pollinator populations (Lonsdorf et al., 2009). As stocks of managed pollinators can be used to offset any pollinator service shortage (Breeze et al., 2014), they should ideally be considered even when wild pollinators dominate the service providing community. In contrast, if managed pollination is not available and there is no wild pollinator replacement, substitution is limited.
5. Now, the economic valuation of pollination services should be undertaken to establish baseline estimates and monitor or project the impacts of changes. If the output of the pollination service is an amenity, the beneficiaries of this amenity should be distinguished. The valuation method used will depend on who the stakeholders are and the case over which the assessment is to occur – e.g. local farmers will be informed sufficiently by a Yield analysis (Section 2.2.1) or Production Function model (Section 2.2.3.) while larger scale analyses should consider surplus valuation (Section 2.4.). Some methods (e.g. stated preferences) are suitable at al scales.
6. Once benefits have been valued (or quantified) introduce economic valuation in a tool for decision making (e.g., CBA or MCA) to determine the impacts of actions. This step is necessary to decide whether to protect or maintain pollination service relatively to the constraints (in terms of time or budget or social, economic and environmental priorities, etc.) of the decision-makers.
7. The last step is the action of protecting or maintaining the pollinators using the economic instruments (PES, incentives/taxes, subsidies, etc. See Chapter 6, Section 5).

There are very substantial uncertainties at each of these steps (see Chapter 6, Section 7), particularly regarding the availability of wild and managed pollinators in a particular place, and the relative contribution of wild and managed pollinators to a particular crop, which are clearly linked.

The next section discusses case studies in details from local to global scale. Some of these cases highlight how economic valuation can be used for decision-making (Ricketts et al., 2004; Cook et al., 2007; Allsopp et al., 2008).

Section 7 – Case studies: from local to global

Since the late 1960s, there has been substantial increase in interest for the economic value of pollinators through their pollination service (Helliwell, 1969; Costanza et al., 1997). The topic received particular interest in the US, several European countries, Australia and New Zealand, where estimates of the value of pollination have been made for a wide range of different crops. A range of studies have shown that pollination makes a very significant contribution to the agricultural production of a broad range of crops, in particular fruits, vegetables, fibre crops and nuts. Estimates of the annual economic value of pollination have been made for the global scale.

Less information is available from many parts of the developing countries, much of which focuses on pollination services to coffee, one of the world's highest priced agricultural crops, where pollination contributes significantly to economic outputs (Klein et al., 2003). This section reviews some of the most significant studies into the economic impacts of pollination services from across the world at various spatial scales.

As currencies vary between studies and the strength of currency can vary throughout time (Section 3), all monetary figures presented in this section have been converted to 2015 US$ using average annual spot exchange rates from the Bank of England (Bank of England, 2015). These dollar estimates were inflated to 2015 US$ using Consumer Price Index (CPI) data from the United Stated Federal Government's Bureau of Labour and Statistics (BLS, 2015a). Inflation was based on the CPI for July of the year the estimate was related to compare with the CPI in July 2015 (BLS, 2015b). If this year was not stated, then the year before the study was published was used instead. If estimates are based on data average across several years, the average exchange and inflation rates across all the relevant years were used. These inflations only represent a change in the value of currency and do not capture any changes such as the relative input prices, price controls or subsidies.

For example – Gallai et al. (2009a) estimated global crop pollination benefits in 2005, using a dependence ratio method at €153bn. This is divided by the exchange rate (0.8053€ per US$) and then multiplied by the rate of inflation (the proportionate change in the consumer price index between 2005 and 2015: 1.221), giving a value of $232bn. Similarly, Lautenbach et al. (2012), widely cited in this report, estimate the economic benefits of global pollination services at $212-$520bn in international dollars (a monetary unit that adjusts all prices based on power purchasing parity) in 2009. As US dollars are the basis of the international dollar, no currency conversion is required so the value is simply inflated by multiplying it by the inflation rate (1.108), resulting in a value of $235bn-$577bn in 2015 US dollars.

7.1. Local and regional scale

At the smallest scales (farms, communities etc.), changes in pollination services are unlikely to affect consumer wellbeing as the loss of production is likely to have little to no impact on crop prices (Section 2.5). As such, almost all studies examining the economic impacts of pollination service losses at these scales have used the Yield Analysis method (Section 2.2.1) to examine the potential market output loss that would occur following a complete loss of pollinators. Kasina et al. (2009) used this method to estimate the economic returns from bee pollination in smallholder farming systems in the Kakamega region of western Kenya in 2005. The net benefit (after accounting for costs) that Kakamega farmers received from bee pollination of eight focal crops was estimated at $3.9M, almost 40% of the annual market value of these crops in 2005. In Brazil, DeMarco and Coelho (2004) assessed the economic benefits of pollination to coffee grown close to native forests in 2003. Pollination resulted in a 14.6% average yield increase in areas close to native vegetation. This increase refers to 25.4 more coffee sacks per ha for the producer, equivalent to $2,414/ha/year (2015 US$).

Coffee has also been the focal crop in a number of studies examining the value of pollinator natural capital from the surrounding landscape. Ricketts et al. (2004) analysed pollination in 480 ha of coffee fields that are within 1km of two forest patches in Costa Rica compared with a hand-pollinated control at each site to represent maximum pollination. Their findings indicate that pollination increases coffee yields by ~21%, with benefits declining towards the centre of the plantation. Considering the differences in coffee yields, coffee prices and the costs of production, they estimate the surrounding forest on the plantation generates annual benefits $82,901 (2015 US$), representing ~7% of the annual income from the plantation.

Olschewski et al. (2006) used a regression based model alongside data from Klein et al. (2003) and locally collected yield data to estimate the marginal benefits of pollination services per hectare of forest patches at different distances to coffee plantations in Indonesia and Ecuador. They found that the marginal benefits of forest patches to coffee depends on the quantity of forest converted, estimating that pollination services increase producer net income by $0-$63/ha (Ecuador, 2015 US$) and $0-$66/ha (Indonesia, 2015 US$) depending on the distance between the habitat and plantation.

A more advanced study was undertaken by Ricketts and Lonsdorf (2013) who adapt the InVEST model of Lonsdorf et al. (2009) using the information from Ricketts et al. (2004). The findings indicate that each hectare of forest fragments provided between $0-$936/year (2015 US$) of pollination services depending on their location relative to the coffee and other forest patches. The highest marginal values are found in forests that provide high-quality resources for which there are few substitutes. The average marginal value of forest parcel declined exponentially with forest cover within a 500 m radius.

More recently, Winfree et al. (2011) estimated the benefits and economic value of pollination services by native bees and honeybees to watermelon pollination in New Jersey and Pennsylvania, USA using both replacement costs and yield analysis. Unlike many other studies, this analysis explicitly considers how producer costs may change because of changing yield. Surplus modelling was not undertaken as the two states contribute less than 2% to US national watermelon supply. Their findings estimate the benefits of pollination services to producer net margins at $4.0M (2015 US$); less than half the benefits estimated if producer costs were not accounted for ($8.5M – 2015 US$). The costs of replacing native pollinators and existing honeybees' colonies with new honeybees' colonies provided even smaller estimates of $0.23M (2015 US$) and $0.2M (2015 US$) respectively. This study highlights the differences in the scale of estimates between methods and the potential over-estimation of benefits if changing producer costs are not considered. However, the study does not specify what variable costs it assumes will change with changing yields and therefore may overestimate the cost change if inputs that are applied before harvest (e.g., fertilizer) are included in this calculation.

Local economic benefits can also be considered from the perspective of indigenous and local knowledge (see Chapter 5 for more details). In several cultural contexts, before the introduction of money or in parallel, indigenous people use honey, and sometimes beehives, as an exchange value (non-monetary). Among forest hunter-gatherers, honey is shared within the group as it is collected and then taken back to the village for further distribution. According to Ichikawa (1981), honey is the medium by which the Mbuti pygmies regulate their social relations. Although honey belongs to the individual who finds it out, the owner alone does not consume it. It is distributed to other members of the camp and it is frequent that the owner of a nest asks the other men to collect his honey. The practice of honey distribution and labour exchange compensates the separatism among the camp members, which is liable to occur during honey season (Ichikawa, 1981). Terashima (1998) stated that like sharing economic reciprocation is important to maintain a strong and durable relationship in the group, but also with neighbours: in exchange for honey, the Efe pygmies obtain from their neighbours, named Lese, clothes and agricultural food like plantain and manioc, which constitute a significant portion of their diet.

Césard (2007) recorded that the Punan Tubu in Indonesian Borneo have exchanged honey and other forest resources with their farming neighbours and with traders for goods that were used in marriage payments. Merchants travelled upstream to trade directly with collectors the products in demand then, using various measurement standards to establish their exchange value. In the Danau Sentarum region, wax was also traded (Césard and Heri, 2015). Hunters, beekeepers, now small-scale herders and agriculturalists, the Ogiek people in Kenya have long traded honey with their Maasai, Kikuyu and Kipsigis neighbours in exchanged for livestock, dogs or grains. Honey and honey beer are also consumed in ceremonies. Muchemi et al. (2011) reported that even if money is now the main medium of exchange, honey is still used in matrimonial payments. During marriage negotiations and as part of the bride price, the boy's relatives give to the girl's relatives several bags of honey and calabashes of honey brew. More than ten large bags (about fifteen litres each) can be demanded and beehives are also exchanged between families in the marriage process (Samorai Lengoisa, 2015).

7.2. National scale

Stanley et al. (2013) assessed the benefits of pollination services to oilseed rape at the national scale in Ireland by extrapolating the results of a yield analysis (Section 2.1.) conducted in ten fields in 2009-2011 across the country. All fields were at least 1 km apart, and only one field was selected per farm to avoid potential bias due to specific management practices on one particular farm. Exclusion of pollinators resulted in a 27% decrease in the number of seeds produced, and a 30% decrease in seed weight per pod in winter crops, with comparable values from a spring oilseed rape field. Extrapolating the results to a national scale, the economic value of insect pollination to winter oilseed rape in Ireland was estimated at $3.9M (2015 US$) per annum, while the contribution to spring oilseed rape was $1.9M (2015 US$), resulting in an overall value of $5.8 M (2015 US$) per annum.

Although upscaling yield analysis has been used for specific crops, the national scale benefits of pollination services to multiple crops are usually estimated using a dependence ratio methodology. Several interlinked studies have used this method to estimate the benefits of pollinators to agriculture in the USA; beginning with Robinson et al. (1989) which estimated that honeybee pollination services added to agricultural production in 1986, estimating a total market price of $20.3bn (2015 US$). Subsequent studies have gradually updated and refined this value; Morse and Calderone (2000) updated the information for 1996-1998 ($21.8bn 2015 US$). Losey and Vaughn (2006) used the same dependence ratios to estimate the value of wild pollinators in 2003 ($4bn 2015 US$), alongside other ecosystem services totalling ~$74bn (in 2015 US$ - $0.5bn for dung burial, $4bn for pollination, $5.8bn for pest control of native herbivores, and $64.8bn for recreation). Most recently, Calderone (2012) estimated the annual benefits of pollination services per hectare of US crop agriculture from 1997-2009, indicating that this value had steadily risen from $4,666.38/ha in 1997 to a peak of $7,399/ha in 2008 (2015 US$). The total value of pollination services in the USA across this time period follows similar but less substantial trends, rising from $15.6bn in 1996 to $17.07bn in 2009 (2015 US$) even as the area of insect pollinated crops gradually decline, indicating that price rises and a growing prevalence of higher value crops drive the average per hectare rise.

Although increasingly comprehensive, these studies only estimate the market benefits rather than societal value. Southwick and Southwick (1992) addressed this shortcoming by analysing the consumer surplus (Section 2.4) related to crop pollination by honeybees in the US in 1987. Based on ~20 years of price and consumption data, they estimate the demand curve for 50 different crops.

Furthermore, the study includes a number of weights to reflect the capacity of wild pollinators to substitute for lost honeybee pollination services. The estimated value of honeybee pollination services to 17 crops was estimated at between $3.4bn (partial substitution by wild pollinators) to $11.6bn (2015 US$ - no substitution). Like many consumer surplus studies, this study unrealistically assumed that producers could freely switch between wind pollinated and animal pollinated crops without costs and therefore suffer no welfare loss from pollinator declines (see Section 2.4. for a full discussion). Furthermore, this study, like Morse and Calderone, Losey and Vaughn (2006) and Calderone (2012), primarily uses the dependence ratios of Robinson et al. (1989) which are mainly drawn from expert opinion rather than field study.

The annual migration of monarch butterflies (*Danaus plexippus*) has high cultural value and recent surveys indicate monarch populations are declining. Understanding how much, and where, humans place value on migratory species can facilitate market-based conservation approaches. Diffendorfer et al. (2014) performed a contingent valuation study of monarchs to understand the potential for such approaches to fund monarch conservation. The survey asked U.S. respondents about the money they would spend, or have spent, growing monarch-friendly plants, and the amount they would donate to monarch conservation organizations. The study found nearly three-quarters of those surveyed support conservation efforts for the species. Combining planting payments and donations, the survey indicated U.S. households valued the existence of monarchs (as a total one-time payment) at $5bn–$6.9bn, levels similar to many endangered vertebrate species. This value is likely an over-estimate as it is based on the assumption that all households would be willing to pay an average of $32-$42 (2015 US$). Nonetheless, it highlights that the financial contribution of even a small percentage of households could generate new funding and resources for monarch conservation through market-based approaches.

Beyond the USA, Gordon and Davis (2003) examined the consumers and producers surplus value of honeybee pollination in relation to 35 crops grown in Australian agriculture using a partial equilibrium model (Section 2.4.). This study calculates demand curves for both domestic and imported production of each crop in order to capture consumer's ability to switch between domestic and imported product. The import elasticity is usually larger the domestic demand elasticity as, on the international market, the Australian products are, in many cases, relatively easily be replaced by products from other countries. The producers' surplus is calculated for three assumptions regarding the loss of income, following a decline in the pollination service that farmer will incur before they switch to another crop; 0%, 30% or 100%. If farmers, following a loss of the pollination service, immediately switch to a new crop that does not depend on pollination, the producers' surplus is zero (equivalent to Southwick and Southwick, 1992). The results estimate the value of pollination services to Australian consumers at $720M (2015 US$), while the producers' surplus varied depending on when producers switched crops from $0 (producers immediately switch to other crops) to $762M (producers switch to other crops at 100% income loss) (2015 US$).

An and Chen (2011) found that the stock of honeybee colonies in China had increased by 161% between 1961 and 2009, while the area of fruit and vegetable cultivation had increased by 472%, and their production had increased by 833%. The total economic value of insect pollination of Chinese fruits and vegetables amounted to $57bn (2015 US$) in 2008, which represented 25.5% of the total production value of the 44 crops produced in China. Similarly, Liu et al. (2011), using a dependence ratio method, assessed the economic benefits of honeybee pollination services to 36 crops during 2006-2008. In total 60-87.95 million colonies were required to supply Chinese pollination demands in 2008.The average economic benefits of honeybee pollination between 2006-2008, was estimated at $56.1bn (2015 US$), equivalent to 76 times the value of apicultural production, 12.3% of the gross output value of agriculture in China. These results indicate that Chinese agriculture benefits substantially from pollination, particularly from managed honeybees with the greatest demand from vegetables, fruits and cotton.

One of the principle challenges in dependence ratio studies is the potential for inaccurate measurements of benefits to bias dependence ratios. Garratt et al. (2014) estimate, based on a yield analysis extrapolated up to a national scale, that insect pollination increases the net income of producers of two major apple cultivars (Cox and Gala) in the UK by of $62.1M (2015 US$). This study found that insect pollination affects the quality and harvesting costs of apples as well as the number of fruits set. These effects are variety-specific however, with greater effects on yield and quality in Gala ($25,020/ha) than Cox ($20,119/ha) (2015 US$). Accounting for the differences between cultivars and the effects on costs and quality, the estimated national scale benefits were over $10.5M (2015 US$) greater than estimates considering the effects on fruit set alone. Furthermore, the study examined the gap between actual and potential yields, identifying a production gap in Gala worth up to $9.6M (2015 US$) at market prices. This case study highlights the importance of accurate, cultivar specific estimates of pollination service benefits on all facets of output (quality, quantity and costs), particularly at larger scales.

7.3. Global scale

Since the 1990s, there have been several attempts to analyse the value of the pollination service at the global scale. Costanza et al. (1997) provide an early estimate of $177.9bn/year (2015 US$) for pollination services, however this value is based on the assumption that 100% of insect pollinated crop yields would be lost without pollination services (see Section 2.1).

More recently, Gallai et al. (2009a) used a dependence ratio method to estimate the contribution of pollinators to the production of 100 crops used directly for human food worldwide as listed by FAO in 2005. The total market price of this additional production from pollination was estimated at $232bn (2015 US$) worldwide, representing 9.5% of the value of the world crop production in 2005. The market price of a ton of the crop categories that do not depend on insect pollination averaged $174/tonne (2015 US$) while that of those that are pollinator-dependent averaged $876/tonne (2015 US$). The study also estimated the economic value of this pollination service loss at $176.2bn-$302.9bn (2015 US$) (based upon a crop price elasticity of −1.5 to −0.8, respectively) in lost consumer surplus using a partial equilibrium model. This difference illustrates that standard dependence ratio models are unlikely to be effective proxies for the true value of pollination services. However, like most consumer surplus studies applied to pollination, these findings are based on the unrealistic assumption that the producers will be able to freely switch between insect pollinators and non-pollinated crops (see Section 2.4.). Gallai et al. (2009a) also identified the economic vulnerability of different regions to pollination service losses by estimating the proportion of the regions total output of crop agricultural that would be lost without pollination services. This analysis identifies Middle East Asia, Central Asia and East Asia as the regions most vulnerable to pollination service losses, with pollination responsible for 15%, 14% and 12% of output respectively.

Lautenbach et al. (2012), used dependence ratio method to develop maps of global pollination service benefits on 5' by 5' latitude-longitude grid based on cropping patterns in the year 2000. Unlike other dependence ratio studies, the price of production estimated is weighted by the Power Purchasing Parity of each country, adjusting the market prices depending on the relative purchasing power (the amount that can be bought, reflecting the general costs of living in that country) from one US dollar in each country (see Section 3). As such, benefits are adjusted upwards in countries where the cost of living is low and downwards in countries with a high cost of living, making the estimates more comparable between countries. Globally the contribution of pollination to market output, estimated at an aggregate $235bn-$577bn (2015 US$), shows an increasing trend from 1993 to 2009. Spatially, these benefits are focused on a small number of countries: particularly China, India, the USA, Brazil, Japan and Turkey. Comparing the proportion of agricultural GDP that depends on pollination for 1993 vs. 2009: countries like Azerbaijan (3% vs. 13.8%), Russia (2% vs. 6.6%) or Armenia (1.2% vs. 7.6%) have increased their pollination dependency, while China (20% vs. 15.3%). Brazil (15.9% vs. 10%), India (9.4% vs. 4.5%), have decreased their vulnerability. Others such as Canada (7.7% vs. 7.6% in 2008) have remained stable. Pollination benefits show a strong spatial pattern at the sub national scale. For the USA, highest values are observed in parts of California and further north along the West Coast. The highest pollination benefits per hectare arable land in Asia can be found in east China, Japan and South Korea. In Europe, large parts of Italy as well as Greece are exceptional.

The spatial distribution of pollination service benefits also depends on crop species. Soybean is an example of a widely grown, pollination-profiting crop with relative high impact on pollination benefits (values up to $543/ha -2015 US$). Pollination benefits through cotton show a similar widely spread pattern that is generally shifted towards the Equator. The highest benefits (up to $1,662/ha – 2015 US$) can be identified on regional scale in the Chinese provinces Jiangsu, Hubei and Shannxi. Apples and pears show strong overlapping patterns of pollination benefits (Lautenbach et al., 2012).

Although an estimate of economic value, the partial equilibrium modelling employed by Gallai et al. (2009a) is limited by its inability to account for producer input substitution and only considers the producers and consumers of a single market rather than a broader, multi-market perspective. Bauer and Wing (2014), address this by comparing consumer and producer surplus estimates resulting from global pollinator losses using both a partial equilibrium model and a general equilibrium model (Section 2.4) that considers losses on other markets besides crop production e.g., agricultural inputs. These markets will be affected by widespread changes to farming practices, affecting the consumers and producers within the market. Their findings indicate that the partial equilibrium model tends to overestimate the value of services to crop markets, ($259.8bn-$351bn - 2015 US$) compared to in the general equilibrium model ($160bn-$191bn – 2015 US$) due to the latter's capacity to account for producers changing strategies to adapt to pollinator losses. However, because it focuses only on a single market the partial equilibrium model underestimates total benefit ($367.9bn-$689.3bn - 2015 US$). At a regional level, the findings indicate that a loss of local pollination services in South America would have the most negative impacts on local crop markets ($6.4bn - 2015 US$) while Eastern Asia would suffer the largest losses to other markets ($115.4bn – 2015 US$) and North America the largest total losses ($90.5bn – 2015 US$). In some regions, the loss of pollinators would increase total crop market value, particularly in East Asia ($26.3bn – 2015 US$) and crop markets in all regions benefit from the loss of services in any other region, with the loss of services in North America increasing crop pollination value in other regions by $15.8bn (2015 US$).

7.4. Synthesis of case studies

7.4.1. Comparing estimates

The studies highlighted above are part of a larger body of literature that has evolved continuously over the last 20 years. However, estimates of the economic benefits of pollinators can vary strongly between countries, regions and crops. Furthermore, price inflation and the resultant changes in the buying power of currency make comparisons between years difficult. To illustrate the impact of these variations, Table 9 collects available studies from a wide range of sources and expresses them in 2015 US$.

Scale issues can create substantial difficulties in comparing estimates of the economic benefits of crop pollination. Studies covering larger areas and crops with a higher market price inherently produce higher estimates than smaller scale studies on crops with a lower market price. Comparison of estimates can be further facilitated by considering values on a per hectare scale by dividing aggregates by the number of ha for crop production considered in the study of concern (Table 10). When considering the six studies at the global scale, the average benefits of pollination services per ha (in 2015 US$) is between $34/ha (2015 US$ - Costanza et al., 1997) and $1,891/ha (2015 US$ - Bauer and Wing, 2014, using a general equilibrium model – Section 2.5.). However, these estimates are hard to accurately compare as they are in reality expressing different things – from the market price of crops (Costanza et al., 1997) to the welfare value of pollination services (Bauer and Wing, 2014). Furthermore, the per hectare values from surplus valuation studies only represent an average of the welfare loss resulting from the complete loss of pollination services and will shift if anything less than the total area of pollinated crop experiences pollinator losses. Of the three global scale dependence, ratio studies two produce relatively similar estimates (Gallai et al., 2009a; Lautenbach et al., 2012). However, Gallai et al. (2009a) only presents a single estimate of value, based on the median dependence ratios in Klein et al. (2007). Furthermore, it does not weight estimates in different regions by the purchasing power parity of the region. As such, although the figures appear very similar, they are actually strongly divergent. Using the same median dependence ratio values as Gallai et al. (2009a), Lautenbach et al. (2012) estimates total global benefits of $400bn (2015 US$), an increase largely due to the weighting effect of purchasing power parity increasing benefits in regions where the cost of living is low (as 1$ is worth more). This average is similar to the estimate by Pimentel et al. (1997) however, this study bases its estimates on an upscaling of the estimates from Robinson et al. (1989), assuming that the USA accounts for approximately 20% of the global benefits of pollination services.

Table 10 also illustrates that estimated benefits differ strongly between crops (Table 10) due to differences in the prices of the crops. For example, in the UK the benefits per ha of raspberries ($7,641/ha 2015 US$; Lye et al., 2011) are lower than the one of apples ($25,210/ha 2015 US$; Garratt et al., 2014). Secondly, studies considering multiple crops return smaller estimates than those considering only a single crop (e.g., the pollinator-dependent market output to all 18 UK crops collectively is estimated at $1,321/ha 2015 US$ – Vanbergen et al., 2014). To facilitate further discussion, Table 11 compiles all estimates of benefits on a per-hectare scale for apple (*Malus domestica*), a widely studied and grown fruit crop with high market value.

Table 11 illustrates that estimates still differ strongly between countries and regions for the same crop e.g., the benefits of pollination service to apples in China ($10,399/ha - 2015 US$) are lower than in the USA (maximum $17,365/ha 2015US$ - Calderone, 2012; Table 11). There are also notable differences between benefits estimated with different valuation methods for the same crop (Table 11) – with replacement costs producing substantially smaller estimates ($791-$1,634 2015 US$, Allsopp et al., 2008) than most dependence ratio studies ($1,566-$21,744 2015 US$; Zych and Jakubiec, 2006; Calderone, 2012). Even with these controls however, it is difficult to compare the different methods as, although each is expressed in monetary units, all methods measure fundamentally different benefits (see Section 2). However, at both aggregate and per hectare scales, it is apparent that the choice of method can influence the magnitude of impacts that decisions are based on, highlighting the need for transparent, clear and comprehensive assessments of economic benefits in the decision process.

7.4.2. Constraints and limits of current economic valuations

Many studies give an economic valuation of pollinators and pollination service and demonstrate the societal impacts a change in pollinators could potentially have. However, most of these valuation studies focus upon the contribution of pollinators to agricultural production without directly linking it with farmer decision-making. While a great number of studies have illustrated the impacts of animal pollination services on the agricultural sector, studies examining the impacts of pollinator management on producer profits (e.g., Ricketts et al., 2004) and marginal producer welfare (e.g., Kasina et al., 2009) are relatively rare, limiting the extent of decision support that can be provided by these estimates. Various knowledge gaps also limit the capacity to accurately transfer these benefit estimates to other regions. Finally, most studies that have estimate the economic value of pollination services (Southwick and Southwick, 1992; Gallai et al., 2009a; Winfree et al., 2011; Ritter, 2013 – Table 9) have almost exclusively focused on the benefits to consumers rather than considering the potential benefits to producers from rising prices (but see Bauer and Wing, 2014).

Most studies focus on pollination services in their entirety – assuming a complete loss of wild and managed pollinators. While this demonstrates the benefits of pollinators as whole, it can under- or over-state the contextual importance of one group or the other, with several studies suggesting that managed pollinators are perfect substitutes for wild species (e.g., Winfree et al., 2011) or that wild species are incapable of fully replacing managed pollinators (e.g., Southwick and Southwick, 1992). In reality Garibaldi et al. (2013) demonstrates that in many systems, wild pollinators cannot be perfectly substituted with managed honeybees (the most widespread managed pollinator) and Rader et al. (2009) illustrate the contextual importance of both groups. Understanding and measuring the relative importance of both groups to crop production would allow for more targeted and effective management strategies.

Finally, as illustrated in the TEV diagram (Figure 1), the benefit to society offered by pollination service is broader than food production alone. The benefits of landscape aesthetics, wild plant diversity and crop genetic resources to present and future generation are also essential for the maintenance of the social welfare. However, very few studies have directly addressed this point, limiting the perspective of benefits to just the most overtly consumable (Mwebaze et al., 2010; Diffendorfer et al., 2014; Breeze et al., 2015).

22. Table 9 – Summary of estimates of the economic benefits of pollination services in 2015 US$.

Study	Region	Crops	Method	Year	2015US$	Notes
Farm/Local scale						
Shipp et al. (1994)	Canada	Sweet Peppers (cubico)	Yield Analysis	1992	$47,784-$75,190/ha	Study in an intensive glasshouse system
Ricketts et al. (2004)	Costa Rica	Forest fragments (Coffee)	Yield Analysis	2000-2003	$82,902/farm	
Olschewski et al. (2006)	Indonesia	Forest fragments (Coffee)	Yield Analysis	2001	$0-$63/ha of forest	Estimate represents the benefits of the pollinator capital per hectare of forest
Olschewski et al. (2006)	Ecuador	Forest fragments (Coffee)	Yield Analysis	2001	$0-$66/ha of forest	Estimate represents the benefits of the pollinator capital per hectare of forest
Whittington et al. (2004)	Canada	Tomatoes	Yield Analysis	2001	$434-$2,344/ha	Estimates vary between sites
De Marco and Coelho (2004)	Brazil	Coffee	Yield Analysis	2003	$2,415/ha	
Sandhu et al. (2008)	New Zealand	NA	Hive Rental	2004	$78-$81/ha	Estimates vary between sites
Nderitu et al. (2008)	Kenya	Sunflower	Yield Analysis	2005	$2,072/farm	
Mouton (2011)	South Africa	Apples (Granny smith)	Yield Analysis	2007/2008	$18,216/ha	
Lye et al. (2011)	UK	Raspberries	Yield Analysis	2010	$7,641/ha	
Ricketts and Lonsdorf (2013)	Costa Rica	Forest fragments (Coffee)	Production Function Modelling	2000-2003	$0-923/ha of forest	Estimate represents the benefits of the pollinator capital per hectare of forest
Regional Scale						
Turpie et al. (2003)	South Africa (Cape Florsitic Region)	All	Dependence Ratio	1999	$426.1M	
Greenleaf and Kremen (2006)	California, USA	Hybrid Sunflower	Yield Analysis	2002	$34.6M	Extrapolated from field studies. Based on the benefits of both wild and managed pollinators
Guerra-Sanz (2008)	Almeria, Spain	8 Glasshouse crops	Dependence Ratio	2002	$764.6M	
Allsopp et al. (2008)	South Africa (Cape Florsitic Region)	Apples, Plums, Apricots	Dependence Ratio	2005	$413.2M	
Allsopp et al. (2008)	South Africa (Cape Florsitic Region)	Apples, Plums, Apricots	Replacement Costs	2005	$94.2M-$529.7M	Estimates vary between different replacements
Chaplin-Kramer et al. (2011)	California, USA	All	Dependence Ratio	2007	$3.1bn-$7.2bn	
Barfield et al. (2015)	Georgia, USA	30 Crops	Dependence Ratio	2009	$673.8M	
Winfree et al. (2011)	New Jersey, USA	Watermelons	Yield Analysis	2009	$4.02M-$4.03M	

Winfree et al. (2011)	New Jersey, USA	Watermelons	Replacement Costs	2009	$0.2M-$0.23M	Replacement of existing honeybees and wild pollinators with new honeybees
Ritter (2013)	Oregon, USA	Blueberry	Partial Equilibrium Model (CS only)	2011	$9.7M-$11.8M	
National Scale						
Metcalf and Flint (1962)	USA	30 Crops	Total Crop Price	1957	$38.2bn	Incorrectly attributes all pollinated crop output to pollination
Levin (1984)	USA	All	Total Crop Price	1980	$54.5bn	Incorrectly attributes all pollinated crop output to pollination
Matheson and Schrader (1987)	New Zealand	All	Total Crop Price	1986	$2.6bn	Incorrectly attributes all pollinated crop output to pollination
Robinson et al. (1989)	USA	All	Dependence Ratio	1986	$20.3bn	
Gill et al. (1991)	Australia	35 Crops	Partial Equilibrium Model (CS only)	1986/1987av	$523M-$10,858M	Estimates vary based on the elasticity parameters used
Southwick and Southwick (1992)	USA	All	Partial Equilibrium Model (CS only)	1987	$3.4bn-$11.9bn	Estimates vary depending on whether honeybees are replaced with wild pollinators or not
Gill et al. (1989)	Australia	35 Crops	Partial Equilibrium Model (CS only)	1989	$0.9bn-$1.8bn	
Carreck and Williams (1998)	UK	All	Dependence Ratio	1996	$479.1M	
Calzoni and Speranza (1998)	Italy	Plums	Replacement Costs	1996	$394.1M	
Morse and Calderone (2000)	USA	All	Dependence Ratio	1996-1998av	$30.8bn	$21.8bn attributed to honeybees, $9bn to wild pollinators
Calderone (2012)	USA	All	Dependence Ratio	1996-2009	$15.8bn-$17.1bn	Estimates vary between years
Calderone (2012)	USA	All	Dependence Ratio	1997-2009av	$4,666-$7,311/ha	Estimates vary between years
Canadian Honey Council (2001)	Canada	All	Dependence Ratio	1998	$770.7M	
Gordon and Davis (2003)	Australia (honeybees)	35 Crops	Partial Equilibrium Model (CS and PS)	1999-2000av	$1.5bn	Consumer Surplus $720M; Producer Surplus $0-$762M depending on crop substitution
Cook et al. (2007)	Australia (honeybees)	25 Crops	Dependence Ratio	1999-2003av	$16.8M-$39.9M*	Benefits estimated in terms of losses avoided by the presence of a *Varroa* prevention programme. Estimates vary based on the elasticity parameters used
Losey and Vaughn (2006)	USA	51 Crops	Dependence Ratio	2003	$4.0bn	

Brading et al. (2009)	Egypt	All	Dependence Ratio	2004	$3.0bn	
Zych and Jakubiec (2006)	Poland	19 Crops	Dependence Ratio	2004	$311M	
Kasina et al. (2009)	Kenya (small holdings)	8 Crops	Yield Analysis	2005	$3.9M	
Sanjerehei (2014)	Iran	32 Crops	Dependence Ratio	2005-2006av	$7.9bn	
Basu et al. (2011)	India	6 Vegetable crops	Dependence Ratio	2007	$831.8M	
Basu et al. (2011)	India	6 Vegetable Crops	Partial Equilibrium Model (CS only)	2007	$1.5bn	
Smith et al. (2011)	UK	18 Crops	Dependence Ratio	2007	$986.1M	
An and Chen (2011)	China	44 Horticultural crops	Dependence Ratio	2008	$57.0bn	
Liu et al. (2011)	China	36 crops	Dependence Ratio	2006-2008av	$56.2bn	Honeybee pollination only
Mwebaze et al. (2010)	UK (pollinators)	NA	Stated Preferences (Contingent Valuation)	2009	$3.0bn	Estimates the existence value of honeybees
Stanley et al (2013)	Ireland	Oilseed Rape	Yield Analysis	2009-2011av	$5.8M	$3.9M winter oilseed rape; $1.9M spring oilseed rape
Garratt et al. (2014)	UK	Apples (2 Cultivars)	Yield Analysis	2010	$62.1M	Extrapolated to national scale from field studies
Calderone (2012)	USA	All	Dependence Ratio	2009	$17.9bn	
Breeze et al. (2015)	UK (pollination service benefits)	NA	Stated Preferences (Choice Experiments)	2010	$1.2bn-$640M	Estimates the existence value of bees, aesthetic wildflowers and UK produce. Estimates very between WTP bounds and upscale extrapolation used
Vanbergen et al. (2014)	UK	18 Crops	Dependence Ratio	2011	$1.2bn	Updated from Smith et al 2011
Giannini et al. (2015)	Brazil	85 Crops	Dependence Ratio	2012	$12.5bn	
Diffendorfer et al. (2014)	USA	NA	Stated Preferences (Contingent Valuation)	2012	$5.0bn-$6.9bn	Estimates the existence value of Monarch butterflies. Estimates cover lower and upper bound WTP
Multinational Scale						
Leonhardt et al. (2013)	Europe	All	Dependence Ratio	1991-2009av	$24.0bn	Estimates based on all EU member states in 2009 across all years
Partap et al. (2012)	Himalayan region (Bhutan, Pakistan, India, China, and Bangladesh)	All	Partial Equilibrium Model (CS only)	2008/09	$3.0bn	This does not include the Himalayan regions of Afghanistan, Northeast India, or Myanmar

Klatt et al. (2014)	EU	Strawberries	Yield Analysis	2009	$1.6bn	Estimates based on all EU member states in 2009
Global Scale						
Pimentel et al. (1997)	Global	All	Dependence Ratio	1986	$435.9bn	Assumes that global value is 5x Robinson et al., (1989)
Costanza et al. (1997)	Global	All	Total Crop Price	1996	$177bn	Incorrectly attributes all pollinated crop output to pollination
Bauer and Wing (2014)	Global	All	Partial Equilibrium Model	2004	$259.8bn-$351bn	Estimates vary based on the elasticity parameters used
Bauer and Wing (2014)	Global	All	General Equilibrium Model	2004	$367.9bn-$689.3bn	Estimates vary based on the elasticity parameters used. $160bn-$191bn loss to crop consumers, $207bn-$498bn lost to other markets
Gallai et al. (2009)	Global	All	Dependence Ratio	2005	$232.1bn	
Gallai et al. (2009)	Global	All	Partial Equilibrium Model (CS only)	2005	$176.2bn-$486bn	Estimates vary based on the elasticity parameters used
Lautenbach et al. (2012)	Global	All	Dependence Ratio	2009	$235.1bn-$577bn**	Estimates vary based on the dependence ratios used. Values are adjusted by Power Purchasing Parity

*: These values are subject to discounting on a 30-year time scale

**: These values are not reported directly in the paper but can be read from Figure 1 – confirmation of these values was provided by Sven Lautenbach

Study: The cited reference in which the original value was found. **Region**: The region over which the estimates of benefit was conducted. **Crops**: The crops that were assessed for value with all denoting all possible insect pollinated crops in the region for which data was available. NA denotes studies where the method does not apply to a specific crop. **Method**: Denotes the method used to estimate benefit: Crop Value (2.2.1), Hive Rental (2.1.2), Yield Analysis (2.2.1.), Dependence Ratio (2.2.2.), Replacement Costs (2.3), Partial Equilibrium Analysis (CS = Consumer Surplus; PS = Producer Surplus) and General Equilibrium Analysis (2.4) and Stated Preferences (2.5.). **Year**: the year the estimate relates to, usually based on what year the data relate to, studies denoted av = average of the years. **2015 US$**: The monetary estimate of the study inflated (and in many cases converted) to 2015 US$ as of July 2015 - this was done to standardize the estimates to some extent.

All estimates were converted into US dollars using average annual spot exchange rates from the Bank of England (Bank of England, 2015). These dollar estimates were inflated to 2015 US$ using Consumer Price Index (CPI) data from the United Stated Federal Government's Bureau of Labour and Statistics (BLS, 2015a, Table 24). Inflation was based on the CPI for July of the year the estimate was related to compared with the CPI in July 2015 (BLS 2015b, Table 1). If this year was not stated, then they were assumed to be the year before the study was published. Where a study used average data from across several years (e.g., Lui et al., 2011), conversion and inflation rates were averaged across the years concerned. These inflations only represent a change in the value of currency and do not capture any changes such as the relative input prices, price controls or subsidies. Note that the value of $1 will still vary between countries based on their purchase power piety (see Section 3).

23. Table 10 - Summary of estimates of the economic benefits of pollination services per hectare in 2015 US$ for several crops in different regions of the world.

Study	Region	Crops	Method	Year	2015US$/ha
Farm//local scale					
Shipp et al. (1994)	Canada	Sweet Peppers (cubico)	Yield Analysis	1992	$47,784-$75,190/ha
Priess et al. (2007)	Indonesia	Coffee	Yield Analysis	2001	$55.34/ha
Olschewski et al. (2006)	Indonesia	Coffee	Yield Analysis	2001	$63/ha
Olschewski et al. (2006)	Ecuador	Coffee	Yield Analysis	2001	$66/ha
Whittington et al. (2004)	Canada	Tomatoes	Yield Analysis	2001	$434-$2,344/ha
De Marco and Coelho (2004)	Brazil	Coffee	Yield Analysis	2003	$2415/ha
Ricketts et al. (2004)	Costa Rica	Coffee	Yield Analysis	2000-2003	$173/ha
Sandhu et al. (2008)	New Zealand	NA	Hive Rental	2004	$78-$81/ha
Nderitu et al. (2008)	Kenya	Sunflower	Yield Analysis	2005	$2072/farm
Lye et al. (2011)	UK	Raspberries	Yield Analysis	2010	$7641/ha
Mouton (2011)	South Africa	Apples (Granny smith)	Yield Analysis	2007/2008	$18,216/ha
Regional Scale					
Allsopp et al. (2008)	South Africa (Cape Florsitic Region)	Apples, Plums, Apricots	Dependence Ratio	2005	$12,579/ha
Allsopp et al. (2008)	South Africa (Cape Florsitic Region)	Apples, Plums, Apricots	Replacement Costs	2005	$2,867-$16,127/ha
Winfree et al. (2011)	New Jersey, USA	Watermelons	Partial Equilibrium Model (CS only)	2009	$5,393-$5,407/ha
Winfree et al. (2011)	New Jersey, USA	Watermelons	Replacement Costs	2009	$267-$312/ha
Ritter (2013)	Oregon, USA	Blueberry	Partial Equilibrium Model (CS only)	2011	$1,242-$1,510/ha
National Scale					
Carreck and Williams (1998)	UK	All	Dependence Ratio	1996	$842/ha
Calderone (2012)	USA	All	Dependence Ratio	1997-2009	$4,666-$7,311/ha
Kasina et al. (2009)	Kenya (small holdings)	8 Crops	Yield Analysis	2005	$163/ha
Basu et al. (2011)	India	6 Vegetable Crops	Dependence Ratio	2007	$458/ha

Basu et al. (2011)	India	6 Vegetable Crops	Partial Equilibrium Model (CS only)	2007	$804/ha
Smith et al. (2011)	UK	18 Crops	Dependence Ratio	2007	$1161/ha
Garratt et al. (2014)	UK	Apples (2 Cultivars)	Yield Analysis	2010	$20,199-$25,201
Vanbergen et al. (2014)	UK	18 Crops	Dependence Ratio	2011	$1,321/ha
Giannini et al. (2015)	Brazil	85 Crops	Dependence Ratio	2012	$1321/ha
Stanley et al. (2013)	Ireland	Oilseed Rape	Yield Analysis	2009-2011av	$652/ha
Multinational Scale					
Klatt et al. (2014)	EU	Strawberries	Yield Analysis	2009	$14,968/ha
Leonhardt et al. (2013)	Europe	All	Dependence Ratio	1991-2009av	$75/ha
Global Scale					
Costanza et al. (1997)	Global	All	Crop Value	1996	$34/ha
Bauer and Wing (2014)	Global	All	Partial Equilibrium Model	2004	$439-$526/ha
Bauer and Wing (2014)	Global	All	General Equilibrium Model	2004	$1,010-$1,891/ha
Gallai et al. (2009a)	Global	All	Dependence Ratio	2005	$624/ha
Gallai et al. (2009a)	Global	All	Surplus Analysis	2005	$473-$1,306/ha
Lautenbach et al. (2012)	Global	All	Dependence Ratio	2009	$717-$1,760/ha

Study: The cited reference in which the original value was found. **Region**: The region over which the estimates of benefit was conducted. **Crops**: The crops that were assessed for value with “All” denoting all possible insect pollinated crops in the region for which data was available. **Method**: Denotes the method used to estimate benefit: Crop Value (2.2.1), Hive Rental (2.1.2), Yield Analysis (2.2.1.), Dependence Ratio (2.2.2.), Replacement Costs (2.3), and Partial Equilibrium Analysis and General Equilibrium Analysis (2.4). **Year**: the year the estimate relates to, usually based on what year the data relate to, studies denoted av = average of the years. **2015 US$/ha**: The per hectare monetary estimate of the study inflated (and in many cases converted) to 2015 US$ as of July 2015 - this was done to standardize the estimates to some extent. Per hectare values were calculated by dividing the value estimates by the area of crop reported by either the paper itself or the data sources it cites.

All estimates were converted into US dollars using average annual spot exchange rates from the Bank of England (Bank of England, 2015). These dollar estimates were inflated to 2015 US$ using Consumer Price Index (CPI) data from the United Stated Federal Government’s Bureau of Labour and Statistics (BLS, 2015a, Table 24). Inflation was based on the CPI for July of the year the estimate was related to compared with the CPI in July 2015 (BLS 2015b, Table 1). If this year was not stated, then they were assumed to be the year before the study was published. These inflations only represent a change in the value of currency and do not capture any changes such as the relative input prices, price controls or subsidies. Note that the value of $1 will still vary between countries based on their purchase power piety (see Section 3).

24. Table 11 - Summary of the estimates of the economic value of pollination service to apple in 2015 $USD per hectare.

Study	Region	Crops	Method	Year	2015US$/Ha
Mouton (2011)	South Africa	Apples (Granny smith)	Yield Analysis	2007/08	$18,216
Garratt et al. (2014)	UK	Apples (Cox and Gala)	Yield Analysis	2010	$20,199-$25,201
Giannini et al. (2015)	Brazil	Apples	Dependence Ratio	2012	$7,715
Vanbergen et al. (2014)	UK	Dessert Apples	Dependence Ratio	2011	$18,902
Calderone (2012)	USA	Apples	Dependence Ratio	2010	$17,365
Leonhardt et al. (2013)	EU	Apples	Dependence Ratio	1991-2009av	$8,016
An and Chen (2011)	China	Apples	Dependence Ratio	2008	$10,399
Smith et al. (2011)	UK	Dessert Apples	Dependence Ratio	2007	$20,730
Calderone (2012)	USA	Apples	Dependence Ratio	2007	$21,774
Allsopp et al. (2008)	South Africa (Cape Florsitic Region)	Apples	Dependence Ratio	2005	$12,137
Gallai et al. (2009a)	Global	Apples	Dependence Ratio	2005	$3,776
Zych and Jakubiec (2006)	Poland	Apples	Dependence Ratio	2004	$1,566
Losey and Vaughn (2006)	USA	Apples	Dependence Ratio	2003	$13,078
Cook et al. (2007)	Australia	Apples	Dependence Ratio	1999-2003av	$15,229
Calderone (2012)	USA	Apples	Dependence Ratio	2002	$15,639
Morse and Calderone (2000)	USA	Apples	Dependence Ratio	1996-1998av	$10,654
Allsopp et al. (2008)	South Africa (Cape Florsitic Region)	Apples	Replacement Costs	2005	$791-$1,634
Partap et al. (2012)	Himalayan region	Apples	Partial Equilibrium Model (CS only)	2008/2009	$3,975
Gallai et al. (2009a)	Global	Apples	Partial Equilibrium Model (CS only)	2005	$6,083

Study: The cited reference in which the original value was found. **Region**: The region over which the estimates of benefit was conducted. **Crops**: The crops that were assessed for value with all denoting all possible insect pollinated crops in the region for which data was available. NA denotes studies where the method does not apply to a specific crop. **Method**: Denotes the method used to estimate benefit: Yield Analysis (2.2.1.), Dependence Ratio (2.2.2.), Replacement Costs (2.3) and Partial Equilibrium Analysis (2.4). **Year**: the year the estimate relates to, usually based on what year the data relate to, studies denoted av = average of the years. **2015 US$**: The monetary estimate of the study inflated (and in many cases converted) to 2015 US$ as of July 2015 - this was done to standardize the estimates and facilitate comparison.

All estimates were converted into US dollars using average annual spot exchange rates from the Bank of England (Bank of England, 2015). These dollar estimates were inflated to 2015 US$ using Consumer Price Index (CPI) data from the United Stated Federal Government's Bureau of Labour and Statistics (United States Bureau of Labour and Statistics, 2015a, Table 24). Inflation was based on the CPI for July of the year the estimate was related to compare with the CPI in July 2015 (BLS, 2015b, Table 1). If this year was not stated, then they were assumed to be the year before the study was published. These inflations only represent a change in the value of currency and do not capture any changes such as the relative input prices, price controls or subsidies. Note that the value of $1 will still vary between countries based on their purchase power piety (see Section 3). Where the area of apples was not reported within the study, the source material for the value of apple production was consulted and area data for the appropriate year were used to calculate these values.

Section 8 – Synthesis and conclusion

This chapter reviewed the conceptual framework and the different methods of economic valuation of pollinators and pollination services. Thus, more than 60 economic valuations of pollination were analysed at different spatial and temporal scales (Table 9, Section 7). These findings demonstrate the substantial economic benefits derived from pollinators and pollination in food production and biodiversity on several components of social welfare as represented by the different economic values (monetary and non-monetary).

The TEV of pollinators and pollination services

The chapter has identified and adapted the economics behind pollinators and pollination services. As explained in Section 1, economic theory gives a well-defined framework to comprehend the status and the value of pollinators and pollination for human wellbeing. There are multiple values identified by the TEV (Total Economic Value) associated pollinators and pollination services diagram (Figure 1). The chapter has highlighted the breadth of benefits that pollinators and pollination services provide within the TEV framework, while the literatures has to date only considered pollination as a provision service and an indirect service (see for example Pascual et al., 2010; Fisher et al., 2009). Pollinators and the benefits they produce through pollination services can be both marketed (honey bees, crops) or non-marketed (wild pollinators, aesthetic wildflowers). While pollinators can be rival, for many crops and wild plants that depend on cross-pollination, their services are non-rival. As such, pollinators often provide valuable, potentially irreplaceable services to human wellbeing. However, despite the breadth of possible benefits, to date, attempts to value these benefits are largely confined to crop pollination services (Section 7), leaving many aspects of pollination services unvalued.

A well-structured framework of methods that largely remains to be applied

A wide range of methods have been developed and used to value the contribution of pollinators and pollination to our society, but also to address the economic consequences of their gains or losses, including both their use (Section 2.2, 2.3 and 2.4) and non-use values (Section 2.5). However, to date, the majority of these methods (Section 2) and the studies applying them (Section 7) do not estimate the true economic value of these changes. Furthermore, many of these methods are limited by available data (Section 5) and are only suitable for application on specific spatial scales (Section 3), or under very specific niche circumstances (Section 2). On local scales, where a shift in pollination services is unlikely to cause price changes, production function models (Section 2.2.3) are more relevant to estimate the impacts of pollinator gains and losses on local producers. On larger scales, however, production function models are better suited to inform more comprehensive surplus valuation models that estimate the impacts on both producers and consumer welfare (Section 2.4).

How to account for the spatial and the temporal scale?

The scales at which ecological processes occur can be different to those at which economic decisions are made. Not taking account of scales could generate biased economic valuations by assuming that benefits are more consistent across time and space than they are. The chapter has adapted existing categories of temporal and spatial scales to encompass the diverse array of variables that affect pollination valuation (Section 3; Table 5). Considering the temporal scale of ecological and/or agronomical processes is essential, whether to understand the renewal rate of pollinator populations or the timing of crop production, among others. It is important that studies consider a range of market prices and productions cycles, but also more theoretical factors such as the discount rate that represent the way we value the future and, the availability of consistent, long-term data sets. Some tools exist in order to address long-term economic valuations, such as the scenario or time-series analyses but to date their use in valuing pollination services has been limited. Considering spatial scale is also fundamental to valuation and land-use decisions, as mismatches can undermine the distribution of economic and conservationist benefits originated from the pollination service quality, with different approaches required between the micro-, meso- or macroeconomic levels. Declining data quality on large scales could be overcome by broader and more detailed record keeping and several spatially explicit methods are available to support multi-scale assessments of pollination benefits, including the effects of landscape design. Although these adaptations are possible within existing methodologies, they have rarely been applied, leaving numerous questions regarding the likely variation of pollination service benefits across the world and to future generations.

The value of pollinators and pollination services also involves risk, uncertainty and resilience values

Although pollinator gains and losses can affect both the levels of pollination services and the potential for future services provided, to date, no study has explicitly quantified the economic risks and uncertainties inherent to populations (Section 4) and few have addressed the uncertainties within the data used to estimate these impacts (Section 7). While a number of suitable methods exist (Section 4.3), they have yet to be applied to pollinator management. Without this information, decision-making may be at risk of over-valuing benefits or under-valuing impacts from management affecting pollinators and populations, particularly over longer time periods (Section 3).

Guiding decision-makers in protecting, maintaining and enhancing pollinators and pollination services, for society

Economic analysis provides powerful information for decision-makers for many reasons. Throughout the chapter, we have defined the status of pollinators and pollination services in relation to property right structure (private good, club good, common good or public good, Section 1), explained how to estimate the (use- or non-use) value of pollinators and pollination, and reviewed the main values.

The type of property rights informs the stakeholder of their level of implication in maintaining the natural service. The estimated value of pollinators or pollination generates a monetary (or non-monetary) indicator that gathers information on the positive or negative impact of pollinators or pollination gain or loss. This indicator can be used in a number of forms including cost benefit (and cost-effectiveness) analysis, Multi-Criteria Analysis, environmental accounting and decision support tools (Section 6). The use of economic valuation varies between stakeholders; a farmer will not use the values in the same way, or for the same reasons, than an industry or a government. This is why the chapter presents the different ways to address the economic value for each level of stakeholder as well as the step-wise guide for using economic valuation for decision-making (Section 6).

Conclusion

The economic valuation of pollinators and pollination services is, in many contexts, an essential step for decision-making by governments and policy makers. Although many studies have been done, they mainly concentrate on the provision role of pollinators while the impact of pollinators on our society is much broader (e.g., the pollination of wild plants that enhance the biodiversity of landscapes or the marginal value of wild pollinators). Furthermore, few of them actively consider these benefits in relation to the costs of management to sustain them (Chapter 6) or, conversely, the benefits of management that may be detrimental to pollinators (Chapter 2). Understanding and quantifying these trade-offs is essential for informed policy and decision making at all scales, but particularly over the long term (Section 3) where a lack of sustainability may hamper resilience (Section 4).

Even more importantly, more comprehensive assessments of the economic impacts of pollinator gains and losses are needed to improve the measurements of the welfare consequences on changing pollinator populations. Further work is required to accurately estimate the benefit on crop production and non-crop production, the impacts on present and future generations, and the local and international consequences. The methods of economic valuation should be developed in this way, taking into account both market and non-market-based approaches. Furthermore, many of the methods would benefit from standardization in order to facilitate the aggregation and comparison of values gathered around the world and over time.

The concept of value is broad and it goes beyond a mere economic approach (Díaz et al., 2015). Chapter 5 addresses these other broader forms of values. Determining the full plurality of these values will be necessary to guide decisions that affect pollinators and secure these benefits for future generations. Chapter 6 gives a detailed presentation of the different tools and existing policies to help maintain pollinators, and their implementation that will strongly benefit from robust valuations of the numerous benefits of pollination services.

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Glossary

.- Benefit – The positive impacts produced by pollinators and pollination services (e.g., increased yield or quality of crops).

.- Capital – Any good, service or skill that can potentially generate production within a market. There are 5 forms of capital: Human (skills, education etc.), Manufactured (tools, buildings etc.), Financial (shares, bonds etc.), Social (institutions etc.) and natural (ecosystem services etc.). Units of capital are called assets. The sum of capital is called wealth.

.- Profit – It is the difference between the benefit of a firm and her total cost, where total cost is the sum of fixed and variable costs.

.- Consumer surplus: Consumer Surplus is defined as the difference between what consummer would accept to pay (WTP) to get a service and the cost they actually bear.

-Producer Surplus: Producer surplus is the difference the amount that a producer willing to sell a good (his marginal cost) and the amount that he receives.

.- Welfare: The welfare measure the well-being of a society. One method to measure the welfare is to summing the producer and the consumer surpluses. A more practical way to measure it is the Growth National Product per capita.

.- Cost-Benefit Analysis (CBA): The CBA is a method where it is evaluating in monetary terms the environmtal impact of a project or an event (e.g., the climate change) and assessing the beenfits and the costs associated with different options of the project or to reduce the event (e.g., reducing the climate change).

.- Economic vulnerability – vulnerability refers to the possibility that the environment could be degraded. Economic vulnerability can be declined in firms' vulnerability and consumers' vulnerability. The firms vulnerability would be the potential loss in profit due to pollinators loss and the consumer vulnerability would be the potential loss in utility due to pollinators loss. The vulnerability concept has been broadly study in the literature. We will retain one definition from Turner et al. (2003)[8] where vulnerability is a function of three overlapping elements: exposure, sensitivity, and adaptive capacity.

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[8] Turner, B. L., Kasperson, R. E., Matson, P. A., McCarthy, J. J., Corell, R. W., Christensen, L., Eckley, N., Kasperson, J. X., Luers, A., Martello, M. L., Polsky, C., Pulsipher, A., and Schiller, A. (2003). A framework for vulnerability analysis in sustainability science. *Proceedings of the National Academy of Sciences of the United States of America*, *100*(14), 8074–8079. doi:10.1073/pnas.1231335100

.- Intrinsic value – It is the value with give to pollination service just because the benefit of this service is good in and of itself. We are not supposed to use in order to acquiring something else. Intrensic value of pollinators is the value of their existence.

.- Instrumental value – It is a good for which we give a value because it provides the mean s for acquiring something else of value. Instrumental value of pollinators is the service provide by their activity as honey or crop production.

.- Monetary valuation – it is the valuation in money of the environmental service offers by pollinators.

.- Net present value – It is a temporal financial expression. It is the sum of actualized future cash flow, both incoming and outgoing.

.- Non-monetary valuation: It is the valuation of the impact of an environemental service in the society not expressed in money. This valuation can be quantitative (e.g., loss in CO2 production) and/or qualitative (e.g., sense of the impact positive or negative).

.- Price – The market (or pseudo-market) exchange value of a good or service.

.- Production functions – it is the function that model the process of transformation of inputs into final output. It could be also defined as the process to convert costs into revenue.

.- Purchasing power parity – value of money expressed in terms of units of goods that money can command.

.- Sustainability - A development process economically efficient, socially equitable and environmentally stable that will enable future generations to be at least as happy as we are.

.- Value – The impact of pollinators and pollination services on welfare via changes in benefits. This can be measured in economic or social terms.

Chapter 5: Biocultural diversity, pollinators and their socio-cultural values

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Executive Summary

Diverse knowledge systems, including science and indigenous and local knowledge (ILK), contribute to understanding pollinators and pollination, their economic, environmental and socio-cultural values and their management globally (*well established*). Scientific knowledge provides extensive and multidimensional understanding of pollinators and pollination, resulting in detailed understanding of their diversity, functions and steps needed to protect pollinators and the values they produce. In indigenous and local knowledge systems, pollination processes are often understood, celebrated and managed holistically in terms of maintaining values through fostering fertility, fecundity, spirituality and diversity of farms, gardens, and other habitats. The combined use of economic, socio-cultural and holistic valuation of pollinator gains and losses, using multiple knowledge systems, brings different perspectives from different stakeholder groups, providing more information for the management of and decision-making about pollinators and pollination, although key knowledge gaps remain (5.1.1, 5.1.2, 5.1.3., 5.1.4, 5.1.5, 5.2.1., 5.2.5, 5.3.1, 5.5, figure 5-2, boxes 5-1, 5-2).

Pollinator-dependent food products are important contributors to healthy human diets and nutritional security (*well established*). Crop plants that depend fully or partially on animal pollinators contain more than 90% of vitamin C, most of lycopene, the antioxidants beta-cryptoxanthin and beta-tocopherol, vitamin A and related carotenoids, calcium and fluoride, and a large portion of folic acid available worldwide. Pollinator insects, including the larvae of beetles, moths, bees, and palm weevils constitute a significant proportion of ~ 2,000 insect species consumed globally, recognised as potentially important for food security, being high in protein, vitamins and minerals (5.2.2).

Pollinators are a source of multiple benefits to people, well beyond food-provisioning alone, contributing directly to medicines, biofuels, fibres, construction materials, musical instruments, arts and crafts and as sources of inspiration for art, music, literature, religion and technology (*well established*). For example, anti-bacterial, anti-fungal and anti-diabetic agents are derived from honey; *Jatropha* oil, cotton and eucalyptus trees are examples of pollinator-dependent biofuel, fibre and timber sources respectively; beeswax can be used to protect and maintain fine musical instruments. Artistic, literary and religious inspiration from pollinators includes popular and classical music (e.g., I'm a King Bee by Slim Harpo, the flight of the Bumblebee by Rimsky-Korsakov); sacred passages about bees in the Mayan codices (e.g., stingless bees), the *Surat An-Naĥl* in the Qur'an, the three-bee motif of Pope Urban VIII in the Vatican and sacred passages from Hinduism, Buddhism and Chinese traditions such as the Chuang Tzu. Pollinator-inspired technical design is reflected in the visually guided flight of robots, and the 10 metre telescopic nets used by some amateur entomologists today (5.2.1, 5.2.2, 5.2.3, 5.2.4 case examples 5-2, 5-16, and figures 5-7, 5-8, 5-9, 5-10, 5-24).

Livelihoods based on beekeeping and honey hunting are an anchor for many rural economies and are the source of multiple educational and recreational benefits in both rural and urban contexts (*well established*). Many rural economies favour beekeeping and honey hunting, as minimal investment is required; diverse products can be sold; diverse forms of ownership support access; family nutrition and medicinal benefits can be derived from it; the timing and location of activities are flexible; and numerous links exist with cultural and social institutions. Beekeeping has been identified as a potentially effective intervention tool for reducing relapses in youth criminal behaviour; a rapidly expanding ecologically-inspired urban lifestyle choice; a source for the growing market demand for local honey; the basis for gaining and transmitting knowledge about ecological processes; and a tool for empowering youth to link biodiversity, culture and society and take action on issues of environmental impacts on pollinators and pollination. Significant unrealized potential exists for beekeeping as a sustainable livelihood activity in developing world contexts (5.2.8.4, 5.3.5, 5.4.6.1, case examples 5-10, 5-11, 5-12, 5-13, 5-14, 5-21, 5-24, 5-25, and figures 5-12, 5-13, 5-14, 5-15, 5-22).

A number of cultural practices based on indigenous and local knowledge contribute to supporting an abundance and diversity of pollinators and maintaining valued "biocultural diversity" (for the purposes of this assessment, biological and cultural diversity and the links between them are referred to as "biocultural diversity") (*established but incomplete*). This includes practices of diverse farming system; of favouring heterogeneity in landscapes and gardens; of kinship relationships that protect many specific pollinators; of using biotemporal indicators that rely on distinguishing a great range of pollinators; and of tending to the conservation of nesting trees, floral and other pollinator resources. The ongoing linkages among these cultural practices, the underpinning indigenous and local knowledge (including multiple local language names for diverse pollinators) and pollinators constitute elements of "biocultural diversity"[9]. Areas where "biocultural diversity" is maintained are valued globally for their roles in protecting both threatened species and endangered languages. While the extent of these areas is clearly considerable, for example extending over 30 per cent of forests in developing countries, key gaps remain in the understanding of their location, status and trends (5.1.3, 5.2.5, 5.2.6, 5.2.7, 5.4.7.2, case examples 5-1, 5-3, 5-5, 5-6, figures 5-4, 5-11).

Diversified farming systems, some linked to indigenous and local knowledge, represent an important pollination-friendly addition to industrial agriculture and include swidden, home gardens, commodity agroforestry and bee farming systems (*established but incomplete*). While small holdings (less than 2 hectares) constitute about 8-16 per cent of global farm land, large gaps exist in our knowledge on the area of diversified farming systems linked to indigenous and local knowledge. Diversified farming systems foster agro-biodiversity and pollination through crop rotation, the promotion of habitat at diverse stages of succession, diversity and abundance of floral resources; ongoing incorporation of wild resources and inclusion of tree canopy species; innovations, for example, in apiaries, swarm capture, and pest control; and adaptation to social-environmental change, for example, the incorporation of new invasive bee species and pollination resources into farming practices (5.2.8, case examples 5-7, 5-8, 5-9, 5-10, 5-11, 5-12, 5-13, and figures 5-14, 5-15, 5-22).

A good quality of life for many people relies on ongoing roles of pollinators in globally significant heritage; as symbols of identity; as aesthetically significant landscapes and animals, in social relations, for education and recreation in governance interactions of indigenous peoples and local communities (*well established*). As examples, the World Heritage site the Agave Landscape and Ancient Industrial Facilities of Tequila depends on bat pollination to maintain agave genetic diversity and health; people show marked aesthetic preferences for the flowering season in diverse European cultural landscapes; a hummingbird is the national symbol of Jamaica, a sunbird of Singapore, and an endemic birdwing the national butterfly of Sri Lanka; seven-foot wide butterfly masks symbolize fertility in festivals of Bwa people of Burkina Faso; and the Tagbanua people of the Philippines, according to their tradition, interact with two bee deities living in the forest and karst as the ultimate authority for their shifting agriculture (5.3.1, 5.3.2, 5.3.3, 5.3.4, 5.3.6, case examples 5-16, 5-17, 5-18, 5-19 and 5-20, and figures 5-16, 5-17, 5-18, 5-19, 5-20, 5-21).

Managing and mitigating the impacts of the declines on pollinators' decline on peoples' good quality of life could benefit from responses that address loss of access to traditional territories, changes to

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[9] In the IPBES Conceptual Framework the definition of biocultural diversity is "the total variety exhibited by the world's natural and cultural systems, explicitly considers the idea that culture and nature are mutually constituting, and denotes three concepts: Firstly, diversity of life includes human cultures and languages; secondly, links exist between biodiversity and cultural diversity; and finally, these links have developed over time through mutual adaptation and possibly co-evolution. Biocultural diversity incorporates ethnobiodiversity" (Diaz et al., 2015)

traditional knowledge, tenure and governance, and the interacting, cumulative effects of direct drivers (*established but incomplete*). A number of integrated responses that address these drivers of pollinator declines have been identified: 1) food security, including the ability to determine one's own agricultural and food policies, resilience and ecological intensification; 2); conservation of biological diversity and cultural diversity and the links between them; 3) strengthening traditional governance that supports pollinators; rights-based approaches; 4) prior and informed consent for conservation, development and knowledge-sharing; 5) recognizing tenure); 6) recognizing significant agricultural, biological and cultural heritage, and 7) framing conservation to link with peoples' values (5.4, case examples 5-18, 5-19, 5-20, 5-21, 5-22, 5-23, 5-24, 5-25, 5-26, figures 5-26, 5-27, box 5-3).

Indigenous and local knowledge systems, in co-production with science, can be sources of solutions for the present challenges confronting pollinators and pollination (*established but incomplete*). Knowledge co-production activities among farmers, indigenous peoples, local communities and scientists have led to numerous relevant insights including: improvements in hive design for bee health, understanding pesticide uptake into medicinal plants and the impacts of mistletoe parasite on pollinator resources; identification of species of stingless bee new to science; establishing baselines to understand trends in pollinators; improvements in economic returns from forest honey; identification of change from traditional shade-grown to sun grown coffee as the cause of declines in migratory bird populations; and a policy response to risk of harm to pollinators leading to a restriction on the use of neonicotinoids in the European Union (5.4.1, 5.4.2.2., 5.4.7.3, tables 5-4 and 5-5).

Many actions to support pollinators are hampered in their implementation through governance deficits, including fragmented multi-level administrative units, mismatches between fine-scale variation in practices that protect pollinators and homogenizing broad-scale government policy, contradictory policy goals across sectors and contests over land use (*established but incomplete*). Co-ordinated, collaborative action and knowledge sharing that forges linkages across sectors (e.g., agriculture and nature conservation), across jurisdictions (e.g., private, Government, not-for-profit), and among levels (e.g., local, national, global) can overcome many of these governance deficits. The establishment of social norms, habits, and motivation that are the key to effective governance outcomes involves long time frames (5.4.2.8, 5.4.7.4).

Foreword to Chapter 5

Pollination, there are many pollinators, not just bees. For example, the birds that fly from one place to another. Bees fly from one branch to another and carry with them the pollen and maybe we see a change in the colour of the trees. An ant visits a flower, travelling to another one, carrying the pollen from one to the next ... Seeing all of this, I have to say that the Guna have a different way of seeing things. We don't see things in their parts, everything is more holistic. When we see a human being, we don't just see two ears, that person has his or her own intelligence. We all need each other—animals, plants and humans. All beings are alive—rocks have their spirit because they help us, perhaps in traditional medicine. Our world is very different, no one dedicates him or herself to just one activity. Belisario López, oral presentation p.41 (López et al. 2015) (**Error! Reference source not found.**.).

We do not see pollination as a separate theme. Rather that everything— trees, rivers, the wind, even human beings—participates in the process. We cannot separate them. Elmer Enrico Gonzalez López, oral presentation p 42 (López et al. 2015).

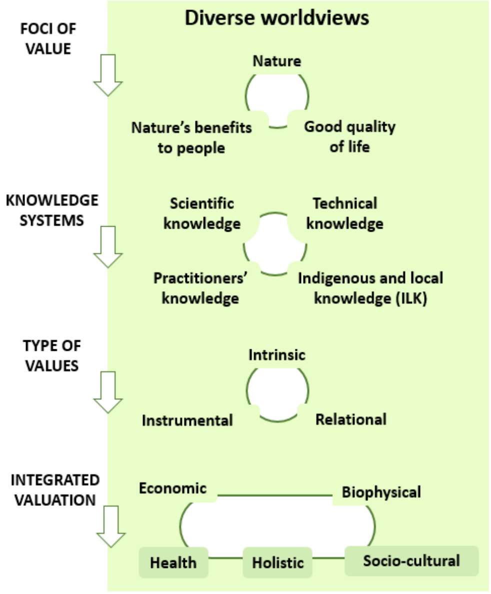
A group of Guna people, as representatives of the host people, attended the *Global Dialogue Workshop on ILK of pollination and pollinators associated with food production*, Panama City, 1-5 December 2014 (Lyver et al., 2015). These quotations are taken from their oral presentations at the Workshop.

37. Figure 5-1. Mola, embroidered cloth made by Guna people, of bee and butterfly spirits.
© The Guna People.
The use of this image is a collective right owned by the Guna People, that has been authorized by the Guna General Congress according to the Resolution No. 1 of 22 November 2002 issued by the Department of Industrial Property Registry of the Ministry of Commerce and Industry.

5.1 Introduction

5.1.1 Diversity of knowledge systems and the IPBES Conceptual Framework

This chapter addresses the topics identified in the scoping study (IPBES 2/17, p. 71) as "non-economic valuation, with special emphasis on the experience of indigenous and local communities, of impacts of the decline of diversity and/or populations of pollinators… Management and mitigation options as appropriate to different visions, approaches and knowledge systems". The IPBES Conceptual Framework, which recognises that the world views of people influence their understandings about nature, and nature's benefits to people and good quality of life, underpins the approach to the chapter (Díaz et al., 2015a). For example, nature's benefits to people can be understood as ecosystem services, such as those provided by bees to pollinate several of the world's main crops (Gallai et al., 2009); and as gifts of the gods, as stingless bees and beekeeping are understood among Mayan-descendant people (Sharer, 2006). Multiple knowledge systems of people, including scientific, technical, practitioner and indigenous and local knowledge systems, influence how pollination is understood and valued. Values and knowledge systems are dynamic, changing in response to new information, and to socio-cultural embeddedness and multidimensionality (Brondizio et al., 2010). Assessment of the values of the contribution of pollination and pollinators to nature's benefits to people, and to good quality of life, therefore requires diverse valuation methods (IPBES, 2015). In this chapter, we provide an assessment of these values, focusing on scientific and indigenous and local peoples' knowledge (ILK) systems, and on socio-cultural and holistic valuation approaches (Figure 5-2.). Chapter 2 and Chapter 4 provide assessments based on biophysical and economic valuation approaches respectively.



38. Figure 5-2. Diverse world-views, knowledge systems, types of values and valuation approaches for assessing nature, nature's benefits to people, and good quality of life.
(Based on IPBES, 2015).

Focusing on different knowledge systems brings greater depth and breadth to our understanding of the value of pollination and pollinators (IPBES, 2015). There are several dimensions that characterise the differences between knowledge systems. These include concepts about what constitutes valid knowledge and how we can obtain it—its epistemology—including domains such as truth criteria, rules of transmission and of validation, attribution of authorship or other rights over knowledge, and many others (Crotty, 1998; Cash et al., 2003; Vadrot, 2014). For example, the notion of individual authorship has become prevalent in Western thought since the late seventeenth century, whereas authorship of songs and poetry is most often attributed to spirits or enemies among Amerindian peoples. Knowledge authority may depend on having been acquired from a chain of authorized knowledge holders, or on first-hand experience, body training or life and dream experience. Knowledge can be esoteric, reserved to some holders such as male children, or exoteric, shared and transmitted openly with anyone in the community (Carneiro da Cunha, 2009, 2012).

A system of knowledge is also distinguished from others according to its ideas about what constitutes reality, about what kinds of things exist — its ontology (Descola, 2014). The world is not just a given, a “reality”, that we simply capture through our senses. Rather, clusters of environmental qualities are understood through “ontological filters”, that allow us to look for certain qualities and detect them, while we ignore others. For example, the Tuawhenua Māori of New Zealand recognize that people, bats, birds, insects, plants, mountains, rivers and lakes are connected together by genealogical ties (*tatai whakapapa*). When a child is born, these ties are enacted by the burying of the placenta and umbilical cord on tribal lands, thus consolidating ties to Papatuanuku, Mother Earth (Doherty and Tumarae-Teka, 2015). In Bangka-Belitung, Indonesia “where spirits are everywhere, the use of natural resources (terrestrial and aquatic) within a territory is supported by custom (*adat*) and the village authority (the *dukun kampung*) who acts as an intermediary between villagers and the local spirits” (Césard and Heri, 2015).

In contrast, seeing nature as separate from culture became dominant in Western societies after the 17th Century, based on Descartes’ portrayal of human beings as masters of nature (Descartes, 1637 [2005]), and the expectation that Newtonian mechanics could predict nature’s behavior by mathematical rules and monitor it by command-and-control systems, removing ideas about spiritual influences (Newton, 1687; [2014], Davoudi, 2014). More recently, contemporary conservation science itself has been characterised as moving from nature – people dualism towards a framing around “people and nature”, which has benefits as well as risks (Mace, 2014). This shift is partly in response to the narrowness and market-orientation of the ecosystem services framework (Turnhout et al., 2014). Sustainability challenges have shifted science towards embracing pluralism and co-production with other knowledge systems through interdisciplinary and transdisciplinary approaches (Repko, 2012). Indigenous and local knowledge systems also change; for example indigenous communities in Australia have adapted to take account of myrtle rust, a serious fungal disease affecting flowers and spread by insect pollinators, among other agents, developing new partnerships with scientists to co-produce knowledge and management (Robinson et al., 2015).

The IPBES Conceptual Framework provides a basis to be inclusive of, and provide linkages among, this wide array of knowledge systems, with their diverse ontologies and epistemologies (Díaz et al., 2015b). While differences among knowledge systems can create profound misunderstandings, people can find points of connection, agreeing on phenomena while disagreeing on their interpretation (da Costa and French, 2003; Almeida, 2013). Diverse knowledge systems can provide a multiple evidence base, leading to a richer understanding and more effective policy-relevant information (Tengö et al., 2014). The remainder of this introductory section explains and justifies our focus on science and ILK; the linkages with the concept of biocultural diversity; the socio-cultural and holistic valuation approaches, and associated categories adopted. Parts two and three of the chapter present an assessment of the values associated with the contribution of pollination and pollinators to *nature's benefits to people*, and part three *to good quality of life*. Part four considers the impacts of declines of pollinators and pollination on these values, and vice versa, and potential management and mitigation options. The methods for conducting the assessment are presented in part 5, and part 6 presents the conclusions from this chapter.

5.1.2 Focus on scientific and indigenous and local knowledge systems

The focus on scientific knowledge systems for this pollination assessment is fundamental, as IPBES was established with the overall goal of 'strengthening the science-policy interface for biodiversity and ecosystem services for the conservation and sustainable use of biodiversity, long-term human well-being and sustainable development'[10]. Scientific contributions to understanding pollinators and pollination are extensive and multidimensional, stimulated by Camerarius' first empirical demonstration in 1694 that plants reproduce sexually (Ducker and Knox, 1985), and Darwin's (Darwin, 1862 [2004]) book on the pollination of orchids. Pollinator and pollination science now includes diverse aspects across the ecology of both wild and domesticated pollinator communities and habitats, the genomics of pollinator-dependent species, the molecular biology of pollinator-attractants produced by flowers, the influence of drivers of environmental change, knowledge of substances such as pesticides, and more. Several contemporary journals and research centres are devoted entirely to aspects of the science of pollination, e.g., Journal of Pollination Ecology and the Center for Pollinator Research at Pennsylvania State University.

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[10] http:www.ipbes.net.au

In addition to this fundamental focus on scientific knowledge, IPBES has adopted as one of eleven guiding principles, a commitment to 'recognize and respect the contribution of indigenous and local knowledge to the conservation and sustainable use of biodiversity and ecosystems'. Indigenous and local knowledge (ILK) systems are highly diverse and dynamic, existing at the interface between the enormous diversity of ecosystems worldwide and the diversity of livelihood systems (e.g., farmers, fishers, beekeepers, pastoralists, hunter-gatherers, etc.) (Thaman et al., 2013). Our treatment of ILK systems here is guided by definitions that recognize the complexity, diversity and dynamism of human communities, and that self-identification, rather than formal definition, is the key (Martinez-Cobo, 1986; ILO, 1989; Borrini-Feyerabend and Hill, 2015). Indigenous societies share common characteristics such as being linked to territories, having continued occupation of those territories over long times, and operating under their own customary law systems. Local peoples are characterized by living together in a common territory where they frequent face-to-face interactions, share aspects of livelihoods, and approaches such as collective management of common property or particular farming practices (Box 5-1).

Box 5-1. Who are indigenous peoples and local communities?

The United Nations recognizes that no formal definition of whom are indigenous peoples and/or local communities is needed — self-identification is the key requirement. This assessment is guided by discussions that recognize the complexity, diversity and dynamism of human communities (Martinez-Cobo, 1986; ILO, 1989; Borrini-Feyerabend and Hill, 2015).

Indigenous peoples include communities, tribal groups and nations, who self-identify as indigenous to the territories they occupy, and whose organisation is based fully or partially on their own customs, traditions, and laws. Indigenous peoples have historical continuity with societies present at the time of conquest or colonisation by peoples with whom they now often share their territories. Indigenous peoples consider themselves distinct from other sectors of the societies now prevailing on all or part of their territories.

Local communities are groups of people living together in a common territory, where they are likely to have face-to-face encounters and/or mutual influences in their daily lives. These interactions usually involve aspects of livelihoods — such as managing natural resources held as 'commons', sharing knowledge, practices and culture. Local communities may be settled together or they may be mobile according to seasons and customary practices. Self-identification is also the key determinant of whether people consider themselves to be local communities.

Communities that come together in urban settings around common interests, such as beekeeping, are considered here to be "communities of interest" rather than local communities.

Dynamism is also a key characteristic of indigenous peoples' and local communities' knowledge systems (ILKS), reflecting innovations, as well as a history of interactions with other peoples through trans-continental contacts over millennia, migrations, and the more recent processes of colonization and postcolonial assertion of rights (Coombes et al., 2013; Roullier et al., 2013). Guided by Berkes (2012) and Díaz et al.'s 2015 definition we consider ILK systems to be cumulative bodies of knowledge, practice and belief, evolving by adaptive processes and transmitted through cultural and intergenerational processes, about the relationship of living beings (including humans) with one another and with their environment.

In many cases, management based on ILK systems has produced sustainably over millennia; in other cases, ILK-based systems have proved mal-adaptive and had a major destructive influence on biodiversity and associated pollinators, sometimes leading to the disintegration of human societies (Diamond, 2005). Ostrom (1990) established that the types of institutional arrangements that support common property systems of governance are critical determinants of whether sustainability results from local management systems. ILK that is relevant to pollinators and pollination therefore importantly includes knowledge of social institutions and governance systems that foster sustainable relationships with pollinators, as well as environmental observations, interpretations, and resource use practices (Berkes and Turner, 2006; Gómez-Baggethun et al., 2013). Language, naming and classification systems, rituals, spirituality and worldviews are integral to ILKS (ICSU, 2002). Validity of ILK arises from the relevant societies exercising their ability to generate, transform, transmit, hybridize, apply and validate knowledge (Tengö et al., 2014); understanding ILK *in-situ* is therefore the priority in working with ILK in biodiversity assessment, rather than a focus on knowledge extracted into literature and other forms (Gómez-Baggethun and Reyes-García, 2013) (Box 5-2).

Box 5-2. What are indigenous and local knowledge systems?

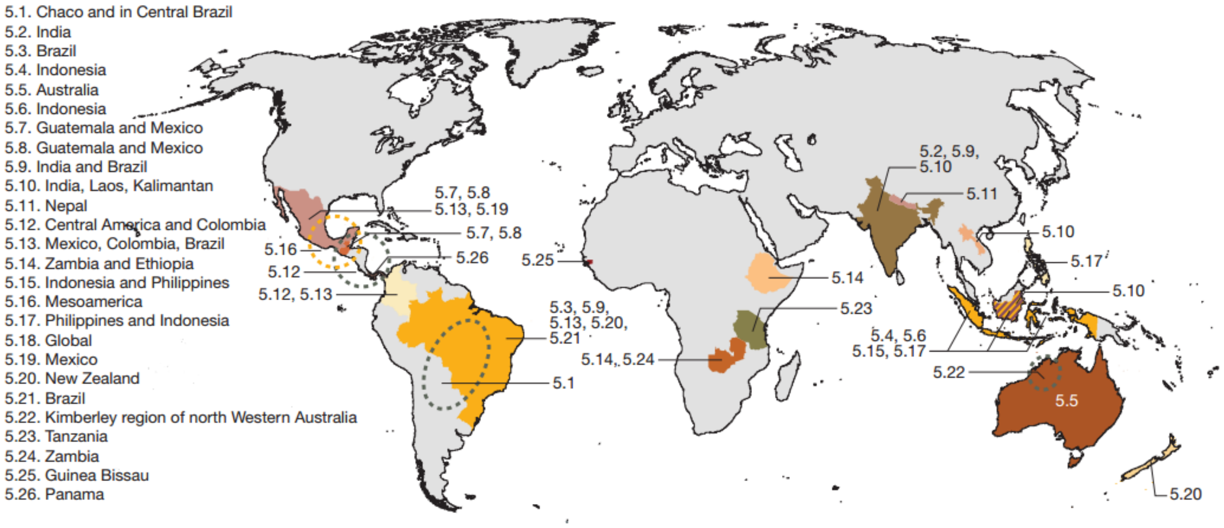
The consideration of indigenous and local knowledge in this assessment is guided by Díaz et al.'s 2015 definition of ILK to be a cumulative body of knowledge, practice and belief, evolving by adaptive processes and handed down through generations by cultural transmission, about the relationship of living beings (including humans) with one another and with their environment. It is also referred to indigenous, local or traditional knowledge, traditional ecological/environmental knowledge (TEK), farmers' or fishers' knowledge, ethnoscience, indigenous science, folk science, and many other titles.

We also recognize that ILKS are *dynamic* bodies *of social-ecological* knowledge, involving *creative* as well as adaptive processes, *grounded in territory*, and *cultural* as well as intergenerational transmission. ILK is often an assemblage of different types of knowledge (written, oral, tacit, practical, and scientific) that is empirically tested, applied and validated by local communities. Hybrid forms of knowledge, negotiated among science, practice, technical, and ILK systems, and variously termed usable knowledge, working knowledge, actionable knowledge, situated knowledge and multiple evidence base are frequently applied pragmatically to the challenges of biodiversity loss (Barber et al., 2014, Tengö et al., 2014, Robinson et al., 2015).

ILKS are found in remote and developing world contexts and also continue within highly industrialised settings. Examples include the "satoyama-satoumi" systems in Japan and Asia (Duraiappah et al., 2012);

many transhumance (the seasonal movement of people with their livestock between fixed summer and winter pastures), agricultural, forestry and fisheries systems across industrialised Europe (Hernandez-Morcillo et al., 2014); and reindeer herders in the Arctic (Riseth, 2007).

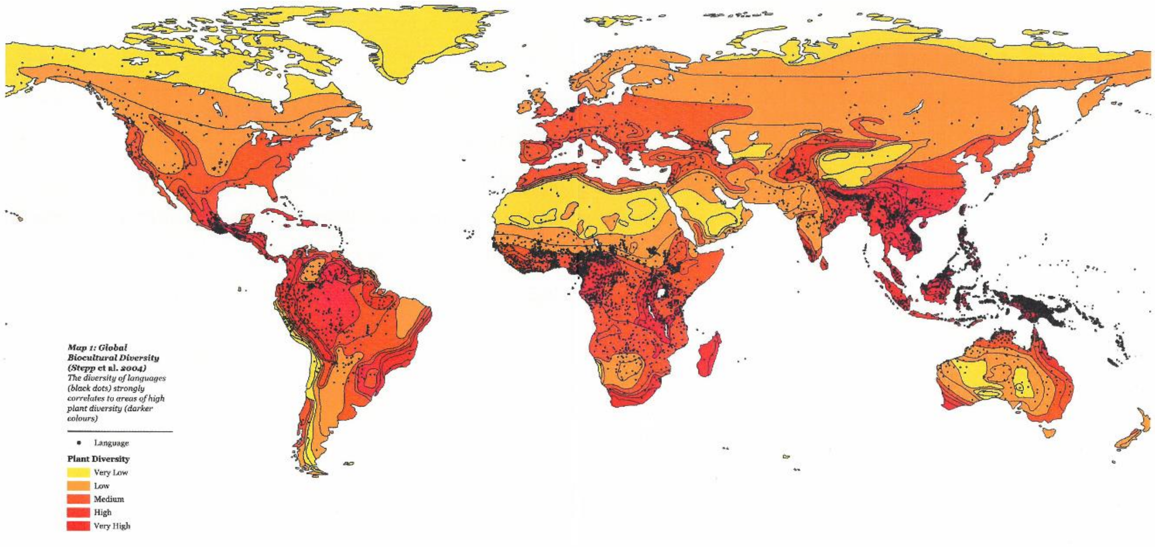
Pollination processes in ILK systems are often understood, celebrated and managed holistically in terms of maintaining values through fostering fertility, fecundity, spirituality and diversity of farms, gardens, and other habitats (Lyver et al., 2015). In this chapter we present case examples from around the world to illustrate aspects of these holistic understandings and their influence on pollinators and pollination (Figure 5-3). We highlight "Co-produced case examples" where direct interaction with ILK-holders has occurred with their *in-situ* knowledge systems.



39. **Figure 5-3 Location of case examples and other features referred to in Chapter 5**

5.1.3 Indigenous and local knowledge systems and biocultural diversity

For the purposes of this assessment, biological and cultural diversity and the links between them are referred to as "biocultural diversity"**.** The term biocultural diversity explicitly considers the idea that culture and nature can be mutually constituting, and denotes three concepts: first, diversity of life includes human cultures and languages; second, links exist between biodiversity and cultural diversity; and third, these links have developed over time through mutual adaptation and possibly co-evolution (Díaz et al., 2015a). Toledo (2001, 2013) encapsulated these ideas into the biocultural axiom: recognition that biological and cultural diversity are mutually dependent and geographically coterminous. Globally, co-occurrence between linguistic and biological diversity is high; for example, mapping places on gradients of plant species diversity and linguistic diversity provides an interesting visual representation of an aspect of these inter-relationships (Loh and Harmon, 2005, 2014) (Figure 5-4.). The relationships between language and biodiversity are of course much more complex than presented in this map — and include for example hybrid cultural landscapes and knowledge systems, and processes of innovation and adaptation as discussed above (Brosius and Hitchner, 2010). Nevertheless, 70% of the world's 6,900 languages occur in the 35 remaining biodiversity hotspots and five high biodiversity wilderness areas globally, suggesting that cultural practices of the speakers of particular indigenous languages tend to be compatible with high biodiversity (Gorenflo et al., 2012). Local communities also play key roles in shaping and maintaining agrobiodiversity, including through fine-scale geographical variations in management related to cultural identity, seed exchange, use of locally-adapted landraces, women's networks to exchange cultivars for specific culinary practices, and adherence to traditional foods for daily consumption (Padmanabhan, 2011; Velásquez-Milla et al., 2011; Botelho et al., 2012; Calvet-Mir et al., 2012; Skarbo, 2015).



40. Figure 5-4. Linguistic diversity and plant diversity map. Source: Loh and Harmon (2014).

Worldwide, local and indigenous cultures have developed unique biocultural associations with pollinators through multiple management, social and farming practices and in the process developed an intrinsic knowledge of their biology and ecology (Quezada-Euán et al., 2001, Stearman et al., 2008). People and communities of interest in industrialized urban settings also interact with pollinators, for example through keeping bees, and running community gardens (Ratnieks and Alton, 2013). Pollinators have become part of biocultural diversity around the world, even in human-dominated contexts such as cities. Claude Lévi-Strauss' (Lévi-Strauss, 1966) analysis of South American mythology of pollinators describes biocultural associations with the diversity of ecosystems. Minute attention to species diversity and habits makes them, as Lévi-Strauss (Lévi-Strauss, 1962) famously put it, not only food for eating but also food for thought (Case example 5.1).

Case example 5.1. Biocultural connections "From Honey to Ashes".

Location: South America

Indigenous people of the South American lowlands (Lévi-Strauss, 1966)

The second volume of Lévi-Strauss´ Mythologiques, titled "Du miel aux cendres" ("From Honey to Ashes") (1966) analyses several dozen myths where honey or bees are present. These myths cover a very large and diverse range of South American lowland indigenous biocultural areas, among them the Chaco, Central Brazil Gê-speaking people, Amazonian tupi-speakers and Arawak-speakers in the Guyana shield. Lévi-Strauss' analysis shows how transformations of these myths, as they travel from one region to another, use an intimate knowledge of biological, climatic and ecosystem specificities. For example, a set of myths, many versions of which were recorded in the Chaco and in Central Brazil, tells the story of a young woman who craved for honey and espoused woodpecker (Family Picidae) master of honey. This position attributed to the woodpecker in several Gê-speaking societies is based on the observation of the extraordinary techniques and stratagems this bird uses for capturing bees' larvae.

5.1.4 Diversity of methods for eliciting values

Values are influenced by the worldviews in which they are grounded, shaped by the social exchanges of everyday life, the power relations, histories and geopolitical interactions of the time (Brondizio et al., 2010). The term value is defined by Díaz et al., (2015) to be "those actions, processes, entities or objects that are worthy or important (sometimes values may also refer to moral principles)" (pg. 13). This definition recognises at least two meanings of value that are important for IPBES assessments — the importance, worth or usefulness of actions, processes, entities or objects, and human-held values, principles or moral duties (Díaz et al., 2015). Societies, groups and even individuals determine what is detrimental, beneficial or value neutral, according to their diverse contexts and perspectives. Values are culturally constructed and contextualized, reflecting diverse and dynamic knowledge systems, and lead to differences in behaviours, interactions and institutions (Brondizio et al., 2010; Descola, 2014).

The IPBES conceptual framework recognises the distinction between intrinsic values, i.e., inherent to nature, independent from any human considerations of its worth, importance, or benefits to people; and anthropocentric, including instrumental and relational values, associated with provision of benefits to people for a good quality of life through both uses and relationships. Intrinsic values of nature acknowledge people as part of the web of life with a relatively recent role in the evolutionary history of life on Earth (Sandler, 2012; Hunter et al., 2014). This separation does not hold in world views of most Indigenous peoples and local communities, who do not recognise a nature-people dichotomy, viewing spiritual presences of people as present in the world from time immemorial.

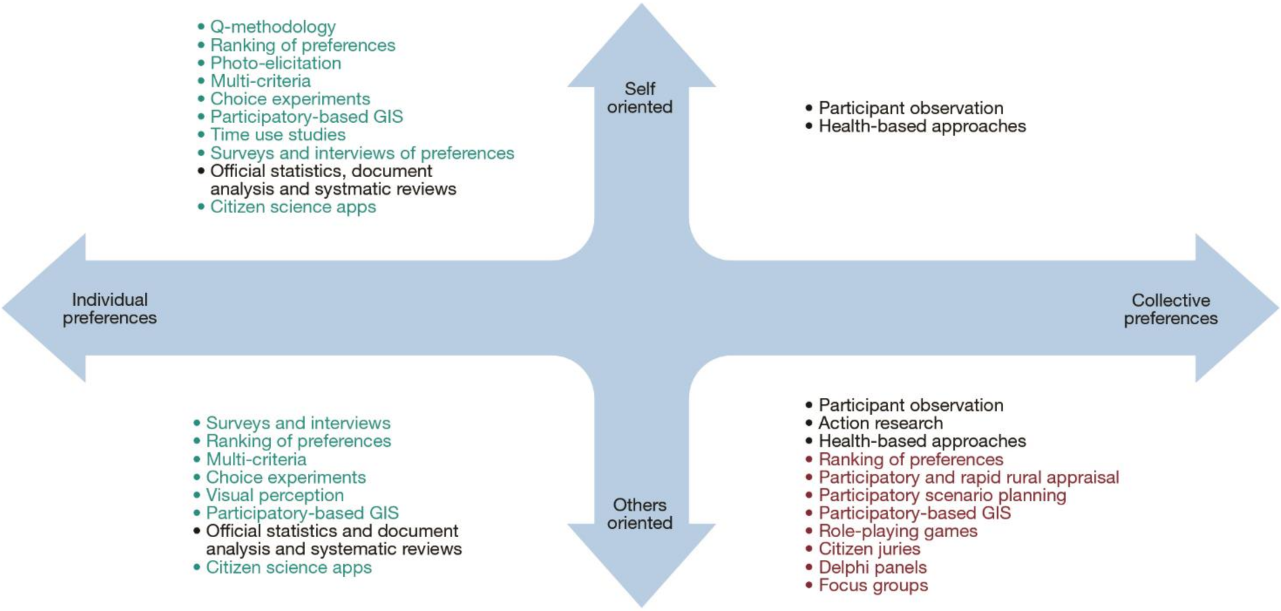
Diverse valuation methods in the biophysical, economic, socio-cultural, health and holistic domains can elicit and characterise intrinsic, instrumental and relational values through both quantitative and qualitative measures (Martin-López et al., 2014; Raymond et al., 2014; IPBES, 2015; Pascual and Balvanera, 2015).

Here we address both socio-cultural and holistic valuation, first of aspects of nature's benefits to people, and then of good quality of life, dependent on pollination and pollinators (Tengberg et al., 2012). While a health valuation is beyond the scope of the chapter, we do pay attention to aspects of nutritional health. We conclude this introduction with a brief summary of how socio-cultural and holistic valuations are undertaken, in recognition that valuation methods shape and articulate values, operating as informal institutions that influence diverse behaviours and perceptions (Gómez-Baggethun et al., 2014; Martin-López et al., 2014; Vatn, 2005). We therefore refer to valuation methods as value-articulating institutions.

5.1.5 Sociocultural and holistic valuation

Because of the multiple concepts and dimensions of nature's values, any socio-cultural or holistic valuation of biodiversity and ecosystem services is relative to a given individual or group of people, in both industrialised and indigenous contexts (Turner et al., 2003). A first critical step for valuation of pollination is actors' identification, through questions such as: whose quality of life and usage of nature's benefits to people depends directly on pollinators and pollination? For whom are pollinators and pollination indirectly important? Who would be negatively affected if pollination would decline? Whose practices are influencing pollinators' populations? What is happening to the environment, landscape, agroecosystem, pollinators and pollination processes as a result? (Reed et al., 2009; IPBES, 2015).

Socio-cultural valuation approaches to find answers to these questions can be viewed as varying across two dimensions: self-oriented to other-oriented (Chan et al., 2012b) and individual to collective (Figure 5-5.). Ethnographical methods such as secondary and documentary data analysis, participant observation and interviews (e.g., formal, semi-structured) are widely used in socio-cultural valuation, with particularly relevance to collective preferences (IPBES, 2015; Scholte et al., 2015). Individual preferences methods require the individual to articulate his/her values according to a consistent logic and specific rationality and reflect pre-analytic conceptions. Individual preferences can be assessed through surveys and interviews, rankings of preferences, multi-criteria analyses, Q-methodology, photo-based or valuation through visual perception elicitation time-use studies, documentary analysis and citizen science tools such as mobile applications (Christie et al., 2012; Brooks et al., 2014; IPBES, 2015). Most of these methods can be used to elicit both self-oriented (for personal well-being) and other-oriented (for societal well-being) values.



41. Figure 5-5. Synthesis of socio-cultural valuation methods.
(Based on Chan et al., 2012a and b, Christie et al., 2012, and Kelemen et al., 2014). Methods in blue are the consultative ones; methods in red are deliberative; and in black are other types of methods.

Valuation by deliberative methods elicits values through social processes, based on communication and collective debate (Raymond et al., 2014). Deliberative methods often aim to assess values while achieving consensus through a process of reasoned discourse, but can also highlight distinct value-choices and trade-offs, such as through participatory scenario planning (Habermas, 1987; Carpenter et al., 2006). Deliberative methods can involve substantial transaction costs and be challenged by power and knowledge asymmetries (Hill et al., 2015a). Deliberative methods include citizen juries, forums, workshops, focus groups, participatory scenario planning, participatory GIS, collective preference ranking, participatory and rapid rural appraisal, role-playing games and Delphi panels (Chambers, 1981, 1994; Susskind et al., 1999; Pert et al., 2013). Valuation methods involve a combination of quantitative, qualitative and mixed methods approaches to data collection and analysis (Creswell, 2014; Kelemen et al., 2014).

Socio-cultural valuation can capture potential impacts such as loss of psychological benefits from viewing pollinators such as butterflies and bees (Kumar and Kumar, 2008; Hanley et al., 2013). Socio-cultural evaluation helps identify how and why different values are relevant for different people; within different times (e.g., seasons) and places; to recognize perceived trends as an early warning of ecosystems deterioration; to reveal intangible values; to explore how these values relate one with the other (e.g., in bundles) and to quality of life; to reveal trade-off options; to integrate different forms of knowledge and to detect power asymmetries and potential social conflicts related to different perceptions, needs and use (Chan et al., 2012a; Plieninger et al., 2013; Martin-López et al., 2014; Oteros-Rozas et al., 2014; Scholte et al., 2015).

Holistic valuation methods are closely aligned to socio-cultural valuation approaches, and use many of the same deliberative other techniques (IPBES, 2015). The central feature that distinguishes holistic approaches is their internalization of the world views of indigenous peoples and local communities (Quaas et al., 2015). The IPBES Conceptual Framework provides that pairing different value systems with different valuation approaches and techniques is important to providing integrated understandings of nature’s benefits to people, and contributions to good quality of life (Díaz et al., 2015a). The diversity of Indigenous peoples’ and local communities’ (IPLC) values systems challenges an easy pairing between valuation approaches and value systems (IPBES, 2015). Nevertheless, two features among ILK systems are commonly encountered as introducing complexity into conventional socio-cultural valuation approaches.

The first feature in ILK systems is the emphasis on the interconnectedness and multiple relationships between people and nature, reflected in concepts such as totems, kin groups, sacred sites, ancestral landscapes, numina and taboo relationships (Berkes, 2012; IPBES, 2015). Cultural values are seen to vary spatially and temporally with the dynamics of these social relations — for example, Aboriginal people in central Australia attribute the wave of mammal extinction to the decline of their ceremonies for those animals (Rose, 1995; IPBES, 2015; Jackson and Palmer, 2015; Pert et al., 2015). Socio-cultural valuations approaches more frequently consider how the diverse social groups assign different values to various parts of the landscape, resulting in values varying spatially with the dynamism of the environmental attributes, and the concept of cultural ecosystem service hotspots (Raymond et al., 2009; Martínez Pastur et al., 2015).

The second feature is the ongoing stories and life-ways through which relationships are forever alive and dynamic, continuously weaving together and co-creating the world (Ingold, 2011; Jackson and Palmer, 2015). Socio-cultural valuation methods typically are based on concepts of a place, such as a wetland, being perceived and hence valued in different ways by multiple stakeholders, rather than being co-created manifestations (Martin-López et al., 2014).

Holistic valuation methods are oriented to indigenous peoples’ and local communities’ own logics; particular examples include the Māori Wetland Indicators (Harmsworth et al., 2011) and the Salish environmental health indicators (Harmsworth et al., 2011). Jackson and Palmer (2015) argue that valuing practices and ethics enables the “possibility of understanding ecosystem services in ways which make legible and enhance the possibility of recognizing, building and expanding upon the reality of indigenous social tenures and reciprocal social relations” (pg. 18). Holistic valuation approaches are used here to give the special emphasis on the experience of indigenous and local communities required by the chapter scope, through a focus on relevant practices based on ILK.

5.2 Pollinators, pollination and nature’s benefits to people

5.2.1 Natures's benefits to people, good quality of life and categories of values

While typologies of values are always somewhat artificial — values can be categorized in many different ways in response to dynamic human cultures, and social-ecological interactions — they are useful to valuation (MEA, 2005; Tengberg et al., 2012). From the socio-cultural valuation perspective, pollination and pollination-dependent products contribute to the delivery of provisioning services, such as food, medicine, construction materials and items of technology (e.g., musical instruments); and provide cultural services such as recreational and educational activities with and for pollinators (gardening, ornamentals, learning from beekeeping), and as a source of inspiration, including through the use of natural motives of artefacts in art, folklore, sacred, religious, technological and other forms of inspiration (**Error! Reference source not found.**).

From the holistic valuation perspective, nature's benefits to people fit key categories of nature's gifts to indigenous peoples and local communities in the form of practices of supporting diversity and fostering biocultural diversity, in landscape management practices, diversified farming systems, innovation and adaptation. While many practices and ethics outside of indigenous peoples and local communities could also be considered as nature's gifts, the scope of this assessment did not extend to investigating this dimension.

25. Table 5-1. Nature's benefits to people and categories of value in this assessment.

Category	Type of values	Focus of values	Categories used in this assessment
Nature's benefits to people	Instrumental	Ecosystem goods and services (socio-cultural valuation)	Provisioning services: Food, medicine, construction materials, technology (e.g., musical instruments)
			Cultural services: Recreational and education (activities with and for pollinators); inspirational (use of natural motives or artefacts in art, folklore, sacred, religious, technological and other forms of inspiration)
		Nature's gifts (holistic valuation)	Practices gifted to indigenous peoples and local communities: the practices of valuing diversity and fostering biocultural diversity; landscape management practices; diverse farming systems; innovation

The categories considered for good quality of life include a range of values that overlap to some extent with those that comprise nature's benefits to people (Table 5-2). For example, quality of life categories include the livelihoods of indigenous peoples and local communities that derive from relationships between ILK-holders, pollinators and pollinator-dependent products, including income, food and medicines. While these can also be viewed as aspects of provisioning services, and part of nature's benefits to people, from the perspective of ILK systems, they fit better with concepts of good quality of life (Díaz et al., 2015). Pollinators support numerous other categories of value that contribute to good quality of life including heritage, aesthetics, identity, social relations and governance attributes. These relational values are assessed in section 5.3.

26. Table 5.2 Good quality of life and categories of value in this assessment.

Category	Type of values	Focus of values	Categories used in this assessment
Good quality of life	Relational	Heritage (socio-cultural valuation)	Both tangible and intangible relationships between people, pollinators and good quality of life
		Aesthetics (socio-cultural valuation)	Appreciation of natural and cultivated landscapes and species
		Identity (socio-cultural valuation)	Group and individual identity linkages with pollinators
		Livelihoods (holistic valuation)	Derived from relationships between ILK-holders, pollinators and pollinator-dependent products
		Social Relations (holistic valuation)	song, dance, art, story, rituals and sacred knowledge associated with pollinators and pollination
		Governance (holistic valuation)	Governance by, with and for pollinators

5.2.2 Provisioning ecosystem services (socio-cultural valuation)

Provisioning services include the pollination of plants, and the use of pollinators themselves, for food and medicine production, pollinators' products such as honey and wax used in objects (e.g. fine musical instruments), and pollinator-dependent construction materials, biofuels and fibre (Krell, 1996; Quezada-Euán et al., 2001).

Many foods and medicines are derived from pollinators and pollinator-dependent resources (Costa-Neto, 2005; Cortes et al., 2011; Eilers et al., 2011; Rastogi, 2011). Around 2,000 insect species are consumed as food globally, including many that are pollinators such as the larvae of beetles, moths, bees, and palm weevils, in both developing and developed world contexts (Jongema, 2015). Insects are now being recognised as potentially important for food security, being high in protein, vitamins and minerals (Rumpold and Schluter, 2013; van Huis, 2013). In Fiji, trees providing fruits for human consumption include coconut (*Cocos nucifera*) and lilly-pilly (*Syzygium* spp.), both pollinated by bats (*Notopteris macdonaldi*, *Pteropus samoensis*, and *Pteropus tonganus*) (Scanlon et al., 2014). Durian (*Durio zibethinus*), a popular and economically high-return fruit throughout southeast Asia, with rich bioactive and nutraceutical properties, relies primarily on pollination by bats (e.g. *Eonycteris spelaea*) (Bumrungsri et al., 2009; Ho and Bhat, 2015)Figure 5-6. Crop plants that depend fully or partially on animal pollinators are important sources of vitamin C, lycopene, the antioxidants beta-cryptoxanthin and beta-tocopherol, vitamin A and related carotenoids, calcium and fluoride, and a large portion of folic acid available worldwide (Eilers et al., 2011).

42. Figure 5-6. Flowers of durian, a high-value tropical fruit, and their bat pollinator (Synconycteris australis) in north Queensland, Australia.
© Barbara & Allen at Wild Wings & Swampy Things Nature Refuge. Reproduced with permission.

Bees and their products (venom, honey and wax) have been used since Ancient Greek and Roman times in curing everything from bladder infections to toothaches and wound recovery (Weiss, 1947; Krell, 1996). Scientific and technological development of bee products such as propolis (the resin collected by honey bees from tree buds, used by them as glue) and honey continue to yield medicinal and pharmacological products and uses, including as anti-diabetic agents (Banskota et al., 2001; Amudha and Sunil, 2013; Begum et al., 2015; Jull et al., 2015). Honey is anti-bacterial, anti-viral and anti-fungal, and all of these properties make it ideal for healing wounds (Kumar et al., 2010). Bee products, primarily honey, are currently used to treat, among other illnesses, multiple sclerosis, osteoarthritis, rheumatoid arthritis, post-herpetic neuralgia, coughs, herpes simplex virus, premenstrual syndrome, sulcoplasty, allergic rhinitis, hyperlipidemia, the common cold, and topically for burns, wound healing, diabetic foot ulcers and for improving athletic performance (Gupta and Stangaciu, 2014). Stingless bees' honey is widely used for medicinal purposes by indigenous peoples and local communities, in regions where they are distributed, as integral parts of their livelihood systems, described in section 0 (Massaro et al., 2011).

Several musical instruments depend on the provisioning services of pollinators. Propolis is an important ingredient of the varnish used on high-quality stringed instruments (Lieberman et al., 2002; Stearman et al., 2008). Bees' wax is an essential ingredient in Asian mouth organs, which originated in what is now Laos more than 3,000 year ago, and have diversified into different forms in China (*sheng*) and Japan (*shô)* (Peebles et al., 2014). Historically, ethnic groups in many countries have a great variety of musical instruments from gourds, which are fruits of pollination. The wax of native bees play a very important role in pre-Colombian Amerindian cultures, (Patiño, 2005) and especially in metallurgic activities, through a technique to produce pieces of metalwork. The Amerindian silversmiths produced gold pieces with the method known as "drain to the lost wax". The cerumen was used to produce a mould of a model of the piece they want, and after several processes, the cerumen was replaced by gold to obtain the finely-crafted object which faithfully reproduces every detail on the surface of the original model (Falchetti, 1999). Lost-wax casting using bees' wax dates back to copper objects found in Israel between 3500-3000 BC (Crane, 1999) (Figure 5-7.). In western Colombia, the propolis of "brea bees" (*Ptilotrigona occidentalis*) called *canturron* was used on torches for lighting and for waterproofing boats and as healing of minor wounds (Galvis, 1987; Nates-Parra, 2005; Patiño, 2005). Cerumen and wax are also critical ingredients in traditional bows and arrows, and contemporary tourist versions of these in the Bolivian Amazon (Stearman et al., 2008). Beeswax has long been an ingredient of surfboard wax, and is resurging in response to interest in eco-friendly products (Falchetti, 1999; Chioi and Gray, 2011).

43**. Figure 5-7. Drain to the lost wax: Gold pieces produced (Pre-Columbian) by Amerindian cultures with this technique using the wax of stingless bees.**
© Banco de la Republica de Colombia. Reproduced with permission.

A: Wax Molds; B: Quimbaya Poporo (Pre-Columbian) C: Muisca Raft ceremonial.

A

B.

C.

Pollination is also critical for ensuring availability of other useful materials such as biofuels (e.g., *Jatropha curcas)*, fibre (e.g., cotton) and construction materials ls (e.g., *Eucalyptus* spp.). The biofuel crop Jatropha oil (*Jatropha curcas)* has highest overall yield and quality under natural pollination by bees (Romero and Quezada-Euán, 2013; Negussie et al., 2015). Maintaining communities of pollinators enhances production on cotton farms, especially in organic production (Pires et al., 2014). Eucalyptus spp. and other tree species important for construction rely on animal pollination (Pavan et al., 2014).

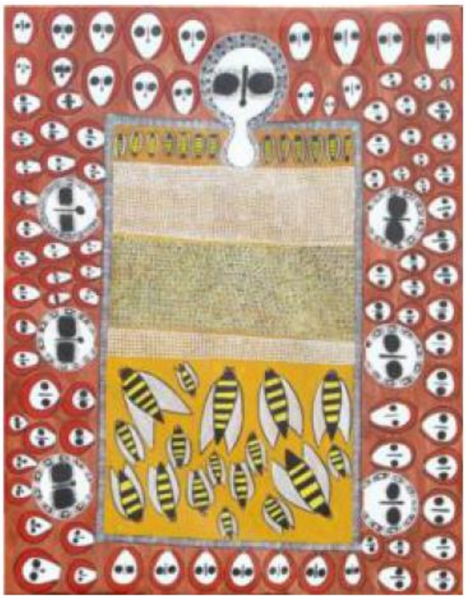
5.2.3 Cultural ecosystem services: sources of inspiration (socio-cultural valuation)

Pollinators, particular bees, have long been a source of inspiration for art, literature, folklore and religion (de Gubernatis, 1872; Andrews, 1998; Kristsy and Cherry, 2000; Bastian and Mitchell, 2004; Werness, 2006). Rock art of honey bees has been identified at 380 separate sites in 17 countries across Europe, Africa and the Indian sub-continent, showing 25 representations of honey harvesting or associated activities (Crane, 2001, 2005) (Figure 5-8.). The earliest records come from rock art in southern Africa dated to 10,000 years ago, with some sites possibly older, and in Europe dated to 8,000 years ago (Crane, 1999; Lewis-Williams, 2000). The wax from honey bees was used to preserve the colors of ancient wall paintings more than 2000 years ago in central Asia and Crimea (Birshtein et al., 1976).

44.Figure 5-8. Rock art of bee-hunting. Mesolithic (c. 10,000/8000–c. 3000 bce).
Cueva de la Arana, Spain. © Museum of Prehistory, Valencia, Spain. Reproduced with permission.

Art associated with ‘sugarbag dreaming’, the term for sacred stories, ceremonies and other practices associated with stingless bees among Aboriginal Australians, is common in both rock-art sites and contemporary bark and other media paintings (Morphy, 1991; Prideaux, 2006) (Figure 5-9. A). Rock art with beeswax, although relatively young in Australian terms, is commonly used for dating in that continent; the oldest beeswax figure known from Australia is a turtle motif dated from 4000-4500 BP, at Gunbilngmurrung, Northern Territory (Langley and Taçon, 2010).

Pollinators, particularly bees, are also inspirations for many sacred and religious traditions, including within Islam, Christianity, Hinduism, Buddhism and traditional Chinese teachings. For example, the coat of arms of Pope Urban VIII, Maffeo Barberini, features three bees as the central symbol, which can be found in various ornamentations including the fresco ceiling of the Barberini Palace (National Museum of Art), painted to celebrate his Papacy, parts of the Vatican building and Saint Peter’s Basilica (Hogue, 2009) (Figure 5-9. B). Moroccan and many other societies’ interactions with bees and honey today are guided by the religious principles set out in the Qur’an, the sacred text of Islam, which includes a passage devoted to bees, the *Surat An-Naĥl* (Adam, 2012) (Figure 5-9. C). Chuang Tzu (Zhuangzi), a defining figure in the religious traditions of Chinese Taoism, writes of the blurred distinction between a man dreaming of being a butterfly, or a butterfly dreaming of being a man, symbolising spiritual transformation of the material (Wu, 1990). In Buddhist text and teaching, bees and pollinators symbolize the enactment of compassionate and conscious living (NAPPC Faith Task Force, 2012). Many of the foundational texts of Hinduism feature pollinators and pollination (Case example 5-2).



45. Figure 5-9. Pollinators in sacred traditional and religious art from three continents.
A: Sandra Mungulu (b.1960), 'Wandjina and Waanungga' acrylic on canvas. Australia. © Sandra Mungulu/Licensed by Viscopy, 2015.

Artist Sandra Mungulu explains, "Waanungga is a word for various forms of bush honey, 'sugarbag', found in trees and termite mounds. The Wandjinas (ancestral beings from the dreaming, present in the landscape today) keep the countryside fresh and healthy which allows the native bees to produce high quality honey. My mother is called 'Guduwolla', the Ngariniyin name of a particular tree which produces white pollen in early summer, and is the main source of sugar bag in the Kimberley region of north-west Australia".

B: Three-bee centrepiece of Pietro da Cortona's Ceiling of the National Gallery of Ancient Art at Palazzo Barterini, Rome. Europe. © Ministero per i Beni e le Attivita Culturali. Reproduced with permission.

The Barberini coat-of arms features the 3-bumblebee crest and appears in the centre of Pietro da Cortona's Ceiling, painted to celebrate Cardinal Meffio Barberini becoming Pope Urban VIII, celebrating divinity. This 3-bee crest appears in the Vatican and St Peter's Basilica.

C: Celebrating pollinators in Islamic Art: Chinese Export Rose Canton porcelain produced for the Persian market, China, Qing Dynasty 1875 AD / 1292 AH.

This porcelain dish, celebrating fruits, leaves, insects, birds, roses, flowers and the nightingale, was commissioned in 1875 AD / 1292 AH for personal use or as a royal gift. Rose Canton porcelains were praised in Iran for their colourful and cheerful composition, bright, meticulous execution and lustrous glitter. The inscribed Persian poem reflects the merry atmosphere with a deeper meaning, contemplating a meditative state, important in Islam. © Islamic Arts Museum Malaysia, 2016. Reproduced with permission.

Case example 5.2. Sacred text on flower morphology, pollinators and pollination from India.

Location: India

Many different communities over millennia

Studies have shown that pollination and pollinators have been an important part of Asian culture and religious traditions for centuries (Joshi et al., 1983). In Asia, India has the most ancient written records of association between humans, pollination and pollinators. Ancient literature (circa 1700-1100 BCE) that comprises the sacred texts of Hinduism — the Vedas (poems and hymns), Upanishads (sacred treatises), the Puranas (sacred writings) — and major Sanskrit epics like Mahabaratha and Bharatayudaall, all contain information on flower morphology, pollinators and pollination (Belavadi, 1993). Several rock paintings in caves in Central India depicting beehives and honey collection show that pollination and pollinators were already an important part of the culture since the Mesolithic era (15000-11000 BCE) (Wakankar and Brooks, 1976).

Bees are famous in literature and poetry, for example from Shakespeare's references about bees and honey in *Julius Caesar*, *King Henry IV, V* and other plays (Miller, 1948), to the prize-winning collection *The Bees* by Poet Laureate Carol Ann Duffy (Duffy, 2011). Bees and honey appear in the literary traditions from the ancient Egyptians, Romans and Greeks, in Sumeria and Babylonia, in Britain and Ireland, France, Finland, in the codices of the Mayans in central America, among the Germanic and Slavonic people of central Europe, in central and southern Asia (Edwardes, 1909; Ransome, 1937 [2004]). Bees and honey are a source of inspiration for both popular (e.g., "Tupelo Honey" by Van Morrison; "King Bee" by Slim Harpo) and traditional classical music (e.g., Flight of the Bumble bee by Rimsky-Korsakov) (Hogue, 2009).

Bees in general are a source of inspiration for technological development, for example in relation to visually guided flight and robotics (Srinivasan, 2011; Sun, 2014). Increased opportunities to observe pill-rolling behaviour by scarab beetles following domestication of large mammals in the Middle East has been identified as a source of inspiration for the invention of the wheel (Scholtz, 2008). Amateur entomology (particularly centered on the pollinators butterflies and beetles) is extremely popular in contemporary Japan and has inspired development of thirty-foot telescopic nets, and bug-collecting video games (Kawahara, 2007).

Native bees are the source of inspiration for contemporary art and wildlife photography, as evidenced by enormous popularity of the USGS Native Bee Inventory and Monitoring Web-site showing high-resolution and close-up photos (Droege, 2010). Canadian artist Aganetha Dyck[11] co-creates delicate sculptures with bees by leaving porcelain figurines, shoes, sports equipment, and other objects in specially designed apiaries where they are slowly transformed with the bees' wax honeycomb (Keshavjee, 2011); she won the Canadian Governor General's Award in Visual and Media Arts in 2007. The Pollinator Pathway® is another award-winning example, developed from participatory art, design, ecology and social sculpture by artist Sarah Bergman to promote ecological corridors for pollinators in urban spaces (Bain et al., 2012). Bergman (2012) now offers certification for others creating such pathways. Bees are a source of inspiration for public and community art. In London, UK, for example, street artists promote the conservation of bees through murals and graffiti; and the annual community mandela project in British Columbia celebrated bees in 2013 (Figure 5-10.).

46. Figure 5-10. Public art inspired by bees.
A: Save the bees project in London, United Kingdom. © Louis Masai Michel. Reproduced with permission.

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[11] http://www.aganethadyck.ca/

B: Mandela with bees in British Columbia, Canada. © Roberts Creek Community Mandela. Reproduced with permission.

5.2.4 Cultural ecosystem services: recreational and educational values of beekeeping (socio-cultural valuation)

Honey bees and beekeeping are highly valued as recreational activities (Gupta et al., 2014). Tierney (2012) found that rural beekeeping was an effective intervention tool for reducing recidivism (i.e., relapse in criminal behaviour) among youth, increasing their self-esteem, confidence, the ability to learn and the frequency of social interactions. In Greater London, the number of beekeepers tripled from 464 to 1,237, and the number of hives doubled from 1,677 to more than 3,500 between 2008 and 2013, leading to concerns that there were insufficient floral resources to keep bees healthy (Ratnieks and Alton, 2013). In Germany, the number of beekeepers has increased by 53% since 2012, and bee-keeping has emerged as a popular ecologically-inspired urban lifestyle phenomenon, alongside growing markets for locally-produced honey (Lorenz and Stark, 2015).

In Sargodha and Chakwal districts of Pakistan, beekeeping activities teach and educate the communities about the values of cooperation in life (Qaiser et al., 2013). Beekeeping activities pass on knowledge about pollination for the youth and rural people in India (Sharma et al., 2012). The Bee Hunt! Program in the

USA involves students across the nation in photographing bees, uploading spatially-located observations and photos to a data-sharing Internet site, enabling understanding of bee distribution relative to drivers such as pesticides, and provides resources to empower them to take action to solve bee problems through technology, education and policy advocacy (Mueller and Pickering, 2010). Beekeeping can also lead to new knowledge. For example, one Spanish beekeeper has found that a moth species, *Galleria mellonella*, regarded as plague for bees, is actually an ally that cleans spores and microorganisms from the hives (Santoja, 2005).

5.2.5 Nature's gift: practices of ILK-holders and their extent of influence (holistic valuation)

Global data on the extent of the Earth's surface under ownership, management and use by indigenous peoples and local communities, are not yet available, a key knowledge gap that needs to be addressed for ongoing biodiversity and ecosystem service assessment. Available data suggest ILK systems provide the foundation for ongoing conservation, management and use of ecosystems over large parts of the planet (Chhatre and Agrawal, 2009; Gómez-Baggethun and Reyes-García, 2013; Kelemen et al., 2013). For example, the area of forests owned by, or designated for, indigenous peoples and local communities in Lower and Middle Income Countries (LMIC) has increased from 21% in 2002 to 30% in 2013 as rights-recognition has strengthened in some countries. (White and Martin, 2002; Rights and Resources Initiative, 2014). Kothari et al. (2012) estimate that Indigenous and Community Conserved Areas[12] may cover as much as 13% of the Earth's terrestrial surface. Indigenous peoples number around 370 million, and live in all regions of the world (Secretariat of the United Nations Permanent Forum on Indigenous Issues, 2014). Nevertheless, many communities are losing land they have occupied for centuries or millennia because of limited recognition of their rights (van Vliet et al., 2012; Rights and Resources Initiative, 2014; Césard and Heri, 2015; Perez, 2015; Samorai Lengoisa, 2015).

Among local communities, part of the 55% of global population who are rural, many are farmers (IFAD, 2011). Small holding farmers in local communities hold knowledge adapted to understanding and managing local ecologies and land capabilities, including of soil fauna and properties, tree dynamics and genetic diversity, landscape-scale vegetation patches, crop diversity, livestock resources and agroforestry species (Netting, 1993; von Glasenapp and Thornton, 2011; Gao et al., 2012; Pauli et al., 2012; FAO, 2014a; Segnon et al., 2015; Valencia et al., 2015). Small holdings (less than 2 ha) constitute 8-16% of global farm land, 83% of the farms and 83% of the global population involved in agriculture (IFAD, 2013; Lowder et al., 2014; Steward et al., 2014).

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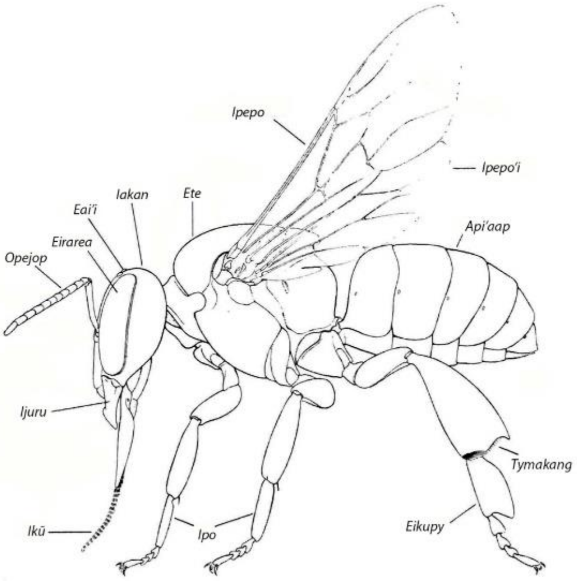
[12] Indigenous and Community Conserved Areas (ICCAs) have been defined by IUCN as 'natural and/or modified ecosystems, containing significant biodiversity values, ecological benefits and cultural values, voluntarily conserved by indigenous peoples and local communities, through customary laws or other effective means' (Kothari et al., 2012).

5.2.6 Practices for valuing diversity and fostering biocultural diversity of stingless bees and pollination resources in central and South America

Many indigenous peoples are known to value diversity in itself, to appreciate the existence of many different living and non-living entities as important (Tsing, 2005; Rival and McKey, 2008). This translates into recognizing and naming very fine distinctions in domains such as landscapes, wild species and cultivated varieties. Observations of these distinctions enable Indigenous peoples and local communities to collect, experiment and select varieties and species. Indigenous peoples in central and south America domesticated many pollinator-dependent crops that are now cultivated globally, including legumes (common bean, lima beans, peanut), cucurbits (chayote, pumpkins, squash), solanaceous fruits (capsicum peppers, husk tomato, pepino, tomato), fruits and nuts (blueberry, brambles, cactus pear, cashew, papaya, pineapple, strawberry), beverage crops (cacao, mate), ornamentals (dahlia, fuchsia, sunflower), industrial crops (cotton, rubber, tobacco), tubers (cassava, potato, sweet potato) and pineapples whose seed production requires pollination (Janick, 2013). This valuable diversity translates into a wide array of connections (relational values) with a wider array of pollinators and their products, including honey, pollen, resins, and oils. For example, the Wayapi people of Guyana and Brazil recognise 17 different varieties of honey that each come from a different stingless bee species, each with a specific name (Grenand, 1972).

Latin Amerindian knowledge of stingless bees is particularly strong. In Colombia, Nates-Parra and Rosso-Londoño (2013) recorded nearly 50 common names used for the stingless bees, with wide variation among regions and informants. Common names do not always correspond one-to-one with scientific names, and such locally recognized entities are termed ethnospecies, which can match, under-differentiate or over-differentiate compared to scientific species (Otieno et al., 2015). Detailed knowledge exists of at least 23 ethnospecies among the Hoti people in Venzuela; 25 bee ethnospecies among the Tatuyo, Syriano and Bara peoples of Colombia and the Guarani-Mbyá people of Argentina, Brazil, Paraguay; of around 43 different bee ethnospecies among Nukak people of northwest Amazon; of 48 bee ethnospecies among the Enawene-Nawe people of southern Brazil and 56 bee ethnospecies among the Gorotire-Kayapo in northeastern Brazil (Posey, 1983a; Cabrera and Nates-Parra, 1999; Rodrigues, 2005; Rosso-Londoño and Parra, 2008; Santos and Antonini, 2008; Estrada, 2012; Rosso-Londoño, 2013). Kaxinawa and Gorotire-Kayapo, as well as many other indigenous peoples, understand nest architecture in detail, naming external and internal parts, as well as the various parts of the bee, a remarkable feat without microscopes, reflecting the strategy of close observation that is so critical to their fostering of pollination and pollinator diversity (Posey, 1986; Camargo and Posey, 1990; Oliveira, 2001). Kawaiwete peoples' close observation extend to fine detail of pollinator-relevant structures, such as the pollen basket (Figure 5-11.) (Villas-Bôas, 2015).

47. Figure 5-11. Morphological structure of bees as recognised by the Kawaiwete close observation techniques that underpin pollinator management.
Source: Villas-Bôas (2015) (adapted from Camargo and Posey, 1990).



Kayapo have specific names for each larval and pupal instar of the stingless bee, and the colony sociality and organization of labor helped to build their imagery, inspiring their social life in the tribe. In addition, the Gorotire-Kayapos developed an ability to locate bees nest by listening to the noise from nest ventilation, which they recognized for each bee species. At night, shamans walk in the forest to locate bee nests. Other Amerindians used to follow the odor that bees used to mark nesting sites. Insects, especially bees, ants and wasps, are of great practical and symbolic importance for the Andoke people (Colombian Amazon forest). They are able to classify bees according to the quality of their honey, the food and nesting habits (Jara, 1996). Aggressive bees like *Oxytrigona* spp. and *Apis mellifera* (African bee invaders) were managed with smoke and a liana which had an effect of calming the bees to sleep, so that people could collect the honey without being harmed (Camargo and Posey, 1990).

Diversity in bees is celebrated in many stories (e.g., Case example 5.3). An Ofaié-Xavante myth talks of a time when animals and people were not distinct and honey came from a single cultivated plant. But the master of animals found it more reliable to confine production of honey to a great diversity of stingless bees, whom he unleashed into the forest. This myth interestingly praises collection in the wild over agriculture – diversity of honey in the wild is preferable to domestication (Lévi-Strauss, 1966). Indigenous lowland people in South America continue to favour their many different types of wild honey. Honey-hunting expeditions, targeting different honey at different seasons, are highly valued and most frequent in the dry season. Honey is considered exquisite food, and while it can be eaten naturally in the forest, it is mainly drunk mixed with water and bees´ larvae. Many Amazonian societies will ferment the beverage and make it into a beer; they will also mix it with several palm fruits´ juice and let it ferment (Villas-Bôas, 2015).

Case example 5.3. Kawaiwete peoples' knowledge and use of numerous stingless bees.

Co-produced case example

Underpinned by direct interactions with indigenous and local knowledge-holders

Location: Southern Amazon, Brazil

Kawaiwete Indigenous people (Villas-Bôas, 2015)

Kawaiwete Indians (previously known as Kaiabi) now live in the Xingu Indian Park, in Southern Amazon. Stingless bees are under the protection of a strong entity who may well punish and inflict "bee illness" onto those who do not show proper respect and observe silence when collecting honey. Hence, as honey may carry some degree of risk, its medicinal use is not as wide as elsewhere. However, it is used for diarrhea caused by undercooked fish. Bee hives containing eggs and larvae, rather than honey itself, are used to calm fever and for rubbing children´s and young peoples´ heads in order to protect them from illness as well as for expelling harmful spirits. Expecting fathers are required to observe several rules related to bees in order to benefit both delivery and the baby´s health.

Kawaiwete have extensive knowledge of and names for 37 stingless bee species, their particular habitats, and their ecological distribution, and they identify 28 forest trees that bees use for nesting as well as 19 plant species on which they like to feed. Kawaiwete consider as edible the honey of 26 out of those 37 bee species. *Eiry,* also rendered as "honey juice", is much appreciated. It is prepared from honey occasionally mixed with bee larvae. Honey found in the forest will also be a man´s sustenance during hunting expeditions. Round pointed arrow tips are made with bees wax and serve to capture ornamental feathered birds. Wax is also extensively used for repairing calabashes. Kawaiwete are aware of the geographic distribution of different bees' species and they sorely regret no longer having access to species endemic to their former territory.

By their practices of favoring heterogeneity in land-use as well as in their gardens, by tending to the conservation of nesting trees and flowering resources, by distinguishing the presence of a great range of wild bees, and observing their habitat and food preferences, indigenous peoples and local communities are contributing to maintaining, fostering and co-creating an abundance and, even more importantly, a wide diversity of bee pollinators and pollination-dependent biota.

These practices extend to other pollinators. For example Ribeirinhos people from Brazil note a specific pollination connection that exists between a cockchafer and the plants *Theobroma* spp. (Couly, 2009); Bribri and Cambécar peoples in the Talamanca of Costa Rica have extensive knowledge of birds who are pollinators, with local names and narratives about their behaviours (Fernández et al., 2005); and Mapuche and Yagane peoples of Chile have many narratives about hummingbirds (Rozzi, 2004).

5.2.7 Landscape management practices and fostering biocultural diversity for pollinators and pollination across the world

A wide range of ILK-holders across the world value nature's gifts of landscape management practices that foster biocultural diversity for pollinators and pollination. Relevant landscape (social-ecological) management practices include: taboos on felling bee-hive trees and pollinator-habitat forest patches (Césard and Heri, 2015); kinship relationships requiring respect and care with pollinators (Hill et al., 1999; Gasca, 2005); fire management to enhance pollination by increasing floral resources (Vance et al., 2004); mental maps and animal behaviour knowledge to hunt honey (Si, 2013); seasonal rotations for prolonged harvests (Titinbk 2013, Samorai Lengoisa 2015); landscape patch management (Bodin et al., 2006); use of biotemporal indicators (observed changes in biological processes over time) including birds and flowering to signal the time for burning vegetation and to harvest honey (Athayde, 2015); placement of pollinator-dependent crops (e.g, cucumber) close to pollinator-rich forests (Calle et al., 2010); and encouragement of bees in housing.

5.2.7.1 Taboos that protect pollinators and pollination resources

Indigenous peoples and local communities often place taboos prohibiting hunting or disturbance of animals, plants and places that extends to protection of pollination resources (Colding and Folke, 2001; Saj et al., 2006; Kideghesho, 2009) (Case example 5.4). For the Berawan people of Loagan Bunut, Sarawak (Malaysian Borneo), the Tanying tree (*Koompassia excels)* is revered for its spiritual values (Franco et al., 2014) with a taboo on its felling, generating conservation of the tree, the bee nests in it and other animals that depend on it.

In Africa, traditionally-protected forests provide habitat for pollinators such as the fruit bat (*Rousettus aegyptiacus*) that pollinates the baobab (*Adansonia digitata*), which is widely used for food and medicine (Start, 1972). Examples include West Africa's sacred groves (Decher, 1997); and the *kayas* of the East African coastal region maintained by the Mijikenda peoples (Githitho, 2003). In southern Madagascar, local taboos provide strong and well-enforced protection for existing patches of forest (Tengö and Belfrage, 2004). Spatial modelling of crop pollination provided by wild and semi-domesticated bees (Apoidea) indicates that, in spite of the fragmented patches of forest across this largely cultivated landscape, these insects still contribute pollination throughout the entire landscape matrix; the taboo system also protect the bees and their pollination (Bodin et al., 2006). In China, communities use indigenous knowledge and cultural traditions to support hunting taboos, and protection of sacred sites and forest habitats (Xu et al., 2005).

Case example 5.4. Prioritising protection of habitat and bee hive-trees in Indonesian forests.

Location: Indonesia

Petalangan indigenous people

Petalangan is a group of indigenous people practicing hunting, fishing, and swidden agriculture, living relatively isolated at the forest margins in Riau Province, Sumatra, Indonesia. The Petalangan community view bees as a symbol of health and prosperity and the *sialang* trees, where the bees nest, as a symbol of the universe. *Sialang* is a generic term of trees that have bees nests on them and includes several species of trees: *Ficus* spp.; *Koompassia excelsa (mangaris); Octomeles sumatrana*; *Artocarpus maingayi*; *Macaranga* spp.; *Koompassia malaccensis (kempas);* and *Metroxylon* spp.

No one can cut down the *sialang* trees and all other trees surrounding the *sialang* trees. The *sialang* trees and surrounding habitat are then conserved (named as *rimba kepungan sialang*, meaning patch of forest surrounding *sialang*). The community views the trees as integral to water for the area. Petalangan people perform a ritual to keep bee trees healthy by watering the base of the tree followed by the slaughtering of chicken (3 colours) (Titinbk, 2013). Fruits are usually harvested from the forests surrounding the habitat of *sialang* trees (Buchmann and Nabhan, 1996).

5.2.7.2 Kinship relationships that protect pollinators and pollination resources

Kinship relationships also place responsibilities on people to care for animals with whom reciprocity means the well-being of both are inter-dependent (Rose, 1996; Sasaoka and Laumonier, 2012). Bees and people have totemic relationships in several Australian Indigenous societies (Hill et al., 1999; Prideaux, 2006) (Case example 5.5). The Lardil and Laierdila people's classification system based on totemism (which differs from their folk taxonomies) divides phenomena from the foundation of the clan totem into two patrimonies and four semi-moities. Interestingly, wind and a wind-pollinated tree are in the same semi-moiety, as are various fruits and pollinators (McKnight, 1999). Uitoto communities in Colombia pay special cultural respect towards scarab beetles, important pollinators, which are used for rituals and as medicine (Gasca, 2005). The Pankararé people from the arid zones of northeast region of Brazil classify bees or "abeias" according to the behavioral aspects as "abeias-brabas" (fierce bees) and "abeias-gentle" (gentle bees), and divide bees into three ethnofamilies depending on the presence and/or absence of the sting. Bees and wasps are protected from human exploitation by guardian spirits of plants and animals called *encantados* (Costa-Neto, 1998).

Case example 5.5. Sugar Bag dreaming. Kinship relationships protecting bees in Australia.

Location: Arnhem Land, Northern Territory, Australia
Yolngu indigenous people (Fijn, 2014)

The stingless bees *birrkuda* and *yarrpany* are classified as Yirritja and Dhuwa by the Yolngu people who separate their world into two kinship groups with these names. This has led to the development of specific songs, dances and power names associated with each bee and their specific products. The Yolngu appreciate the bees' role in pollinating native plants (e.g., *Melaleuca* spp.) and their nest associations with particular plants [e.g., Stringybark trees (*Eucalyptus tetradonta*)]. The collection of honey and other products (wax, pollen and larvae) provides both dietary health and social benefits. Psychological benefits include improved social relationships through cooperation among people. Hunting and harvesting of the honey, bee products and larvae is considered favorite activity for Yolngu of all ages and of both sexes (Figure 5-12.). Apart from glucose, dietary benefits from the consumption of honey and larvae include carbohydrates, protein, fat, and essential minerals.

Both bees provide the Yolngu with strong connections that influence culture, social interactions and interaction with nature itself. Existing artefacts and paintings demonstrate a very long relationship between indigenous Australians and stingless bees. More specifically, historic evidence includes the presence of wax figurines from Arnhem Land (North-Eastern Northern Territory) (dated to be more than 4,000 years old) and rock wall paintings depicting bee hunting that has been dated from the Mesolithic period (Langley and Taçon, 2010).

48. Figure 5-12. Yolngu women collecting sugarbag in Arnhem Land, northern Australia.
Still photos from the video "Sugarbag Dreaming". © Natasha Fijn. Reproduced from Fijn (2014) with permission.
A: A woman and two children in search of stingless bees, northeast Arnhem Land. Still from "Sugarbag Dreaming" video. © Natasha Fijn.

B: The extraction of honey pots filled with bright yellow pollen from a Yirritja stingless bee nest, within a stringybark trunk. Still from “Sugarbag Dreaming” video. © Natasha Fijn.

C: Scooping up liquid honey using a makeshift spoon made from a stick with a frayed end. Still from “Sugarbag Dreaming” video. © Natasha Fijn.

5.2.7.3 Mental maps and animal behaviour knowledge as management practices

Knowledge in itself is a vital management practice for honey-hunters. For example, the Solega people of southern India have extensive mental maps of the location of individual trees and significant harvesting sites in the forest. Their knowledge of different migration and settling patterns of the various honey bee species of the region, and of their breeding schedules, is vital to their honey-hunting technologies (Si, 2013). Detailed knowledge of local people about behaviour of *Apis* spp. underpins diverse swarm capture, especially of wild swarms around the world (Marchenay, 1979). Indigenous people in Yuracaré, Cochabamba, Bolivia have detailed knowledge of the native birds that are pollinators of the forest, the trees that they pollinate, and their behaviour, which is vital to their customary forest usage (Castellón-Chávez and Rea, 2000). The Jenu-Keruba people, honey hunters in Kodagu southern India, identify 25 different micro-habitats in their forest and take advantage of four different bee species producing honey in habitats and seasons (Demps et al., 2012a).

5.2.7.4 Fire management to enhance pollination resources

Vegetation fires in bear ‘grass’ (*Xerophyllum tenax*, in the Lilieaceae family*)*, pollinated by pollen-eating flies (primarily members of the family Syrphidae), beetles (primarily *Cosmosalia* and *Epicauta* spp.), and small bees (Vance et al., 2004), are managed by First Nations peoples in northern America to ensure production of this grass and promote qualities suitable for contemporary traditional purposes, such as basketry that requires strong, flexible, straight leaves (Charnley and Hummel, 2011). Traditional First Nation fire practices “favored beargrass, its habitat, its cultural uses, its flowers, and presumably, associated pollinator communities as well as other species that use it for food, habitat, and nesting material” (Charnley and Hummel, 2011). Experiments on abandoned farmland in south-eastern USA have found that fire promotes pollinator visitation indirectly through increasing the density of flowering plants, in that case the forb *Verbesina alternifolia*, suggesting the usefulness of fire management as a tool for supporting pollination (Van Nuland et al., 2013).

5.2.7.5 Manipulation of pollination resources in different seasons and landscapes patches

Diverse management practices manipulate and access different resources in different parts of the landscape at different seasons. In the Petalangan community in Indonesia, pollination is enhanced through seasonal patterns of planting and harvesting, so that bees (*Apis dorsata* and *Apis florea*) can nest up to four times a year in the *sialang* trees, in accord with the flowering of different crops and during the slash and burn period that opens the forest to start planting (Titinbk, 2013). In the Kerio Valley of Kenya, papaya farmers maintain hedgerows for both practical, aesthetic and cultural reasons that conserve habitat and resources for hawkmoth pollinators of this dioecious pollinator-dependent crop (Martins and Johnson, 2009). Similar patterns can be observed in relation to cacao and biodiversity in Ghana (Rice and Greenberg, 2000; Frimpong et al., 2011) and cowpea in Nigeria (Hordzi et al., 2010).

Farmers in Roslagen (Sweden) protect bumble bees as important pollinators, including by restricting cutting of a tree species that flowers in early spring when other pollen- and nectar-producing plants are rare. In both locations, pollinator presence is further enhanced by the making of beehives and the management of field boundaries and mixed land that provides suitable insect habitat (Tengö and Belfrage, 2004). Producers of *maracuyá* (*Passiflora edulis,* passionfruit*)* in Colombia highly value pollinators, particularly black carpenter bees (*Xylocopa* spp.) which use dry trunks as their main habitat. Social bees (*Apis mellifera* and *Trigona* spp.) and hummingbirds are also important, and all three groups depend on proximity to forest. Farmers value the pollination from the forest highly (Calle et al., 2010).

5.2.7.6 Biotemporal indicators for management actions

Seasonal “biotemporal” indicators, or “indigenous knowledge markers” trigger diverse management practices (Leonard et al., 2013; Athayde, 2015). Flowering is the main indicator of times for honey harvests among Indonesian forest communities (Césard and Heri, 2015) (Case example 5.6). Among the Kawaiwete (Kaiabi) people in the Brazilian Amazon, indicator species inform the start of the rainy and dry season. *Kupeirup*, a powerful female ancestral being, created crops and taught her sons how a flock of birds (a type of parrot) announces the right time to burn the fields (Silva and Athayde, 2002). The Boran people from Kenya deduce the direction and the distance to the honey nest from the greater honeyguide’s (*Indicator indicator*) flight pattern, perching height and calls, and reward the bird with food that is more accessible after they have opened the nests (Isack and Reyer, 1989). Interactions with honey-guides have been found to increase the rate of finding honey by Hadza people in northern Tanzania by 560% (Wood et al., 2014). The Ogiek people of Kenya use two types of birds for indicators when honey-hunting in the forest, and have migratory patterns that follow the production of different bees in the lowlands and the highlands (Samorai Lengoisa, 2015).

Case example 5.6. Biotemporal indicators for honey hunting.

Location: East Kalimantan, Indonesia

Punan indigenous peoples and local communities

In East Kalimantan, the Punan Kelay’s (in Berau Regency) practices of bee-hunting are full of rituals that are stimulated by biotemporal indicators (Inoue and Lugan-Bilung, 1991). Natural signs trigger honey harvesting activities (Widagdo, 2011). If they hear certain calling of birds, they refrain from climbing the trees, because it is an indicator that the process will not be successful or may be dangerous. Before they start harvesting, traditionally they “call” the bees by the *keluwung* ceremony early in the honey season – usually around early October. The ritual involves erecting a tree branch and forming “nest like” figures from clay, followed by a ceremonial ritual expulsion of ghost/spirits from the tree, by throwing a partridge egg to the base of the tree. All these rituals are performed by chanting and praying, including a Christian element to traditional ceremonies (Widagdo, 2011).

Among the Punan Tubu (in Malinau Regency), the season for honey harvesting is signaled by the flowering of meranti (*Shorea* spp.), sago palm and several fruit trees, accompanied by singing of birds (e.g., great argus pheasant *Agursianus argus*) and cicadas, and followed by the breeding season for the wild pig (*Sus barbatus*). Hordes of boars migrate in anticipation of fruits. The mythology of the Punan Tubu tell of the link between bees on huge tree branches and pigs underneath since the creation time (Mamung and Abot, 2000).

5.2.7.7 Providing pollinator nesting resources

Management practices for pollinators link landscape management with traditional housing in the Nile delta. Egyptian clover, part of mandated crop rotation, is pollinated by *Megachile* spp. (solitary bees) that nest in tunnels in the walls of mud houses. The bees depend on people to create a dynamic nesting habitat by constantly renewed mud walls, alfalfa and clover fields. However, populations of *Megachile* spp. in mud houses have been displaced or eliminated as modern brick and cement block buildings have replaced traditional mud houses (FAO, 2008). In Bolivia, one particular stingless social bee (“chakalari”) is well known locally, in part because it makes its hives on the sides of the adobe houses (FAO, 2008). Other stingless bees like *T. angustula*, a species very appreciated for its honey, also use any cavity or container available in the houses to build their nests (Nates-Parra, 2005).

5.2.8 Diversified farming systems that influence agrobiodiversity, pollinators and pollination

Diversified farming systems of Indigenous peoples and local communities across the globe contribute to maintenance of pollinators and pollination resources, and represent an important multi-functional alternative and adjunct to industrial agriculture (Kremen et al., 2012). These farms integrate the use of a mix of crops and/or animals in the production system. They employ a suite of farming practices that have been found to promote agro-biodiversity across scales (from within the farm to the surrounding landscape), and incorporate ILK systems, often involving hybrid forms of knowledge, negotiated between science, practice, technical, and traditions (Barber et al., 2014). These farming practices in reality merge with the landscape management practices in the previous section. Here we consider some pollination-related aspects of several farming systems: swidden cultivation; home gardens; commodity agro-forestry; and farming bees.

5.2.8.1 Shifting cultivation

Swidden (shifting cultivation) systems, demonstrating diverse interdependencies with pollinators, remain important in tropical forest systems throughout the world, and are the dominant land-use in some regions (van Vliet et al., 2012; Li et al., 2014). For example, the traditional Mayan Milpa, multi-cropping swidden cultivation, produces a patchy landscape with forests in different stages of succession through spatial and temporal rotation, a dynamic system that produces a diverse array of plants, nearly all of which are pollinated by insects, birds and bats (Ford, 2008). Milpa has co-created some, and fostered much, of current forest plant diversity and composition during millennia of gardening the forest (Ford and Nigh, 2015). This system produces a territory of farms that combine agricultural, forestry and stockbreeding activities, organized around a domestic group, depending on local knowledge on the vegetation species and their uses, the domesticated animals and the crop systems (Estrada et al., 2011) (Case example 5.7).

Case example 5.7. Mesoamerican Milpa systems, diversity and fecundity.

Location: Guatemala and Mexico
Mayan-descendant people

The Popol Vuh, the Sacred Quiche Mayan book of Creation, begins with the clarification that “this book’s face is hidden”, directing the knowledge seeker to revelations in the way of living, the memories, culture, oral transmission, beliefs, spirituality and worldview of the people. In the Popol Vuh are stories of the hero twin gods, *Hunahpu* (Blowgun hunter) and *Xbalanque* (Young hidden/Jaguar Sun). The twins play a ballgame in the Underworld court and defeat the Gods with help of various animals and for their victory, their father, *Hun Hunahpu*, is resurrected in the form of maize (Raynaud, 1977).

The contemporary traditional Mayan Milpa systems keep these traditions alive today, an evolving and active response to changing contexts (Schmook et al., 2013). The Milpa system also maintains in the surroundings diverse sources of food for people and resources for pollinators: macuy (*Solanum* sp.), bledos (*Amaranthus* sp.), Chaya (*Cnidoscolus chayamansa),* Tz’oloj-bell tree dahlia *(Dahlia imperialis),* Malanga (*Xanthosoma violaceum*), *Amaranthus caudatus*; and cultivated species like chayotes (S*echium edule*), chile (*Capsicum* spp.), and black beans (*Phaseolus*), as wild relatives or in process of domestication, producing the high diversity of the system (Azurdia et al., 2013; Janick, 2013).

A product largely related with fecundity is the honey from the Mayan Sacred Bee *Melipona beecheii* (*Xunan-kab*), associated with the concept of the Earth as a living entity composed of spirit, blood and flesh. Honey from *Xunan-kab* is considered “warm” and is seen as a living and essential fluid from the land where the bees are maintained and that men extract to obtain some of its vitality and fertility, but that eventually needs to be given back in the form of sacrifices (de Jong, 2001; González-Acereto et al., 2008).

Honey from *Xunan-kab* is used in special ceremonies to bless the Milpa for good crops (Quezada-Euán et al., 2001).

5.2.8.2 Home Gardens

Home Gardens, capitalised here to indicate those with food, support agro-biodiversity globally, in both developed and developing world contexts (Eyzaguirre and Linares, 2004; Gautam et al., 2006; Bailey et al., 2009; Reyes-García et al., 2012). Home Gardens produce a variety of foods and agricultural products, including staple crops, vegetables, fruits and medicinal plants. They are characterized by structural complexity and multi-functionality, acting as social and cultural spaces where knowledge is transmitted, income and livelihoods improved, and pollinators find habitat (Agbogidi and Adolor, 2013). Home Gardens in Chinango, Mexico achieve almost double the fruit set of both wild and managed populations of the columnar cactus *Senocereus stellatus* (Arias-Coyotl et al., 2006). Management practices in these gardens appear to reduce some negative pollination impacts associated with human cultivation; although flowers in the gardens received fewer total visits, they received significantly more visits from long-nosed bats (*Leptonycteris* spp.), and significantly more pollen grains on the stigmas (Arias-Coyotl et al., 2006) (Case example 5.8). Many traditional Home Gardens are forms of agroforestry; in tropical south-west China local people continue to collect, utilize and manage wild forest resources into these systems, thereby maintaining diverse genetically diversity, for example of the pollinator-dependent *Acacai pennata* (Gao et al., 2012).

Case example 5.8. Home Gardens, pollinator diversity and domestication in Mesoamerica.

Location: Guatemala and Mexico

Mayan-descendant people

Home Gardens have ancient roots in Mesoamerica. The practice originated around 6,000-200 BC probably as a way to keep food resources close and to attract animals for harvest – white tail deer, peccaries, squirrels and birds, including the great curasow, oscillated turkey, and quail. Since the Spanish invasion, Home Gardens have been integrating exotic domesticated species for many different purposes: medicine, food, ornament, diversity itself, raw materials for clothing, firewood and wood for construction (Janick, 2013). Home Gardens contain perennial habitat for pollinators (insects, birds and bats). Mesoamerican Home Gardens include at least 811 cultivated species, 426 plant species with multiple uses, 19 domesticated animal species and 25 semi-domesticated wild fauna. Mesoamerican Home Gardens are where the most ancient technologies for stingless beekeeping originated with the "Mayan honey bee" *Melipona beecheii*, kept in east-west oriented, especially built huts called *Nahil-kab*. Colonies are reared in horizontal hollow logs called *hobones* (Quezada-Euán et al., 2001). In Mayan mythology, beekeepers are seen as guards and caregivers of *Melipona beecheii* rather than owners (de Jong, 2001). Other indigenous Mesoamerican groups like Nahuas and Totonacs practice stingless beekeeping along the highlands of the Mexican east coast, cultivating hundreds of colonies of *Scaptotrigona mexicana* (*Pisil-nek-mej*) in clay pots (Quezada-Euán et al., 2001).

5.2.8.3 Commodity agroforestry

Agroforesty systems globally support commodity production, particularly of coffee, rubber, areca nut and cacao, with variable outcomes for pollination highly dependent on the intensity of management, for example of synchronicity of flowering (Boreux et al., 2013; Robbins et al., 2015). Two decades of ecological research into traditional shaded coffee plantations in Latin America show they provide refuges for biodiversity and a range of ecosystem services such as microclimatic regulation, and nitrogen sequestration into soil and pollination. One study identified the most predictive factors for bee abundance and species richness which were tree species, the number of tree species in flower, and the canopy cover of the coffee agroforestry (Jha and Dick, 2010; Jha and Vandermeer, 2010). An inverse relationship has been identified between farm size and agricultural productivity — in a number of countries smaller farms have higher crop yields than do larger ones (FAO, 2014c; Larson et al., 2014). While these farms are more labour-intense than capital-intense, which limits their extent, especially in contexts of rural-urban migration, evidence is accumulating that in the tropical world the resulting landscape matrix with fragments of high-biodiversity native vegetation amidst the agriculture produces both high-quality food to the most needy and maintains ecosystem services such as pollination (Perfecto and Vandermeer, 2010; Nicholls and Altieri, 2013). Commodity agroforests with date palms have developed traditional direct hand pollination, including different techniques for date palms in several countries (Battesti, 2005; Boubekri, 2008; Tengberg et al., 2013).

5.2.8.4 Farming of domesticated and semi-domesticated bees

The diversified farming systems of indigenous peoples and local communities include a range of practices for farming fully- and semi-domesticated bees. Family farmers in southern Brazil, settlers of the agrarian reform, *quilombola* (Afro-descendant peoples), and indigenous peoples of the region confirm that the presence of hives generates beneficial results for their crops, and noticeable improvements in the swarms that occur when the hives are installed next to abundant and diverse forests (Wolff, 2014). Traditional beekeepers in Morocco utilise the heterogeneity of their landscape, placing *taddart* (traditional hives) to adapt to climatic variations (long period of drought) and varying priorities, such as honey production, pollination of cultivated fields, swarm multiplication, and pollination of argan (*Argania spinosa*) trees (Simenel, 2011; Roué et al., 2015). The beekeepers use knowledge about the specific influences of different plants on bee behaviour in their management (Crousilles, 2012). Many rural farming communities in sub-Saharan Africa include beekeeping as a means of sustainable development and for nutrition, managing wild plants, hedgerows, fallow areas and agro-forestry systems for improved pollinator and livestock nutrition. Some farming landscapes are known to have especially high bee diversity adjacent to forested areas (Kasina et al., 2009; Gikungu et al., 2011).

Meliponiculture (stingless bee keeping) is presently increasing throughout the tropical and sub-tropical world and is supported by a range of practices and innovations for rearing stingless bees, farming their honey in unique hives, managing their pests and for stimulating their multiplication (Cortopassi-Laurino et al., 2006)(Case example 5.9). Local communities (indigenous peoples and settlers) in the "impenetrable chaqueño" (Argentina) are using meliponiculture as a tool for preserving this region through the application of modern techniques of reproduction and management of the stingless bees (Meriggi et al., 2008).

Case example 5.9. Farming and semi-domesticating stingless bees by tribes in India and South America.

Location: Tamil Nadu, India and Brazil

Kani Tribes (Kanikudiyiruppu, Mayilar and Periyamayilar) and Gorotire-Kayapo Indians

Tribal people of Western Ghats of India are rearing stingless bee (*Trigona* sp.*)* very successfully for pollination (Kumar et al., 2012). The Kani tribes, in Kalakkad within Mundanthurai Tiger reserve (Tirunelveli district, Tamil Nadu) are using a very peculiar bee hive to rear these bees, which are normally wild. The honey produced by *Trigona irredipensis* is highly valued for treatment of many infections, and is a weaning food for infants. *Trigona irredipensis* are reared in hollow sections of bamboo that are tied below the roof of a hut and produce around 600-700g honey per year. Traditional knowledge about the honey’s medicinal properties has recently been confirmed by a meta-analysis of three double-blind randomized clinical trials that found honey-coffee mixture outperforms the drug prednisolone in treatment of post-infection persistent cough (Raeessi et al., 2014).

The Gorotire-Kayapo Indians have a semi-domesticated system of beekeeping for nine species of bees, including *Apis mellifera*. Brazil has a strong tradition in meliponiculture, especially in the northeast and northern regions (Cortopassi-Laurino et al., 2006). The species *Melipona scutellaris*, *M. quadrifasciata*, *M. rufiventris*, *M. subnitida*, *M. compressipes*, *Tetragonisca angustula* and *Scaptotrigona* spp. are the most common species raised. Diverse indigenous names for these species have linguistic heritage values: *jataí, uruçu, tiúba, mombuca, irapuá, tataíra, jandaíra, guarupu,* and *mandurim* (Lenko and Papavero, 1996; Nogueira-Neto, 1997; Villas-Bôas, 2008).

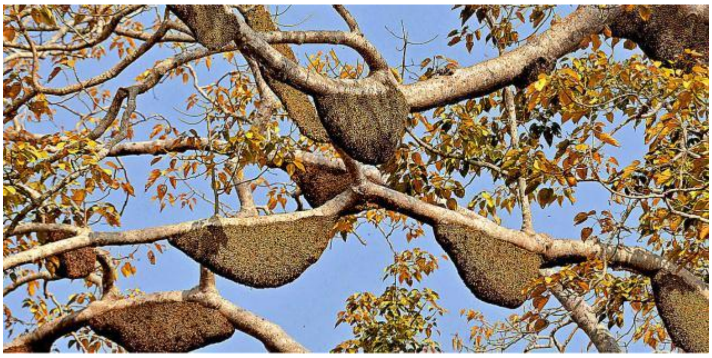
5.2.9 Innovations in honey hunting, hives, bee handling and bee products

Traditional beekeeping and honey hunting practices have generated a wealth of innovations across the planet (Brown, 2001; Hausser and Mpuya, 2004). An array of diverse non-destructive stratagems are used by honey-hunters (Joshi and Gurung, 2005) (Figure 5-13. A and B), diverse apiaries and husbandry methods are used by human beekeepers, and a multitude of products have been derived from bees (Crane, 1999).

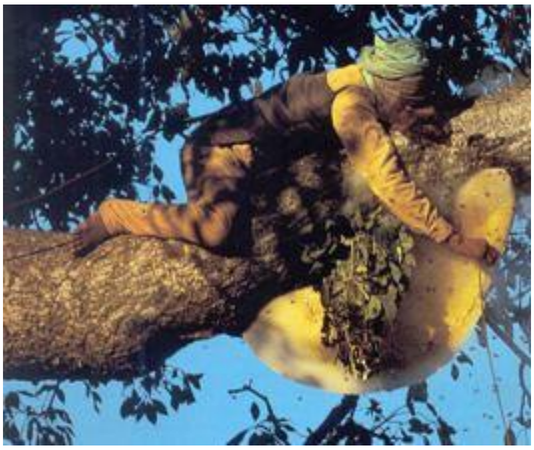
Honey hunters in Ethiopia manufacture a permanent system for scaling trees in order to make their task easier (Verdeaux, 2011). In India, honey hunters scale towering cliffs of the Nilgiri Hills of South India using ladders and social technology of songs at various stages of the operation (Anderson, 2001; Sunil Kumar and Reddy, 2011) (Figure 5-13. C). In Nepal they use large bamboo ladders (Valli and Summers, 1988) (Figure 5-13. D). Honey hunting in the tropical forests of Cameroon is a perilous activity involving climbing large tree trunks with a rope made of liana, carrying a small L-shaped axe to cut open the nest, a smoking tube for fumigating the aggressive bees, and a container to keep the precious liquid without losing a single drop (Ngima Mawoung, 2006). In central Africa, the indigenous peoples of the rainforest have developed many specific tools for honey collecting, including instruments to climb trees, and also gestures to communicate during honey hunting (Bahuchet, 1989) (Figure 5-13. E).

49. Figure 5-13. Innovations in honey hunting from around the world.

A: Colonies of giant honeybees (*Apis dorsata*) in Bahatpur village in Kulsi Reserve Forest in Kamrup district, India. ©Ritu Raj Konwary. Reproduced with permission.



B: Honey hunter collecting from the nests of *Apis dorsata*. © Girish Chandra. Reproduced with permission.



C: Kurumba Indigenous people of the Nilgiris starting their yearly harvest, scaling precipitous cliffs and risking their lives to collect honey of the wild *Apis dorsata*. © Riverbank Studios. Reproduced with permission.

D: Honey hunter from the Gurung population of Nepal risk their lives to harvest *Apis dorsata laboriosa* on Himalayan cliffs. © Andrew Newey. Reproduced with permission.

E: Bakaya (forest-dwelling indigenous people) man in Cameroon climbing a tree to harvest honey. © Timothy Allen. Reproduced with permission.

In France and Spain, innovations in use include traditional swarming methods, extended beekeeping vocabulary, harvest and honey extraction techniques, and diverse smokers and smoking methods (Mestre and Roussel, 2005). Diverse traditional beekeeping techniques for construction of hives, the capture, promotion and delay of swarms have been reported across Asia (Case example 5.10. -10) and west Africa (Villières, 1987), east and north Africa (Hussein, 2001; Roué et al., 2015) (Figure 5-14.), and in Chad (Gadbin, 1976). In the southern part of Algeria, the local people’s tradition is to implement isolated hives in open areas, or organize houses and villages specially built for bees ("houses-apiary" located in "villages-apiary") (Rivière and Faublée, 1943; Hussein, 2001).

50. Figure 5-14. Traditional Ethiopian bee hives in trees.
© Peter Kwapong. Reproduced with permission.

The hives are simple six-foot cylinders made of cane and lined with leaves. They are placed empty in the forest tree tops with the leaves of the Limich plant (*Clausenia anisate)* used to attract swarms of honey bees.

In Indonesia, traditional beekeepers use a rafter system, where a piece of wood is paced in a tree to attract nesting bees (Case example 5.10. 10, Figure 5-15.). Enduring traditional beekeeping in the Cévennes (a mountain range in the South of France) uses a specific type of hive, dug in a portion of a tree trunk, that is called *ruchers-troncs* (Lehébel-Perron, 2009). Chestnut tree hives repel wood parasites and remain in production for several hundred years without any chemical treatment (Chevet, 2010; Pierlovisi, 2015). Pastoral beekeeping, also called transhumance of bees, has existed for a long time in the landscapes of Europe. Traditional pastoral beekeepers transport their hives directly to orchards during flowering periods, delivering mutual benefit for beekeepers and farmers, resulting in many different types of honey (Mestre and Roussel, 2005).

Case example 5.10. Technological innovations for hives and honey harvests in Asia.

Location: India, Laos, Kalimantan

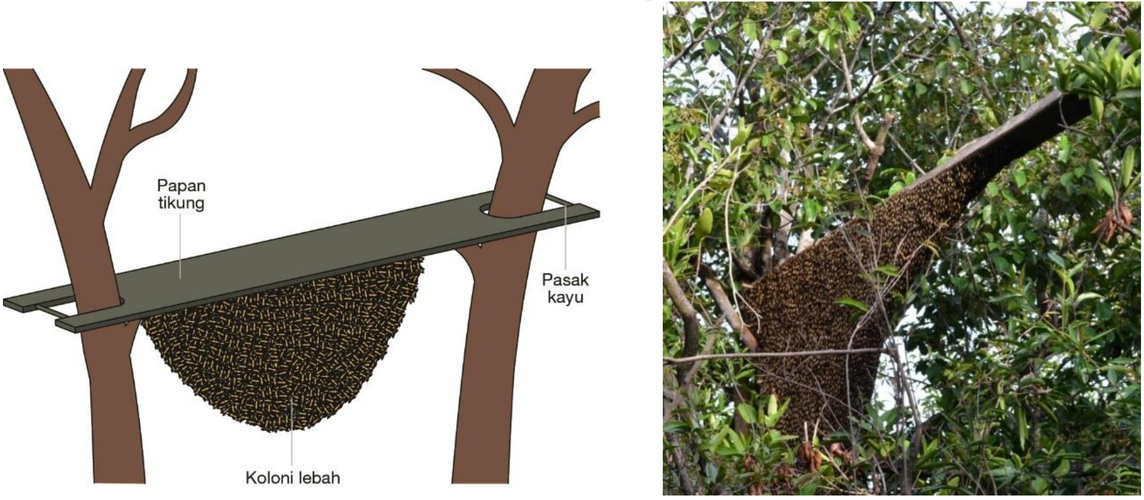
Several local communities across these locations

Several traditional honey harvesting methods with various materials and types are used by some local people in India. Kinnaur people used bamboo to make log hives (Beszterda, 2000). Chamoli people used wall hives made from cow dung or clay, log hives from bamboo and rectangular wooden box hives with various sizes in different localities (Tiwari et al., 2013). Kani tribes used bamboo hives for stingless bees (Kumar et al., 2012).

Local people in Laos, particularly in Northwestern region of Laos (Meung district of Bokeo Province) use rustic log hives for their traditional beekeeping practices (Chantawannakul et al., 2011). In Indonesia, the basic structure for beekeeping involves putting two poles into the ground, or using two tree branches, and adding a third pole or sheet of wood on top. In Western Kalimantan the structure is called *tikung* (Figure 5-15.), in Sulawesi it is called *tingku*, and in the Belitung it is known as *sunggau*. Several communities have also developed "nesting sites" to attract feral colonies of *Apis dorsata* (Hadisoesilo and Kuntadi, 2007).

In Belitung, people link gelam flowers (*Melaleuca leucadendron*) to attracting large swarms from the nearby islands of Sumatra and Borneo. Honey bees are seen to first arrive for the pollen, then proceed to build wax comb and wait on the rafter until the flowers produce nectar (Césard and Heri, 2015).

51. Figure 5-15. A honey plank (tikung) used in traditional beekeeping in the Danau Sentarum National Park, West Kalimantan province, Indonesia.
Source: Hadisoesilo and Kuntadi (2007). Photo © N. Césard. Reproduced with permission.



In Nepal and India, innovations extend to pest management practices such as use of cow dung (effective against wax moth, wasp, lizard) and polythene sheets to protect against lizards and tree frogs (Singh, 2014) (Case example 5.11. 11). In south Morocco, beekeepers manage *Varroa* sp. mite by smoking hives with certain plants that inhibit the action of the mite, and by placing their hives near plants from which bees harvest latex that is transformed into propolis with mite-inhibiting effects (Roué et al., 2015). In Brazil, technologies and innovations of traditional practices of stingless beekeeping have been brought together into several manuals (Nogueira-Neto, 1997; Venturieri, 2008; Witter and Nunes-Silva, 2014).

Many innovations have developed from use of bee wax in east Africa. Wax is seen as a negotiable residue or can serve to repair objects, to soften skins, and to make crafts or jewellery (Gadbin, 1976; Villières, 1987). In Australia, cerumen (wax made by bees from plant materials and their excretions) has been found in protective covers, fashioned around ancient rock paintings, to protect them from rain and erosion, and to create shapes of humans, dingoes, turtles, and spirit figures on the rock surface (Halcroft et al., 2013). Cerumen is still used by Australian Aboriginal artists and craftsmen to manufacture items for use and sale including hunting tools such as spears (“kek”) and woomeras (“thul”), as well as firesticks “(thum pup”) and mouth pieces for didgeridoos, a traditional musical instrument (Yunkaporta, 2009; Koenig et al., 2011).

Case example 5.11. . Innovations for swarm capture, bee handling and disease management in Nepal.

Location: Jumla, Western Nepal (Saville and Upadhaya, 1998)

Jumla indigenous people

Apis cerana, the Asian bee, is threatened throughout Asia. The *Apis cerana* variety found in Nepal is high yielding compared to other Himalayan strains. Hollowed out logs are used to made cylindrical and square cross section hives in Jumla. The timber logs, i.e., *Ilex dipyrena* (kharso), *Juglans regia* (okher) and *Pinus wallichii* (sallo) are used for bee hives. About 85% of farmers used different baits to attract and capture the swarm. Mostly beekeepers used baited hives, rubbing their hives with ‘gosard’ (a hive baiting substance), and few of them used raw honey only. Some farmers scorch the inside of the hive and scrub it with fresh walnut leaves. Other materials are also used: cow ghee (clarified butter); wild rose flowers (*Rosa moschata*); dhoopi (*Juniperus* spp.); (roasted) de-husked rice; (roasted) barley; or mustard oil and cloves.

For handling bees, a local *Artemisia* species known as *gwiepattior titepatti* (*Artemisia vulgaris*) is placed near the bee hive and rubbed to give off a strong scent. A *kangreto,* made out of old cotton cloth, is tied into a roll and used as smoker. Some people used specific herbs to produce a good smoke that encourages bees to leave the combs without inducing too much disturbance.

Jumla farmers recognize diseased bees in various ways: angry bees, absconding, inactive bees, or bees hanging together by the feet. Brood disease is recognized when bees are seen throwing out dead larvae, or by sour smell and black combs. Buckwheat is valued for its bitter properties and applied around the exit hole of bee hives during the spring. Bees encounter it on their way out for the first foraging trips of the year as a medicine against disease that affects bees at this time. In Jumla, some farmers use *Juniperus* spp. smoke for disease treatment (Saville and Upadhaya, 1998).

5.2.10 Adaptation to change

Beekeeping has been demonstrated to be closely linked with traditional knowledge and adaptation to climate change in Ethiopia (Bogale, 2009; Kumsa and Gorfu, 2014), and it is connected to self-reliance in Southern Africa (Illgner et al., 1998; Nel et al., 2000). Seven mechanisms of environmental adaptation have been identified among the Xingu Kawaiwete (Kaibai) of Brazil: 1) knowledge innovation in development of nomenclature for ecological zones and new species of bees; 2) increase in diversity of resources used for different purposes (e.g., to build canoes) due to village sedentarization and scarcity of important forest resources; 3) agrobiodiversity conservation and recuperation of crop diversity, including through cultivating pollinator resources; 4) travel to ancestral land to collect resources; 5) substitution with other local species; 6) exchange of varieties and seeds among families, villages and other ethnic groups; 7) semi-domestication (e.g., of invasive bees) or intentional management – through experiments for planting and protecting key resources (Athayde et al., 2006; Athayde, 2010; Athayde, 2015) (Case example 5.12. 12).

Case example 5.12. . Innovations to foster pollinators and pollination based on traditional techniques.

Location: Central America and Colombia

Indigenous women; Florina López Miro, oral presentation, p. 39 (López et al., 2015)

Co-produced case example

Underpinned by direct interactions with indigenous and local knowledge-holders

"In many cases insects like bees and butterflies that we used to see in great quantities in our communities are not there anymore. Regarding food production, we have lost our people's tradition seeds and propagules because the work of the pollinators has been affected. Our knowledge has been eroded by the impact of climate change in our communities, related to the loss of traditional seeds and propagules."

"Many women in different places traditionally manage and control the seeds and propagules, but this is decreasing. Now women are working to recover IK and use seeds (which require pollination) as well as propagules, for example, in the processing of yuca (*Manihot esculenta*). In Colombia, a group of Witoto (Huitoto) women working to recuperate traditional seeds are running a restaurant that sells traditional cuisine … they've developed a fruit ice cream [that provides income]. In other words, they are developing projects to support biocultural diversity, [including seeds requiring pollination, not just vegetative propagation]. In Guatemala, Mayan women are working on orchid production, encouraging pollination. In El Salvador, they are working with petals of the veranera flower to produce a medicinal syrup. We are also working with young people. In sum, we are innovating with IK, looking for ways to improve traditional techniques …Pollination is very important."

5.3 Pollinators, pollination and good quality of life

5.3.1 Good quality of life and categories of values

Pollinators support numerous categories of value that contribute to good quality of life (26). Here we consider three categories of relational values through a socio-cultural valuation lens — heritage, aesthetics and identity — and a further three categories through a holistic valuation lens — livelihoods, social relations and governance.

5.3.2 Heritage values, pollinators and pollination (socio-cultural valuation)

Heritage can be understood as *tangible* physical objects and places that are passed between generations, and *intangible* aspects such as language or practices. Historical features, practices and places are considered heritage because we ascribe value to them (Muňoz Viňas, 2005). The *Convention for the Safeguarding of the Intangible Cultural Heritage*[13] and the *Convention Concerning the Protection of the World Cultural and Natural Heritage*[14] are international agreements to recognize and protect intangible and tangible heritage, and several have been listed where the heritage values depend on peoples' interactions with pollinators and pollination webs. The Globally Important Agricultural Heritage Systems[15] (GIAHS), an initiative of the Food and Agricultural Organization (FAO) of the United Nations, supported by a number of partners, has five criteria for selection, one of which (biodiversity and ecosystem function) specifically recognizes pollinators and pollination services.

The GIAHS initiative aims to safeguard and protect the world's agricultural systems and landscapes that have been created, shaped and maintained by generations of farmers and herders based on diverse natural resources, using locally-adapted management practices (Koohafkan and Altieri, 2011). There are now 32 designated GIAHS sites globally, and a further 95 potential sites, of an estimated 200 diverse systems around the world (FAO, 2015). The designated Pu'er Traditional Tea Agrosystem of China recognises the agro-biocultural diversity of (pollinator-dependent) wild tea tree populations, together with tea plantations that rely on traditional multi-layered forest cultivation methods of the Blang, Dai, Hani and other minorities, and their local institutions that protect the ancient plantations[16,17]. The designated Lemon Gardens of Southern Italy recognises the unique pergola-growing that produces distinctively flavoured

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[13] http://www.unesco.org/culture/ich/en/convention

[14] http://whc.unesco.org/en/conventiontext/

[15] http://www.fao.org/giahs/giahs-home/tr/

[16] http://www.fao.org/giahs/giahs-sites/asia-and-the-pacific/puer-traditional-tea-agrosystem-china/en/

[17] http://whc.unesco.org/en/tentativelists/5810/

high-value (pollinator-assisted) lemons grown in small farms that rely on traditional intensive labour systems[18].

The Representative List of Intangible Cultural Heritage has recognised the "Argan practices and know-how concerning the argan tree (*Argania spinosa*)" from Morocco as globally significant. This cultural heritage relies on insect-pollination success to produce a fruit with diverse forms that is harvested, dried, pulped, ground, sorted, milled and mixed to derive an oil used in cooking, medicines and cosmetics, relying on traditional knowledge of recipes and tools (Bani-Aameur and Ferradous, 2001). Other listed Intangible Cultural Heritage that rely on successful pollination of particular fruits include Kimjang, making and sharing kimchi in the Republic of Korea (chillies)[19]; and Washoku, traditional dietary cultures of the Japanese, notably for the celebration of New Year, relying on pollination of a diversity of crops (vegetables and edible wild plants)[20].

Several Cultural Landscapes on the World Heritage List rely on pollinators and pollination and their interactions with humanity. In the *Coffee Cultural Landscape of Colombia*[21], coffee production is linked to their traditional landownership and the distinctive small farm production system (Winter, 2015). The Landscape forms a corridor that connects different forest fragments, with diverse herbaceous and shrubby plants providing habitat with food sources, nesting sites and protection for resident and migratory animals, including 230 species of birds and 50 species of bees (Botero et al., 1999; Jaramillo, 2012) (Figure 5-16.). The stingless bees *Paratrigona eutaeniata* and *P. lophocoryphe* build their nests on the branches of the coffee trees, and are known as "angelitas del café" (little angels of coffee). Native bee communities within shade coffee farms ensure against the loss of introduced honey bees (Winfree et al., 2007), increase coffee yields (Klein et al., 2003) and maintain the reproduction and genetic diversity of native trees (Jha and Dick, 2010; Nates-Parra and Rosso-Londoño, 2013).

52. Figure 5-16. Colombian coffee landscape in the Risaralda Department.
© Catalina Gutiérrez Chacón, Reproduced with permission.

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[18] http://www.fao.org/giahs/giahs-sites/europe/lemon-gardens-southern-italy/detailed-information/en/
[19] http://www.unesco.org/culture/ich/RL/00881
[20] http://www.unesco.org/culture/ich/RL/00869
[21] http://whc.unesco.org/en/list/1121

Other pollinator-dependent World Heritage sites include the *Classical Gardens of Suzhou*[22] which celebrates the Chinese traditions of gardens that mimic natural processes with many flowering plants. The dense forest of the *Osun Sacred Grove*[23] on the outskirts of the city of Osogbo, is protected by Yoruba peoples as the abode of the goddess of fertility Osun, without whose involvement plants do not bear fruit and rains do not fall (Probst, 2011; Onyekwelu and Olusola, 2014). The *Agave Landscape and Ancient Industrial Facilities of Tequila*[24] in Mexico recognizes the biocultural diversity of the plant used since at least the 16th century to produce tequila spirit and for at least 2,000 years to make other fermented drinks, fibre and cloth. Tequila production today relies on clones from offshoots of mother plants, which is believed to be facilitating rapid spread of diseases due to the crop's low genetic variability (Torres-Moran et al., 2013). Efforts at controlling the disease organisms and vectors have achieved limited success, and attention is now focusing on traditional management practices that produce *Agave* spp. landraces with high genetic diversity, relying on bats for pollination (Dalton, 2005; Zizumbo-Villarreal et al., 2013; Tlapal Bolaños et al., 2014) (Figure 5-17.). Indigenous farmers have selected plants with desired traits from diverse individuals, producing at least twenty different land-race, and continue to use wild agave supporting ongoing diversification (Arita and Wilson, 1987; Colunga-GarciaMarin and Zizumbo-Villarreal, 2007; Zizumbo-Villarreal et al., 2013; Trejo-Salazar et al., 2015).

53. Figure 5-17. Bats (Leptonycteris sp.) pollinating Agave sp. flowers.
© Rodrigo Medellín. Reproduced with permission.

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[22] http://whc.unesco.org/en/list/813

[23] http://whc.unesco.org/en/list/1118

[24] http://whc.unesco.org/en/list/1209

5.3.3 Identity values and pollinators (socio-cultural valuation)

Pollinators feature as symbols that identify nation-states, indigenous nations, tribes and other communities throughout the world (Kristsy and Cherry, 2000; Werness, 2006; Dell, 2012).

The New Year festival of the Jewish religion, Rosh Hashanah, celebrates the creation of humanity in the Garden of Eden and is marked by eating honey cake, or apples dipped in honey which symbolizes the aspiration for a sweet future year (Goodman, 1970). Honey bees are the state insect of Utah, and are of profound importance to the Mormon culture, symbolising the industry, harmony, order and frugality of the people and the sweet results (Dickason, 1992) (Figure 5-18.).

54. Figure 5-18. Bees hive symbol on road signs and in front of Utah State Capitol building, United States of America.
© Gretchen Le Buhn. Reproduced with permission.

The beehive is a symbol of industry, perseverance, thrift, stability, and self-reliance.

The hummingbird (***Trichilus polytmus***) is the national symbol of the island Jamaica (Bigley and Permenter, 2009) (Figure 5-19. A). Many different indigenous tribes in the United States of America (USA) use hummingbirds in myths or legends (Bastian and Mitchell, 2004). For example, Hopi and Mojave Creation myths say that a hummingbird guided the people from their underground kingdom to light and taught them to make fire (Courlander, 1971; Mullett, 1979; Leeming and Page, 2000). Taino Indians, the indigenous people of Puerto Rico (Borikén) believe hummingbird is a noble warrior, teacher and sacred pollinator who brings new life (Jatibonicu Taino Tribal Nation of Borikén, 2015). The crimson sunbird (*Aethopyga siparaja)* is the national bird of Singapore (Minahan, 2010). The National Flower of Mauritius is *Trochetia boutoniana,* a rare endemic that produces a coloured nectar that attracts its lizard pollinator, the Mauritius Ornate Gecko *Phelsuma ornata* (Hansen et al., 2006) (Figure 5-19. B).

55. Figure 5-19. Hummingbird (*Trichilus polytmus*), the National Symbol of Jamaica and the National Flower of Mauritius (*Trochetia boutoniana*) with is pollinator Mauritius Ornate Gecko *Phelsuma ornata.*

A. Stamps celebrating the national symbol of Jamaica. © Unknown.

B. *Phelsuma cepediana* nectar-feeding at *Trochetia blackburniana*, the National Flower of Mauritius. Picture on the right shows the gecko preferentially feeding on the coloured nectar supplied by this unusual flower. Source: Hansen et al. (2006). Reproduced with permission.

Butterflies are also commonly used as symbols of nations and states, and in festivals across the globe (Howse, 2010). The endemic birdwing butterfly *Troides darsius* is recognised as the national butterfly of Sri Lanka (van der Poorten et al., 2012). Twenty-three states in the United States of America have butterflies as their state insects, commonly the monarch, which is also used in corporate symbols. In Africa, the Bwa of Burkina Faso dance at agricultural festivals wearing huge butterfly masks, up to seven feet across, with circles and designs representing the markings on the wings, to symbolise fertility and new life brought by the first rains (Wheelock and Roy, 2007) (Figure 5-20.). Celtic culture in Europe uses butterflies as symbols of rebirth and transformation in contemporary culture shamanistic practices and Celtic designs in diverse crafts, including body tattoos (Pearce, 1996; Conway, 2001).

56. Figure 5-20. Bwa butterfly plank mask. Wood, paint and rafia.
© Christopher D. Roy. Reproduced with permission.

The butterfly (horizontal) mask is danced in a festival, and symbolises the life-giving power of nature. Bumble bees have symbolic significance among many north and central American peoples: the Chiricauhua Apaches have a myth that bumble bees preserve fire in their home in a yucca stalk; Shasta people tell of bumble bees surviving the flood (Farrand and Frachtenberg, 1915; Olper, 1942). The Nadaco (or Anadarko) tribe from eastern Texas are named Nadá-kuh meaning "bumble bee place" (Fogelson and Sturtevant, 2004) and the Hohokam had a 'Bumblebee Village' (Ferg et al., 1984). For Thalhuicas (Pjiekakjoo) people in Mexico, bumble bees themselves symbolise the ancestors' souls that appear around the day of the death to visit their families (Aldasoro, 2012).

5.3.4 Aesthetic values and pollinators (socio-cultural valuation)

Pollinators are valued indirectly via their link to insect-pollinated plants, particularly those with showy flowers such as orchids, roses, sunflowers and many others that are aesthetically important as components of landscapes, vistas, gardens or parks (Hochtl et al., 2007; Schmitt and Rakosy, 2007; Wratten et al., 2012) (Figure 5-21.). Traditional European agricultural landscapes with flowering plants are also highly regarded for their cultural values (Reif et al., 2005; Rusdea et al., 2005). In Switzerland, studies have shown that people favour improving and creating field margins as habitat for species, landscape diversity and aesthetic value, and also showed marked preference for the season when plants are flowering (Junge et al., 2009, 2015).

57. Figure 5-21. Gardens for pollinators.
A: *Bombus* spp. in Oxford Gardens. © Berta Martin-López. Reproduced with permission.

B: *Bombus* spp. in gardens of the Colombian Andes (La Calera, Cundinamarca). © Guiomar Nates. Reproduced with permission.

C: The BEE-UTIFUL Gardens at Lake Merritt, California.

Traditional European beekeeping apiaries and their protective structures also add aesthetic value to the landscapes. Apiaries are built in specific areas in order to protect bees from cold, heat, wind and predators. In Slovenia, little wooden houses that protects bees are painted with pictures, so that bees can find them more easily, and to help the beekeeper distinguish hives and remember which colonies had already swarmed. The picturesque images depicting historical events, Bible stories, and everyday village life, enrich the cultural Slovenian heritage, transforming the landscape into an outdoor art gallery (Rivals 1980, Beattie, 2006). The Museum of Ancient Beekeeping in Lithuania, in the Aukštaitija National Park, celebrates the God of bees Babilas and the goddess Austėja from Lithuanian mythology and is surrounded by wooden sculptures representing the mythology of the origin of the bee in different cultures: Egyptians, American Indians and Lithuanians (Association of Lithuanian Museums, 2014). In Southern Europe, especially in France and Spain, it is common to meet specific apiaries, called *mur à abeilles* (bee-walls) directly constructed in a rock wall or protected by an enclosure in the landscape. Similar beekeeping apiaries are found in other European countries, especially those where rock is frequently used for human constructions (Mestre and Roussel, 2005).

5.3.5 Livelihoods of indigenous peoples and local communities — income, foods and medicines (holistic valuation)

Pollinators, primarily bees, provide a source of income, food and medicines that are vital to the livelihoods of many indigenous peoples and local communities globally (Gupta et al., 2014). Beekeeping provides a critical anchor for many rural livelihoods: minimal investment is required; diverse products can be sold; land ownership or rental is usually not necessary; family nutrition and medicinal benefits derive; timing and location of activities are flexible; and links to ILK and traditions are usually numerous (Hilmi et al., 2011). Recovery of stingless beekeeping with diverse hives and techniques is currently underway across central and South America (Case example 5.1313, Figure 5-22.).

Case example 5.13. Recovery of stingless beekeeping for sustainable livelihoods in Latin America.

Location: Mexico, Colombia, Brazil.
Diverse indigenous peoples and local communities across Latin America

Stingless beekeeping probably represents one of the best examples of a sustainable practice that is slowly recovering from a reduction in some areas of Mesoamerica to a thriving activity nowadays, practiced by various indigenous groups in Central Mexico, Colombia and Brazil.

Across the Americas, detailed identification systems of stingless bee species, their biology and behaviour is part of the knowledge of the Maya and Nahuas groups in Mexico and Guatemala, in the Brazilian Amazonia (by the Gorotire-Kayapo, Ticuna, Cocama and Mura) and the Midwestern, Southeastern and Northeastern Brazilian regions (Guarani M’Byá, Kawaiwete, Enawene-Nawe and Pankaraé), in Ecuador (Cayapa) and the Colombian tropics (Andoque, Eastern Tukano (Siriano and Bará) and Nukak) and temperate regions (the U´wa) (Posey 1983b, a; Camargo and Posey, 1990; Costa-Neto, 1998; Cabrera and Nates-Parra, 1999; Quezada-Euán et al., 2001; Rodrigues, 2005; Ballester, 2006; González-Acereto et al., 2006; González-Acereto et al., 2008; Santos and Antonini, 2008; Rosso-Londoño, 2013).

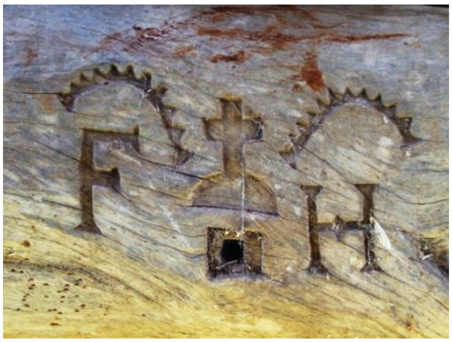
Recently partnership efforts led mainly by academics and universities have been reviving and strengthening stingless beekeeping, bringing science and tradition together. Several modern techniques and innovations have been developed to maintain and reproduce colonies efficiently, to improve the quality and marketability of products and also by starting to use colonies for services such as commercial pollination. Stingless beekeeping is showing signs of recovery for various indigenous groups of Argentina, Bolivia, Brazil, Colombia, Ecuador, Mexico, and Venezuela and people outside these communities are also getting involved in stingless beekeeping and commercialization of products.

Key elements for the recovery of stingless beekeeping have been: teaching and extension work, respect for their local costumes and traditions, increased value of products, and development of a market niche for stingless bee products. Key elements for the recovery of stingless beekeeping in the Yucatan and Brazil have been: teaching and extension work, respect for their local costumes and traditions, increased value of products, and development of a market niche for stingless bee products (González-Acereto et al., 2006; Jaffe et al., 2015).

58. Figure 5-22. Stingless beekeeping in Central and South America.

A. Jobones (Meliponaries, stingless beehives), in Mexico. © Javier Quezada-Euán reproduced with permission.

B. Mayan family with jobones in Mexico. © Javier Quezada-Euán Reproduced with permission.



C. Meliponarie Nahua (*Scaptotrigona mexicana*) in earthenware pots, Sierra Norte de Puebla, Mexico © Javier Quezada-Euán Reproduced with permission.

D. *Melipona favosa* nests in earthenware pots in Guanare, Venezuela. © Guiomar Nates Parra. Reproduced with permission.

E. Different kinds of nests for stingless bees in Colombia. © Guiomar Nates Parra. Reproduced with permission.

F. Stingless beekeeping in Northeast of Brazil. © Juan Manuel Rosso. Reproduced with permission.

Traditional honey-hunters in India organise to send their honey to a local tribe cooperative where it is sold for medicinal properties, as well as using it themselves. Prayers and rituals accompany these harvests, linking the customary and market economies (Barlagne et al., 2009). Ethiopian farmers have developed beekeeping as a good source of income, through multiplication and selling of honey bee colonies in the local market as domesticated animals (Adgaba, 2000). Local people in Kechifo, Ethiopia both trade white honey for both cultural and economic purposes (Avril, 2008). Many communities in Africa keep bees for the direct economic benefit of selling honey and other honey bee-derived products (Adjare, 1990), and also appreciate and value bees as a long-term means towards to improve household food and nutritional security (Villières, 1987; Fischer, 1993; Sanginga, 2009).

Beekeeping has improved rural household nutrition in many subsistence farming communities across Africa (Wilson, 2006; Martins, 2014) and is used to make honey beer (Adgaba et al., 2008). In Nigeria in both rural and peri-urban settings household nutrition is improved through beekeeping (Azeez et al., 2012). Collection and harvesting of honey occurs across sub-Saharan Africa by: the Abayanda of Uganda (Byarugaba, 2004); Batwa and other pygmy peoples in the Congo Basin forests (Crane, 1999; Kajobe, 2007; Kajobe, 2008); the Hadza in Tanzania (Marlowe et al., 2014); the Ogiek in Kenya (Rambaldi et al., 2007); and by nomadic pastoralists in Somalia and other regions of the Horn of Africa (Tremblay and Halane, 1993). In Australian Aboriginal societies, stingless bee honey (sugar-bag) is a popular food (Fijn, 2014).

Case example 5.14. Honey beer and honey wine.

Location: Zambia and Ethiopia

Honey beer is important for multiple reasons in Zambia. It is taken during the initiation ceremonies when boys and girls reach mature age, during traditional chiefs' ceremonies and as payment for cultivating or harvesting fields. After a day's hard work, some people go to bed early and start drinking honey beer at

3am and by 6am are ready for hard manual work. Local communities warn the smell of the honey beer on people irritates the bees to attack, so you cannot work with bees. Honey beer cannot be stored for more than 48 hours (Malichi, 2009).

In Ethiopia, honey is made into Tej, honey wine. Tej is a very important drink in Ethiopian cultural life, served at traditional gatherings and special religious ceremonies. Tej is often it is drunk before the brew has started to ferment, when it still has a strong yeasty flavor. This drink is called birz and is popular with children and, being non-alcoholic, is acceptable to Muslims. Tej is made in huge wooden barrels, which are cleaned and then scoured with special leaves. The barrel is then filled, one part of honey with five parts of water and covered with a clean cloth and left for a few days to ferment. Gesho, leaves of *Rhamnus prinoides*, which have been chopped up and then boiled are added, stimulating sugars to convert to alcohol and the Tej increasingly acquires its distinctive dry and bitter flavor. Finally, just before serving, a further half bucket of honey is tipped in to give sweetness to the final brew.

Tej is served in special glasses called birrille, held in a special and rather dainty way between the first two fingers and thumb. In Africa it is usually women who brew beer, make Tej, and sell these products (Bradbear 2009).

Honey is also used as food for several tribes and local communities in Indonesia, such as Anak dalam tribe (Ibrahim et al., 2013), Sakai tribe (Suparlan, 1995), Petalangan people (Titinbk, 2013) and Kelay Punan tribe (Widagdo, 2011). Crane (1999) recorded that native people in other Southeast Asian countries such as Vietnam (Annam people), Cambodia, Malaysia, Thailand (Lao people), Myanmar (Burmese people) also used bee products as food. In Thailand, people believe that consuming honey and brood will have a good impact on their health (Chantawannakul et al., 2011). Flying foxes are recognised as a vital pollinator and also a delicate and very popular dish in Vanuatu and Fiji (Palmeirim et al., 2007).

For indigenous communities from South America (Andoque, Guaycurúes, U'wa, Yuquí, Toba-pilagá, Tukano), stingless bees are part of their cosmogony and mythology and important as nourishment and to obtain products used in the elaboration of alcoholic beverages, instruments and handicrafts (Ruddle, 1973; Jara, 1996; Cabrera and Nates-Parra, 1999; Falchetti and Nates-Parra, 2002; Arenas, 2003; Falchetti, 2003; Stearman et al., 2008; Medrano and Rosso, 2010; Zamudio et al., 2010; Zamudio and Hilgert, 2011; Estrada, 2012; Zamudio and Hilgert, 2012; Nates-Parra and Rosso-Londoño, 2013; Rosso-Londoño, 2013). Stingless bees' honey is greatly valued for its medicinal properties, e.g., antibiotic and antibacterial properties, especially with *Tetragonisca angustula* honey (called angelitas, rubitas, señoritas) in Andean countries (Posey, 1983b, a; Estrada, 2012; Fuenmayor et al., 2013; Rosso-Londoño, 2013; Vit et al., 2013; Zamora et al., 2013) and *Melipona beecheii, Trigona nigra, Cephalotrigona zexmeniae, Frieseomelitta nigra, Scaptotrigona hellwegeri, Melipona fasciata* and *Geotrigona acapulconis* in Mexico and Central

America (Quezada-Euán, 2005; Ocampo-Rosales, 2013; Reyes-González et al., 2014). In the Misiones province (Argentina) research has focuses on the usage of stingless bee products and plants of the region in traditional medicine, giving also relevance to different names given to bees by the local communities (Zamudio and Hilgert, 2011; Zamudio and Hilgert, 2012).

Honey has been used for medicinal purpose by many societies, such as the Mayan, for millennia (Ocampo-Rosales, 2013). In Polish traditional medicine, for example, honey has been a popular remedy to treat respiratory diseases, gastrointestinal disorders, dermatological problems, heart disorders and for contagious diseases (chickenpox, measles). Different mixtures suit different purposes—to treat cold and flu, honey, butter and garlic are added to hot milk or vodka; to treat contagious diseases, like measles, lacto-fermented cabbage juice is mixed with whey, honey and fat. Local communities in Argentina of Polish and multiethnic populations now distinguish honey from seven different *Hymenopteran* ethnospecies to treat respiratory, dermic, osteo-artomuscular, nervous, digestive and circulatory disorders (Zamudio et al., 2010). Honey has been found to be more important as a medicine than a food for local peoples in Brazil and Mexico (Ramos-Elorduy et al., 2009). In Ethiopia, wild honey is usually consumed without filtration, still including wax, pollen, and royal jelly, constituents that strengthen its nutritional properties (Avril, 2008). The Pankararé from Brazil uses honey, pollen and wax as medicine, and use specific honey from different species of stingless bees to treat specific diseases; 11 species provide 13 raw materials used to prepare remedies to treat or prevent 16 illnesses (Costa-Neto, 1998).

Honey is very widely used in traditional medicine in Africa. It can be used alone or in combination with medicinal plants to treat numerous pathologies, especially those concerning respiratory tracts or dermatologic problems, fever and traumas. Honey has been widely used in Africa to help with the healing of wounds (Armon, 1980) and other ailments (Manyi-Loh et al., 2011), with recognition of its anti-microbial properties being linked to the plants that the honey bees foraged on (Basson and Grobler, 2008). This is a value appreciated by many communities in the Greater Horn of Africa region, where bitter honeys that result when honey bees forage on certain plants, including succulent euphorbias and *Commiphora* spp. in drylands, are especially useful for treating infected wounds and other skin problems (El-Kamali, 2000). This usage of honey for treating wounds is also widely employed among pastoralists in this region for treating their livestock (Gakuya et al., 2010). In some local communities, for instance from South of Morocco, each kind of honey has special therapeutic indications (Crousilles, 2012; Simenel, 2015). Local people in Maningri, Benin report many medicinal uses of honey (Yédomonhan and Akoègninou, 2009). Several communities in Africa make use of the honey bees themselves for medicines. For example in Burkina Faso both honey and honey bee brood (larvae) are widely used to treat a range of ailments (Meda et al., 2004). Analyses of honey used by the Hadza people in Tanzania has shown that it does have higher protein, fat and ash content that is thought to be related to the inclusion of bee brood when harvested/consumed (Murray et al., 2001).

For Petalangan people in Indonesia, bees are seen as a symbol of health and cheap sustenance (Titinbk, 2013). Many indigenous peoples across Asia use honey as a medicine, mixing or cooking the honey with other ingredient. For example the Siddhi tribes used *Momordica charantia* leaf juice together with few drops of honey as cough medicine, and for congestion and chest pain for children (Joseph and Antony, 2008). Local people in Kalla Chitta of Pothwar region in Pakistan used a decoction of *Cicer arietinum* (chick pea) fruit mixed with honey to relieve abnormal menstruation and throat pains. Honey is also used by these people to relieve other pains such as chronic flu, sunstroke, antidiabetic and chronic constipation (Arshad et al., 2014).

5.3.6 Social relations: song, dance, art, story, rituals and sacred knowledge about pollinators

Indigenous peoples and local communities value pollinators through texts, song, dance, art, religious and spiritual knowledge, and revelations (Case example 5.15). Stingless bees are also present in popular songs and in the Brazilian imagary (Souza et al., 2013). Near Pedu Lake, in the Kedah province of northern Malaysia near the border with Thailand, honey hunters chant ancient prayers as they gather honey from giant tualang (*Koompassia excelsa*) trees (Buchmann and Nabhan, 1996). The Burmese and Thai people believe that if bees move to their house, it is a sign of luck and prosperity (Chantawannakul et al., 2011).

The O’odham people from the Sonaran Desert of southern Arizona and northern Mexico have a song about the intoxicating effects of thornapple *(Datura* sp.) alkaloids on nectar-feeding hawkmoths (*Manduca* spp.), first recorded in 1901, although undoubtedly of much greater antiquity. The real value of such songs is highlighted by scientific investigations to understand this intoxication, which was ‘discovered’ by science in 1965, challenging theory about the level of alkaloids in nectar (Nabhan, 2000).

Case example 5.15. Valuing pollinators through song and ceremonies.

Location: Indonesia and Philippines
Palawan and other indigenous people

The Palawan people (Philippines, Upland Palawan) pass on knowledge about the stinging bee (*Apis florea* or *Apis cerana indica* called *mugdung Nigwan* or *tämaing*) and stingless bee (*Trigona* ‘sensu lato’ probably called *kätih kätih*) through ceremonies. Both *tämaing* and *kätih kätih* are associated with many myths, legends, rites, and others ceremonies. They have specific rituals requesting god (*ampuq),* to allow flowering and blossoming of the flowers to take place, then invite the bees to come and build nests and produce honey.

Songs are always sung to pass on knowledge while harvesting honey in the East coast of North Sumatra (Hadisoesilo and Kuntadi, 2007). The first song is sung before climbing a tree to introduce oneself to the tree and the spirits in that tree. The second song mollifies the bees in order for them to become gentle and provide larger quantities of honey. In Danau Sentarum National Park, West Kalimantan, climbers sing mantras at different stages of the honey collection (Hadisoesilo and Kuntadi, 2007). When the ladder is ready, they welcome its strength. Once on the branch, while smoking the bees, they sing again to appease the spirit of the tree, and when cutting the comb, they welcome the upcoming harvest. Once honey is harvested, they ask their ancestors to protect the basket in its descent. One last song marks the end of the harvest, the final descent of the climbers and the return to the village (Césard and Heri, 2015). Parts of the lyrics are improvised, not without humor (often as honey alludes to a beautiful young woman and to her charms) (Mulder et al., 2000).

The Ikpeng group in the Brazilian Amazon sing a song of a bee to avoid thunder during storms. They say that this song is very dangerous and should not be sung when there are no storms (Athayde, 2015). Ogiek songs and prayers relay all the knowledge about how to care for the forest; learning is in the circle of life (Samorai Lengoisa, 2015).

Pollinators in ancient and modern Mesoamerican civilization have divine affiliations. For instance hummingbird feathers were believed to be the seed from which a major deity among the Aztec was born, the war god *Huitzilopochtli* (Spence, 1913 [2010]). Today hummingbirds are seen as sacred creatures capable of communicating with the gods (Figure 5-23.). Similarly, bats were seen as messengers from the underworld and symbols of fertility (Retana-Guiascón and Navarijo-Ornelas, 2012). Ancient Mayan rituals in relation to bees have continuity with today’s requests for the protection of hives, of a good honey harvest and good fertility in the flowers that feed the bees. These rituals support continuity in production, consumption and offering of drinks sweetened with honey (sacá and balché) that are also given to birds that are sacrificed (González and Noguez, 2009). Stingless bees are part of the cosmogony and mythology, being of similar importance to the cultivation of maize, the staple food for Mesoamerican civilizations (de Jong, 2001). Within the mythology U’wa (Sierra Nevada del Cocuy, Colombia), bees are considered important as the beings that made possible the gestation of the life and natural light in the universe, and honey is associated with purity, vitality, strength, fertility and procreation (Falchetti and Nates-Parra, 2002). Lima and Moreira (2005) report that the Tupinambás people in Brazil associate stingless bees with their cosmology, and name constellations with bee names.

59. Figure 5-23. Hummingbirds, pollinators with divine affiliations.
A: The Spine Peak Blackback (*Ramphomicron dorsale*), endemic species from the Sierra Nevada de Santa Marta, Colombia. © Proaves, Alonso Quevedo. Reproduced with permission.

B: Humming bird *Eriocnemis mirabilis.* Endemic Bird of the cloud forest of the Pacific slope of the Cordillera Occidental de Colombia. Series Stamps: Biodiversity endemic of Colombia in danger of extinction. Issued in 2015. Reproduced with permission.

5.3.7 Governance by, with and for pollinators and their spiritual presences among indigenous peoples and local communities (holistic valuation)

Governance has been defined as:

> the interactions among structures, processes and traditions that determine how power and responsibilities are exercised, how decisions are taken and how citizens and other stakeholders have their say (Graham et al., 2003).

In many ILK systems, these interactions place pollinators in key roles with ultimate authority for governance. Pollinators including birds, bats, butterflies, bees and other insects feature as spiritual presences and symbols of authority amongst indigenous peoples and local communities across the world (Kristsy and Cherry, 2000; Werness, 2006). Pollinators' spiritual and symbolic significance and authority in social organization is well documented amongst Native Americans, on both northern (Sturtevant, 1978; Fogelson and Sturtevant, 2004) and southern continents (Case example 5.16).

Case example 5.16. Social organization of bees as a model for human society among pre-Columbian Maya people.

Location: Mesoamerica

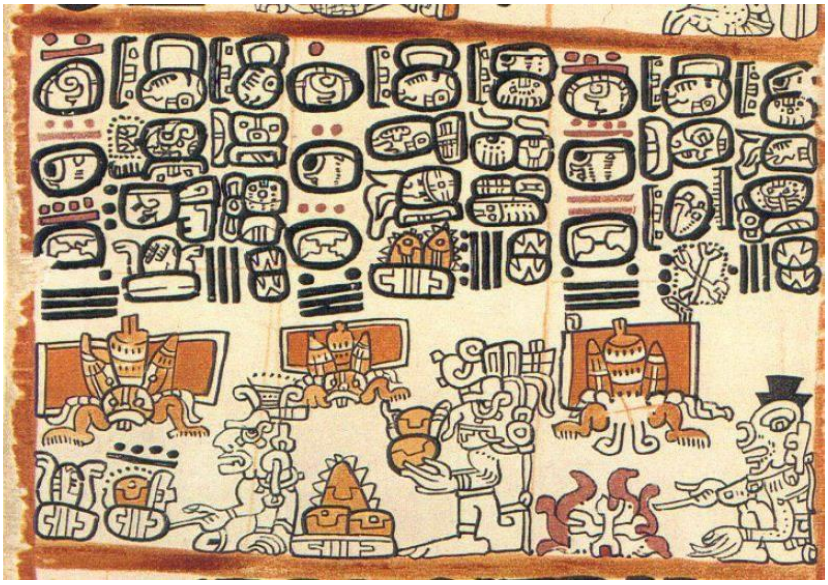
Mayan peoples

Evidence of the economic and religious importance of the bee *Melipona beecheii* (*Xunan-Kab*) is seen in the various manifestations of Mayan art that have reached us. The most important is the codex of Madrid, one of the three surviving Maya codices, in which stylized images of *Xunan Kab* bees and their guardian gods are represented in various scenes probably associated with the harvest of the honey and colony multiplication (pages 103-112). Some of these deities are *Ah-Mucen-Kab* (the descending honey god), *Noh Yum Kab*, *Hobnil*, *Balam-Kab* and *Moc-Chí* (Figure 5-24. A and B). All of them are represented with a mixture of anthropomorphic and bee-like features, sometimes involving characteristics of other sacred animals like the jaguar (de Jong, 1999; Quezada-Euán et al., 2001) (Figure 5-24. C). The Mayan *Miatschahale*s (philosophers) used stingless bee (*Melipona beecheii*), as a model for adequate social organization as well as ecological and political ethics. Thus, several values and strategies are explicitly modelled on *Melipona beecheii*´s social organization. Among these are: cooperation and solidarity; adaptation to changes that occur outside the colony; optimization of the use of natural resources for the well-being of the group over individual well-being; avoidance of over-exploitation of natural resources; control of population size to adapt to variable conditions; prediction of droughts; and food security measures (López-Maldonado, 2010; López-Maldonado and Athayde, 2015).

60. Figure 5-24. Mayan Codex and art representing Xunan Kab (*Melipona beecheii*).
A: Ah Muzen-Cab, God protector of bees and the crops. © Luis A. Medina. Museo Palacio Cantón, Merida, Yucatán. Reproduced with permission.

B: Ah Mucen Kab by removing honey from a nest of stingless bees. Codice 104 Maya Itzá of Mayapán. © Juan C. C. Medina. Reproduced with permission.

C: Parts of the Mayan Codex and the bee (*Melipona beecheii*) ©.



Bee deities are important among ILK holders in Asia (Gupta et al., 2014; Césard and Heri, 2015). For example, Punan honey hunters in Borneo express the respect that they carry for bees by referring to them as "Hitam Manis", "Blooming Flowers" or "Fine Friends", and indicate their subservient relationship by referring to themselves as the Dayang, the handmaidens of Hitam Manis (Buchmann and Nabhan, 1996). Dressler (2005) presents great detail about the governance of the Tagbanua swidden-honey complex by spiritual presences of "bee deities" (Case example 5.17). Dressler (2005) recommends these Tagbuana knowledge and beliefs as the basis of involving Taganuan in management of the Puerto Princesa Subterranean River National Park, and recognition of their ancestral title.

Case example 5.17. Sinada, the bee deity, and ceremonies govern the swidden-forest-honey complex.

Location: Palawan Island, Philippines and Indonesia
Tagbanua, Palawan and Petalangan indigenous peoples

Tagbanua people of Palawan Island believe that the ultimate authority for their swidden-honey complex lies with two bee deities, *diwata* and *panya'en*, living in the forest and karst (towers, cliffs and ridges of limestone). Both spirits take the shape of bees, and among them is Sinada, the highest ranking bee deity. Communication with these spirits occurs through the babalyan (senior cultural leader, shaman) who conducts ceremonies and prayers that express hope and security to Sinada. "Sinada thus governed the social order and function of the bee kingdom while offering honey collectors strength and fortitude. *Sinada's* subordinate is the *panya'en*, *Ungao*, the creator and guardian of honey bees in Cabayugan. *Ungao* transmitted *Sinada's* "message of assistance" as laws instructing other spirits to influence the behaviour of honey bees. *Ungao* asked his subordinates to "convince" bees to build hives visibly and in permanent locations" (p. 25-26) (Dressler, 2005).

The Palawan people of Palawan island view bees (and their products) as something that needs to be negotiated through appropriate behaviour and ceremonies. They conduct the *Simbung* ceremony to ask the Gods for the flowering of trees. The Palawan people see that the decline of bees and their products will negatively impact on the ceremony, and on the skills, knowledge, and mythological connections and awareness of the next generation of Palawan people and vice versa (Novellino, 2002).

Amongst the Petalangan community, Indonesia, the rituals of bee-hunting have created social groups based on their functions during the collection process. The collector group, known as a *menumbai*, consists of several people with different roles and responsibilities. The *juragan tuo* is the coordinator of the harvesting team, usually someone who is older, with significant experience in harvesting honey, and substantial knowledge about the bees, their behaviour and the habitats of the trees. The *mudo* is an assistant to the *tuo*, always someone who is younger with less experience. The *juragan tuo* passes knowledge on to *juragan mudo* who will climb the trees, and *tukang sambut*, the receiver of the honey, at the bottom of the trees. The bee-hunting activities enhance cooperation supported through rituals and cultural ceremonies. Distribution of the honey is determined by membership of the social groups, with between 20-40% for the *menumbai group*/harvester (40-60% for the rest of the communities and 20% for the head of the village (Buchmann and Nabhan, 1996).

In Andean communities, the concept of "Buen vivir" values solidarity, community, freedom, respect for nature, responsibility and equality, and emphasizes the links between good governance and relations with nature, of the good life and the rights of nature (Fatheuer, 2011). These principles underpin the indigenous Potato Park, which is protecting genetic diversity and pollination-based reproduction associated with approximately 1,300 different varieties of potato (Argumedo and Pimbert, 2005). In the Siddhi tribes in Uttar Kannada (India), honey harvesting is valued for its social institutions that require and teach good teamwork among the harvesters (Kumsa and Gorfu, 2014).

Governance systems also recognize tenure, systems of ownership, over important pollination resources. In Indonesia, there are diverse rights associated with trees that have bees nesting on them (Césard and Heri, 2015). In Tesso Nilo National Park, Riau Province Indonesia, the local beekeeper association marks the coordinates of each honey tree (sialang) owned by their members. In Sumatra and Kalimantan, honey bee trees belong to the first person who found the trees and the ownership is inherited to the children. In

Dompu, Sumbawa, the trees are owned by the village authority, but after each harvest season, the trees are open for bidding. In Ujun Kulong National Park, West Java, there is no ownership of the trees and everyone is entitled to gather honey on any trees they find (Césard and Heri, 2015). Land tenure systems based on ILK are often complex, with overlapping rights enabling access to resources with sets of checks that contribute to ensuring that pollinators' resources and pollination resources are not over-exploited (Ostrom, 2003, 2005). For example, in the Cordillera of the Philippines tenure regimes include communal, corporate and individual lands (Prill-Brett, 1986, 2003).

5.4 Impacts, management and mitigation options

5.4.1 Risks to nature's benefits to people and good quality of life

The contribution of pollinators and pollination to nature's benefits to people and good quality of life, assessed through socio-cultural and holistic valuation approaches, are clearly very high (5.2, 5.3). Risks associated with pollinators and pollination therefore will potentially impact on these benefits and quality of life. Chapter 6 presents a summary of the risks and opportunities associated with pollinators and pollination (Table 6.2.1), and provides relevant responses organized across sectors. Here we focus on those risks most relevant to the instrumental and relational types of values of pollinators and pollination considered in this chapter (Tables 5-1 and 5-2). Four main risks will impact on these values (Table 5-3).

27. Table 5.3. Risks and impacts on values.

Risk	**Impacts on values assessed through socio-cultural and holistic approaches in this chapter**
Direct and indirect impacts on food crop production	Decline in human health and nutritional security due to less availability of crop plants that are major contributors to micronutrients, vitamins and minerals in the global human diet.
Direct and indirect impacts on honey production and bee numbers	Declines in rural economies that are anchored by beekeeping and honey hunting as livelihoods with many advantages (e.g., low investment, links with cultural institutions). Declines in educational and recreational benefits derived from beekeeping and honey hunting (e.g., as an intervention tool for youth criminal behaviour).
Loss of distinctive ways of life, cultural practices and traditions in which pollinators or their products play an integral part	Loss of nature's benefits to people from declines in pollination-dependent products used in medicines, biofuels, fibres, construction materials, musical instruments, arts and crafts. Loss of cultural services through declines in pollinators and pollination as sources of inspiration for art, music, literature, religion and technology.

	Declines in nature's gifts to Indigenous Peoples and Local Communities of pollination-promoting practices of valuing diversity and fostering biocultural diversity and of their diversified farming systems.
Loss of aesthetic value, happiness or well-being associated with wild pollinators or wild plants dependent on pollinators	Loss of good quality of life from declines in the availability of pollinators and pollination resources as globally significant heritage, as symbols of identity, as aesthetically significant landscapes, flowers, birds, bats, and insects, and for their roles in social relations and governance interactions of Indigenous Peoples and Local Communities.

Losses and declines in nature's benefits to people and good quality of life have been evident in the past as well as the contemporary context. For example, a pollinator extinction is associated with a cascade of impact on quality of life for Easter Islanders:

The Polynesian rat (*Rattus exulans*), which arrived on Easter Island with the Polynesians, may have caused the extinction of a parrot species that once pollinated a now extinct *Jubaea* palm (Van Tilburg, 1994; Diamond, 1995; Robert et al., 1998). The rats also destroyed palm and other tree seeds, diminishing the native forest until the Polynesians could no longer construct canoes for fishing; thus the subsequent cultural decline may be more of a result of pollination disruption to seedling recruitment than of human overexploitation of forest resources (Cox and Elmqvist, 2000).

Contemporary impacts of pollinator and pollination declines on nature's benefits and good quality of life are being highlighted by organisations such as Greenpeace[25], and National Geographic (Holland, 2013), and Time (Pickert, 2008), for example the loss of appreciation of the beauty of butterflies[26]. Wider issues of loss of aesthetic value of landscapes (Farber et al., 2006), and of inspiration for art, music, and literature are key concerns, reflected for example in the Faith Taskforce and publications of the North American Pollinator Protection Campaign[27] and the Sentimiel initiative of the Institut de recherche pour le développement[28].

5.4.2 Peoples' experiences of declines and associated drivers

People in many parts of the world have reported declines of pollinators and pollination. Chapter 2 provides a scientific assessment of the drivers of the change to pollinators and pollination, together with examples of contributions from ILK systems. Here we provide an overview of how people have experienced these declines, and the drivers of declines.

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[25] http://sos-bees.org/

[26] http://www.learner.org/jnorth/tm/monarch/ConservationValuesBrowerQA.html

[27] http://pollinator.org/nappc/index.html

[28] https://en.ird.fr/content/download/63580/513428/version/3/file/excellence_in_research_2012.pdf

People's experiences are associated with environmental, socio-economic and cultural change including: habitat loss, fragmentation and degradation; pesticides and herbicides; changes to and loss of bee management practices and knowledge; loss of access to traditional lands; changes to and loss of bee management practices; loss of access to traditional estates; changes to and loss of traditional knowledge, tenure and governance systems that protect pollination; and pollination governance deficits. Often the decline of pollinators and the decline of ILK systems occur simultaneously as a result of the expansion of agriculture and commodity extraction frontiers, and associated habitat loss and territorial acquisition (Reyes-García et al., 2014b).

5.4.2.1 Habitat loss, fragmentation and degradation

Many peoples' experiences of pollinator and pollination declines are associated with habitat loss and degradation, including replacement of biodiverse habitat with monocultures (Athayde, 2015). Co-production between science and ILK is strengthening understanding of these declines, for example identifying how declining bird populations associated with transformation of traditional shaded coffee agriculture to simplified systems with fewer trees or treeless monocultures, referred to as sun coffee, result from this destruction of wintering habitat for millions of migratory birds (Perfecto et al., 2014). Guna people have noticed the disappearance of both a hummingbird that pollinated hibiscus flowers, and the hibiscus flower itself, the syrup of which was formerly used as a drink by pregnant women (López et al., 2015).

In Brazil, the agricultural frontier expansion is putting pressure on both demarcated indigenous lands and other forests, driving a "containment" of bee populations in smaller forest fragments (Villas-Bôas, 2015). The Kechifo people from Kafa (Ethiopia) harvest three types of honey, each associated with a particular plant, and consider one of them, white honey, as a marker of biodiversity decline — white honey disappears with the introduction of monospecific crops of coffee trees (Verdeaux, 2011). In Kodagu (India), once famous for abundant honey projection, intensification of coffee plantations has reduced populations of melliferous plants, particularly *Litsea floribunda*, to such an extent that honey production is now only symbolic (Barlagne et al., 2009). Honey hunters in India note both forest fires and forest loss as causes of declines in honey availability (Demps et al., 2012a). Honey-harvesters in Sentarum Lake, Indonesia report that smoke coming from the deforestation for plantations has a direct negative impact on the arrival of the swarms in season and therefore on honey production (Césard and Heri, 2015). Degradation of habitat extends to direct impact on pollinators, such as through over-hunting of large flying foxes (*Pteropus vampyrus natunae*) in Central Kalimantan, Borneo, Indonesia (Struebig et al., 2007).

5.4.2.2 Pesticides and herbicides

Pesticides have also been associated with declines. Beekeepers in the United States of America (USA) have reported wide-spread deaths of honey bees, and the phenomenon termed colony collapse disorder (CCD) (Suryanarayanan and Kleinman, 2013). While the US beekeepers' perspectives on the causes of CCD are heterogeneous, several commercial beekeepers with decades of migratory beekeeping experience claim experiential and practical knowledge that CCD is caused by proximity of their hives to agricultural crops treated with neonicotinoids such as imidacloprid. Beekeepers in Europe and France have similarly attributed colony losses to this same group of insecticides (Suryanarayanan and Kleinman, 2014; Suryanarayanan, 2015).

Beekeepers in Burkina Faso note a direct link between increased cotton production and declines of honey, which they similarly attribute to pesticides (Gomgnimbou et al., 2010). Sichuan pear producers in Hanyuan County in China have adopted hand-pollination as insect pollinators have disappeared due to the use of herbicides and pesticides (Ya et al., 2014). In Korea, one survey of traditional beekeepers found that 94.7% had experienced damage to their bee colonies from pesticides, and considered pesticides the most critical problem in apiculture, one that they cannot escape (Choi and Lee, 1986; Park and Youn, 2012). Honey hunters in India related declines in honey to pesticides on coffee estates (Demps et al., 2012a).

Mbya Guaraní, peoples from the Paraná State of Brazil have noted pollinator declines associated with use of pesticides (Cebolla-Badie, 2005). Tūhoe Tuawhenuaare are concerned about many chemical residues posing a threat to pollination and pollinators, and through co-production with science have identified that the pesticide '1080' is taken up into their medicinal plants, with unknown effects (Doherty and Tumarae-Teka, 2015).

5.4.2.3 Changes to and loss of bee management practices and knowledge

A recent global review across Mexico, Costa Rica, Brazil, Africa, Australia and Asia found that stingless beekeeping is disappearing in some areas, such as the Yucatan. In other places, such as Brazil, meliponiculture is increasing as an important secondary economic activity (Cortopassi-Laurino et al., 2006). The traditional use of stingless bee products in medicine and handcraft is also declining (Sterman et al., 2008; Roig Alsina et al., 2013). In Colombia, stingless beekeeping practices are being challenged by loss of local names, abandonment of hives due to mismanagement, and homogenization and standardization of bee species and beekeeping techniques (Rosso-Londoño, 2013). The disappearance of stingless beekeeping from indigenous communities is problematic (Villanueva-Gutiérrez et al., 2013), as it may represent a threat to the survival not only of various native bee species but also to the sustainability of the ecosystems due to their contribution as pollinators and also to ancient medicinal and cosmological traditions, and other cultural aspects (González-Acereto et al., 2006). Some species of stingless bees like *Melipona beecheii* in the Yucatan find their most important populations in the hands of Mayan farmers, as large trees from the central Yucatan have disappeared, resulting in the absence of feral colonies of this species in such areas (González-Acereto et al., 2006). The survival of *M. beecheii* in the Yucatan strongly depends on the continuity of stingless beekeeping.

Stingless beekeeping decline is affected by multifactorial trends, involving ecological, social and economic drivers, such as the greater commercial returns from the introduced honey bee (*Apis mellifera*) (Cortopassi-Laurino et al., 2006). Loss and decline of the stingless bees in also linked with a loss of traditional knowledge and practices, including cosmogony and ethnomedicine, and associated loss of biocultural diversity (Joshi and Gurung, 2005; Ngima Mawoung, 2006; Freitas et al., 2009; Corlett, 2011; Césard and Heri, 2015; Samorai Lengoisa, 2015; Villas-Bôas, 2015). Key bottlenecks to increasing stingless beekeeping include how to collect and conserve their honey, how to rear them in large quantities, how to prevent impacts from pesticides and maintain the bees, and how to provide qualified information and training in all levels (Cortopassi-Laurino et al., 2006). Co-production between ILK and science is proving effective in overcoming some of these challenges (Case example 5.1312).

Traditional beekeeping knowledge and practices are also declining in Europe. For example, in Sicily the "férula" hive is known to be strong and not expensive, but was progressively replaced with frame hives, and traditional knowledge such as the "partitura" used by Sicilian beekeepers to recognize an artificial swarming is also declining (Roussel, 2009).

Honey hunting among forest-dwelling communities who hunt at low levels in Kenya, Indonesia, Nepal, India, Brazil and Cameroon and practice non-destructive methods supports protection of pollinators and pollination resources (Joshi and Gurung, 2005; Ngima Mawoung, 2006; Rosso-Londoño, 2013; Césard and Heri, 2015; Samorai Lengoisa, 2015; Villas-Bôas, 2015). However a large rise in unsustainable honey hunting is now posing a significant threat to stingless bees in Asia (Corlett, 2011) and the neo-tropics (Freitas et al., 2009). The demand for wild nests to deliver honey, resins and cerumen for food, medicines and other products has led to honey hunters now being targeted as one of the main causes of loss of bee colonies and of destruction of habitat trees. However, Rosso-Londoño’s (2013) socio-environmental analysis identified that there are now many other stakeholders, including stingless beekeepers, research and government institutions, and industry, because markets and new projects (for production, education, hobby and even research) are part of the context that is driving the demand for wild nests. Among Indonesian honey hunters, changes are occurring at the social-cultural level and interacting with environmental change. For instance, Anak Dalam people in Sumatra are using honey as an exchange value (non-monetary) to buy other necessities, such as food, that are not available in the forest (Ibrahim et al., 2013) (see also 4 7.1). Local knowledge guarded by the indigenous communities is disappearing, or beginning to be ignored. Natural habitat that used to be preserved (i.e. *sialang* trees as an indicator for preservation of habitat) and is believed to be the source of life, is now being replaced by widespread plantation and development (Césard and Heri, 2015).

5.4.2.4 Invasive species

Invasion by Africanized bees is perceived as a particular risk for Guna people in Panama, as they killed a number of people since they arrived more than twenty years ago. Elephant grass (*paja canalera, Saccharum spontaneum*) is an aggressive alien grass also causing problems; it is the main cause of the degradation of the soil due to the fires and the decline of forested and agricultural landscape (López et al., 2015). Among the Kayapo in Brazil, the invasive *Apis mellifera scutellata* (African bee subspecies) was initially considered highly problematic due to its aggressiveness and competition with native bees, but after two decades it came to be recognised as the strongest bee who takes care of other bees (Posey, 1983a). Mbya Guaraní, peoples from the Paraná State of Brazil, have noted that the exploitation of the introduced Western honey bee (*Apis mellifera*) is impoverishing their ecosystems and decreasing honey yields from native bees (Cebolla-Badie, 2005). Māori people in New Zealand believe that the introduction of exotic invertebrates and vertebrates has caused major declines in pollinator communities over the last 75 years, for example through introduced possums eating flowers (Doherty and Tumarae-Teka, 2015). On the other hand, feral bees became an important part of the Tuawhenua way of life, providing honey that was used for old people, honoured guests and babies, until their decline in the 1990s. Introduced plant species are also noted as supporting some native birds with floral and fruit resources (Doherty and Tumarae-Teka, 2015).

5.4.2.5 Climate change

Climate change affects Indigenous peoples and local communities' relationships with pollinators (Athayde, 2015). In the Himalayas, Kullu beekeepers have noted changes to swarming times and population sizes, with every season occurring about one month earlier. Pest levels are higher due to drought conditions, and the quality of seed production is adversely affected by lower bee populations (Sharma, 2004). In central America, Guna people have noticed that birds once restricted to latitudes south of Ecuador are now arriving in Panama, bringing with them the plant species that they eat. On the other hands, a bird that their grandparents' generations used as warnings of danger at home are no longer seen, which they attribute to climate change-driven migrations. Climate changes is also changing the timing of biotemporal signals of when to plant and harvest, changing the agricultural calendar (López et al., 2015).

5.4.2.6 Loss of access to traditional territories

Indigenous groups have also lost access to their traditional territories, leading to a decline in traditional bee management practices (Césard and Heri, 2015; Samorai Lengoisa, 2015).

Ogiek people of Kenya, whose migratory patterns follow the production of different bees from the lowlands to the highlands, have now been excluded from access to rock- and ground-nesting bees because their traditional lowlands forests have become part of Lake Nakuru National Park (Samorai Lengoisa, 2015), causing serious and sudden loss of biocultural diversity, language and traditional practices. They believe this exclusion to be unlawful. Māori people acknowledge that individuals negotiating land settlements of behalf of their people are required to give up their lives and also those of their families for the fight, losing the time to connect with land, people and culture, and to pass on ILK, in the process (Doherty and Tumarae-Teka, 2015).

5.4.2.7 Changes to and loss of traditional knowledge, tenure and governance systems that protect pollination

Substantial research on traditional knowledge has identified loss and decline as small-scale societies became more integrated within nation-states and the market economy (Gómez-Baggethun and Reyes-García, 2013; Oteros-Rozas et al., 2013). These losses extend to declines in knowledge about pollination-related agricultural and management practices, for example of knowledge of flowering plants that attract pollinators (Reyes-García et al., 2013a). Amongst Māori, the rural-urban migration in the 1950s, driven by economic and environmental change, took many people away from their elders, customs, and practices, driving loss of ILK relevant to pollination (Doherty and Tumarae-Teka, 2015).

More recent studies have focused attention on the dynamic nature of traditional knowledge, so that while specific bodies of knowledge have undoubtedly been lost, where societies retain the ability to generate, transform, transmit, and apply knowledge, traditional knowledge retains a vital role, for example in retaining land races and fruiting trees that foster a diversity of pollination resources alongside commercial varieties in home gardens and agroforests in Spain, Portugal and Mexico (Castro-Luna and Galindo-Gonzalez, 2012; Reyes-García et al., 2014a; Vallejo et al., 2014; Vallejo et al., 2015). The types of (secular) ILK that are retained also adapt to the context (Reyes-García et al., 2013b). Governance and tenure arrangements strongly influence whether or not societies are able to generate, transform, transmit and apply their traditional knowledge. Both governance and tenure are also experiencing declines and disruptions in diverse developed, emerging and developing economies (Hill et al., 2012; Mannetti et al., 2015; Tang and Gavin, 2015).

National law and development projects focused on agricultural production, rural development and nature conservation have led to breakdown of tenure systems and fragmentation of governance arrangements that are vital to shifting agriculture and other practices that protect pollination, even where some recognition of land rights occurs, for example in the Bolivian Amazon and the northern Philippines (Prill-Brett, 2003; Reyes-García et al., 2014b). Traditional diverse farming systems are threatened by lack of payment for the non-market ecosystem goods and services they provide, out-migration of farmers due to economic crisis and opportunities elsewhere, and cultural erosion (Koohafkan and Altieri, 2011). In southern Madagascar, the World Bank's clearing and plowing the land campaign undermined the Tandroy people's social-ancestral relationships that govern practices including protection of forests with bees that serve as pollinators of nearby bean crops (von Heland and Folke, 2014). In relation to intellectual property, national copyright law allows aappropriation of Native American imagery and symbology for sporting and other mascots, leading to a loss of cultural values associated with pollinators. Native Americans have pursued legal challenges to this appropriation, but the issues are not resolved and remain controversial (Johansen, 2007; King, 2013).

5.4.2.8 Pollination governance deficits

The International Risk Governance Council (IRGC, 2009) has identified governance deficit arising because the threats to pollination and related risks are not adequately taken into account in policies and regulations that may affect pollinators and their services. Their review of the current regulatory and governance context identified the main deficit is that most regulations that affect pollinators and pollination are not specific to pollination (IRGC, 2009). Their report then focuses on five particular aspects of governance deficits: uncertainty of science; lack of adequate economic schemes to internalise environmental costs; absent or inadequate land use policies; inadequate stakeholder participation and consultation; and difficulty of medium- to long-term planning. Chapter 6, section 6.2.1.2, summarises the progress towards reducing these barriers, and additional responses.

Here we consider impacts that result from the overall deficit in pollination risk governance, the lack of specificity to pollination. Governance of pollination extends across many sectors such as agriculture, trade, nature conservation, and encompasses the complex roles of, and power relationships between, for example, civil society, governments, the private sector, indigenous peoples and local communities from local to nation-state to global scales (IRGC, 2009). While governance has many definitions and indicators (Ernstson et al., 2010), in this context of lack of specificity, the Graham et al. (2003) definition is useful, as it highlights *interactions,* and these pose both risks and opportunities in pollination governance (IRGC, 2009). For example, Ernston et al.’s (2010) empirical analysis of the governance of pollination and seed dispersal services in Stockholm highlighted how interactions lead to key risks including highly contested land use, numerous, fragmented multi-level administrative units that trigger under-valuing of pollination services, marginalization of key actors oriented to protection of pollination, scale mis-matches, networks that cross scales but do not span (e.g., cemetery managers do not link with allotment gardeners), and low levels of flexibility for adaptation.

Analysis of pollination governance within the European Union identified problems from (horizontal) interplay across sectors, e.g., contradictory goals between agricultural and nature conservation that impact on pollination resources, and from (vertical) interplay between fine-scale cultural variation in motivations and practices that protect pollination and the homogenizing effect to EU directives (Ratamäki et al., 2015). Empirical analysis of the factors affecting farmers’ decisions to adopt pollination-friendly practices in coffee plantations identified farmers’ perceptions and attitudes, social-location factors, institutions, certification schemes, and markets as powerful drivers across local, regional and larger scales; a conceptual model of these interacting forces was created to provide the foundation for future research into interventions that would enhance pollination (Bravo-Monroy et al., 2015).

5.4.3 Introduction to management and mitigation options

As noted in the introduction, this chapter addresses management and mitigation options as appropriate to different visions, approaches and knowledge systems, of impacts of the decline of diversity and/or populations of pollinators. The concept of *management and mitigation* options is very similar to Chapter 6 concepts of *responses* to risks and opportunities associated with pollinators and pollination, although perhaps with greater emphasis on avoiding situations that create a need to "respond". The Chapter 6 responses focus on the drivers identified in Chapter 2 (see Table 6.2.3). Again, many of the people's experiences of declines and associated drivers identified through the assessment for this chapter are the same as, or similar to, those in Chapter 2, but there are several differences. Notable differences include the identification in this chapter of drivers related to loss of access to traditional lands, and changes to and loss of traditional knowledge, tenure and governance systems that protect pollination and pollination governance deficits Table 5-4. Chapter 6 does discuss pollination risk governance deficits, but as a response rather than a driver.

28. Table 5.4. Similarities and differences between Chapter 2 drivers and peoples' experiences of drivers identified in this chapter.

Drivers (Chapter 2)	**Similarity and differences with people's experiences of declines and associated drivers (Chapter 5)**	**Most relevant responses (management and mitigation options) described in this chapter (Chapter 6 relevant section)**
Land use and its changes (2.2)	Similar: Habitat loss, fragmentation and degradation (5.4.2.1)	Food sovereignty and ecological intensification (6.4.1 Agriculture, agroforestry and horticultural practices)
Pesticides, GMOs, veterinary medicines and pollutants (2.3)	Similar: Pesticides and herbicides (5.4.2.2)	Included in other responses
Pollinator diseases and pollinator management (2.4)	Similar: Changes to and loss of bee management practices and knowledge (5.4.2.4)	Livelihoods and beekeeping (6.4.4 Pollinator management and beekeeping)
Invasive species (2.5)	Some differences: Invasive species – people experience these as both declines and gifts (5.4.2.3)	Biocultural conservation (6.4.3 Nature conservation)
Climate change (2.6)	Similar: Climate change (5.4.2.8)	Included in other responses
Multiple interacting threats: • Climate change and land use	Different. People's experiences are mostly of multiple interacting	Values and frames approaches to conservation (6.4.6 Policy,

- Pathogens and chemicals in the environment - Bee nutrition and stress from disease and pesticides (2.7)	threats that impact widely on their values.	research and knowledge exchange across sectors)
Indirect drivers in the context of globalization - International trade - Increasing human footprint - Shifting pesticides to less regulated countries (race to the bottom) (2.8)	Different. - Loss of access to traditional territories (5.4.2.5)	Rights-based approaches to conservation (6.4.6) Participatory management approaches (6.4.3 Nature conservation)
	- Changes to and loss of traditional knowledge, tenure and governance systems that protect pollination (5.4.2.6)	Biocultural conservation (6.4.3 nature conservation) Knowledge co-production (6.4.6) Strengthening traditional governance systems (6.4.3)
	- Pollination governance deficits (5.4.2.8)	Collaborative governance (6.4.6 Policy, research and knowledge exchange)

28 also presents the management and mitigation options considered here in response to these drivers. These options represent a range of integrated responses that focus on minimizing impacts in ways that ensure protection of the many contributions of pollinators as part of supporting nature’s benefits to people and good quality of life. Chapter 6 also includes material that is relevant to minimizing such impacts, for example in relation to options such as “diversify farm systems”. To avoid repetition, we have included cross-references to relevant material in this chapter in the Chapter 6 text on responses.

As largely integrated responses, the ten options reviewed here generally focus on protecting aspects of both nature’s benefits and good quality of life, and address multiple drivers. Nevertheless, there are some differences of emphasis – for example, rights-based approaches respond directly to the driver of lack of access to traditional lands, and biocultural conservation explicitly recognizes ecosystem dynamism and in some cases welcomes invasive species. Table 5-4 indicates where particular management and mitigation options are relevant to specific drivers, together with the related section in Chapter 6.

5.4.4 Management and mitigation options most relevant to the agricultural sector

5.4.4.1 Food security and ecological intensification

Lack of access to food, and extreme poverty, remain key concerns for many Indigenous peoples and local communities in their relationships with pollinators (Perez, 2015). “Food sovereignty” is an umbrella term for particular approaches to food security that include the ability to determine one’s own agricultural and food policies, resilience and ecological intensification. Tackling problems of hunger and malnutrition is thereby linked to the rights of peoples to define and maintain healthy and culturally appropriate food, produced through ecologically sound and sustainable methods grounded in rural livelihoods (Windfuhr and Jonsen, 2010; Sahu, 2011). Food sovereignty is relevant to pollination protection because of its connection with diversified farming systems and management practices that foster diversity and abundance of pollinators and pollination resources (Kremen et al., 2012). Food sovereignty focuses on reducing global food trade and reorienting food systems around local production and agro-ecological principles, opposing several of the key risks to pollinators and pollination such as habitat loss, fragmentation and degradation (4) (Wittman et al., 2010; Clapp, 2014). While diverse in its interpretations across the globe, food sovereignty acts as a powerful mobilizing frame for social movements, as well as a set of legal and quasi-legal norms and practices aimed at transforming food and agriculture systems (Edelman, 2014). Food sovereignty emphasizes local initiatives, such as barter markets, that can help overcome the homogenizing effect of globalized corporate economies and trade, recognized as a driver of risks to pollination (Argumedo and Pimbert, 2010; Pirkle et al., 2015).

Food sovereignty is a developing approach that shows the promise of integrating a wide range of positive opportunities, including the quality, quantity, availability, and origin of food, the identity of the producers and styles of farming that have been recognized as pollinator-friendly (van der Ploeg, 2014) (5.2.8). Food sovereignty protects peasant agriculture systems that see agriculture as co-production, i.e., the ongoing interaction, intertwinement and mutual transformation of humanity and living nature. Food sovereignty builds the capacity for enhanced agricultural productivity through social networks that join together interdependent producers and places, and enable sharing of traditional and agro-ecological knowledge, cultivating alternate circuits of exchange, and building urban-rural partnerships (Aguayo and Latta, 2015). van der Ploeg (2014) describes how these systems of peasant agriculture strengthen the complementary among species, such as between pollinators and plants, as one of their strategies for improving productivity.

Interest in the potential of food sovereignty and ecological intensification to meet food and nature conservation goals is growing (FAO, 2014b) (Case example 5.18). Partnerships that support sustainable and ecological intensification have proven effective in increasing yields, with one study of 286 projects involving 37 million ha and 12.6 million chiefly small-holding farmers showing an average of 79% yield increase across diverse systems (Pretty et al., 2006; Pretty and Bharucha, 2014). Food sovereignty has recently been identified as a key strategy to overcome situations where agricultural trade liberalization leads to increased food insecurity, malnutrition, and exposure to environmental contaminants (Pirkle et al., 2015). In addition, a recent global analysis of nitrogen transfers in terms of functional relationships among crop farming, livestock breeding and human nutrition shows that slight improvements in agronomic performance in the most deficient regions (namely Maghreb, the Middle East, sub-Saharan Africa, and

India) would make it possible to meet the global protein requirements with much less international trade (hence more food sovereignty), and reduce N environmental contamination (Billen et al., 2015).

Case example 5.18. Indigenous Pollinators Network to support food sovereignty.

Location: global (Roy et al., 2016)

Indigenous peoples around the world

The Indigenous Partnership for Agrobiodiversity and Food Sovereignty established the Indigenous Pollinators Network to draw attention to the roles of traditional indigenous production systems of beekeepers, farmers and honey hunters in managing bees. The Network strengthens people to counter the marginalization process these local indigenous knowledge holders face on a daily basis. In particular, the initiative is providing inputs about how the traditional knowledge of indigenous peoples and the work of modern scientists could be linked more equitably and usefully. The network is promoting: bottom-up evidence to value indigenous agroecological knowledge on pollination; identification of good practices for enhanced livelihood opportunities; and awareness raising and knowledge exchange among indigenous communities, for example through learning routes. Case studies underway have highlighted great challenges to traditional practices that maintain pollinators and beekeeping from climate change, proliferation of commercial crops replacing forests, and indiscriminate use of agrochemicals. Many people were concerned that their food security was threatened by pollinator decline, and sought agricultural development based on strengthening their traditional production systems.

5.4.5 Management and mitigation options most relevant to the nature conservation sector

5.4.5.1 Heritage listing and protection

Identification, listing and protection of heritage values has been established globally since the *Convention Concerning the Protection of the World Cultural and Natural Heritage*[29] was finalized in 1972 and the *Convention for the Safeguarding of the Intangible Cultural Heritage*[30] in 2003. Many nation-states also have their own heritage acts and lists. Several landscapes that are vital to pollinators are already protected; on the World Heritage List (0); opportunities exist to strengthen the protection of others that are on the Tentative List (e.g., the Tsavo Parks and Chyulu Hills Complex with many bird pollinators)[31].

Preparation of heritage lists generally involves establishment of a set of criteria that must be met in order to qualify for listing. Protection requires development and implementation of a management plan, and ongoing monitoring to ensure that values are being maintained, which includes pollinators' values where

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[29] http://whc.unesco.org/en/about/

[30] http://www.unesco.org/culture/ich/en/convention

[31] http://whc.unesco.org/en/tentativelists/5515/

they are recognized as part of the significant heritage. The “World Heritage List In Danger” is established when a listed site is losing its values – if the processes of degradation continue, the site will be removed.

The *Convention on the Intangible Cultural Heritage* primarily uses knowledge to achieve its aim of safeguarding the uses, representations, expressions, knowledge and techniques that communities, groups and, in some cases, individuals, recognise as an integral part of their cultural heritage. The Representative List promotes understanding of practices, and management approaches are also listed, for example the protection of traditional knowledge of Totanac people, which includes agroforestry systems that protect pollinators and stingless beekeeping (Case example 5.19). Heritage listing and management activities conducted in ways that empower associated communities can also protect biocultural diversity (Hill et al., 2011a).

The *Globally Important Agricultural Heritage Systems* list also uses knowledge to promote public understanding, awareness and recognition, and dynamic conservation approaches that concurrently foster nature and culture, sustainable agriculture and rural development. Projects have been established in 19 countries to support national and local stakeholders to develop and implement adaptive management[32].

Case example 5.19. Xtaxkgakget Makgkaxtlawana Centre for Indigenous Arts - Best Practice Cultural Heritage management[33].

Location: Mexico

Totonac people of Veracruz

The Center for Indigenous Arts was established by Totonac people to create an educational institution to transmit their teachings, art, values and culture, while also providing favorable conditions for indigenous creators to develop their art. Totanac people are credited with being the first to cultivate and domesticate the vanilla orchid, and their traditional knowledge and practices include stingless beekeeping and their own agroforestry system, which incorporates diverse pollinators and pollination resources (Alcorn, 1990; Arce Castro et al., 2015). The structure of the centre represents a traditional settlement with separate ‘Houses’ specialized in one of the Totonac arts, including pottery, textiles, paintings, art of healing, traditional dance, music, theatre and cuisine. At the ‘House of Elders’, students acquire the essential values and beliefs of the Totonac through integral and holistic transmission of knowledge. The house-schools link each practice to its spiritual nature. This cultural regeneration is renewing Totonac language as the vehicle for teaching, reestablishment of traditional governing bodies, and reforestation of the plants and trees needed for cultural practice, protecting pollinator-pollination webs. The centre also promotes ongoing cooperation with creators and cultural agencies from other states of the country and from around the world.

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[32] http://www.fao.org/giahs/giahs-home/tr/

[33] http://www.unesco.org/culture/ich/en/Art18/00666

5.4.5.2 Participatory management approaches

Globally, there are many good examples of participatory conservation approaches that engage indigenous peoples and local communities in ways that promote socio-cultural values (Borrini-Feyerabend et al., 2004). The Programa para la Conservacion de Murcielagos Migratorios (PCMM; Conservation Program for Migratory Bats) in Mexico provides a mix of research, education, and participation that brings people closely into conservation work. PCMM mobilizes people to protect bat roosts, focusing particularly on the important pollinators lesser long-nosed bats *(Leptonycteris curasoae*), near where they live, to design management plans, and has helped establish interpretive trails, ecotourism facilities and the local production of bat-based arts and crafts, facilitating people to become local stewards (Withgott, 1999). Local community involvement is helping protect the pollination by bats in New Zealand (Case example 5.2020, Figure 5-25).

Case example 5.20. Local community projects to maintain Wood Rose pollination by the Lesser Short-Tailed Bat.

Location: **New Zealand**

Local conservation groups and Māori people

The New Zealand lesser short-tailed bat (*Mystacina tuberculata*) is the primary pollinator of the wood rose (*Dactylanthus taylorii*) (Ecroyd, 1996), New Zealand's only completely parasitic flowering plant (La Cock et al., 2005). Both species have seen significant declines. Once widespread, bat numbers have been decimated through introduced predators (rats, stoats, and cats) (Molloy and Daniel, 1995) and today they are thought to exist in less than 5 per cent of their range prior to human settlement (Ministry for the Environment, 2007). The wood rose, popular with woodworkers and historically collected from New Zealand forests, is also chronically threatened and in serious decline (La Cock et al., 2005), due primarily to its consumption by the introduced brushtail possum (*Trichosurus vulpecula*) (Ecroyd, 1996). Protection of wood rose flowers requires cages that excludes possums, but allows bat access (Ecroyd, 1995).

Many local groups are empowering the community to take action. The Tongariro Natural History Society has focused on the identification and caging of wood rose plants in the Kakaramea region and the Ngāti Tahu-Ngāti Whaoa Runanga Trust in the Tutukau forest (The Runanga, 2015). The Nga Manu Trust is actively monitoring wood rose and using photography by David Mudge to gain new insights into the plant-pollinator relationship (Balance, 2015). Research by Pattemore (2011) has been a driver for kick-starting a project with wide community support to reintroduce short tailed bats to the Auckland region. Ark in the Park, a project by Forest and Bird (2015) aims to re-introduce wood rose into the Waitakere Ranges near Auckland.

61. Figure 5-25. The New Zealand short-tailed bat (*Mystacina tuberculata*) and the wood rose (*Dactylanthus taylorii*).
© Megan Gee. Reproduced with permission.

In Ethiopia, Non-Timber Forest Product and Participatory Forest Management projects support agreement-making between governments and local communities to recognise community rights to use and manage the forest. Interestingly, in one project, government staff initiated on-farm beekeeping to alleviate pressure on forests through alternative, non-forest based livelihoods. Through the partnership with locals, the project team realized that introducing on-farm beekeeping methods was inappropriate, while supporting traditional forest beekeeping keeps people connected with the forest, which is essential for conservation. Instead, the partnership focused on business development systems, developing supply-chain links with traders that have resulted in improved incomes (Abebe and Lowore, 2013). Women in these Ethiopian communities commonly use products of beekeeping, specially make tej (honey wine) and honey beer; opportunities for their great involvement in market activities appear available (Adgaba et al., 2008).

In Nyika National Park, Malawi, mutual benefits have developed from government supporting local people to place beehives in suitable foraging locations within the park; the beekeepers in turn undertake early burning near their hives which protects the forest from later destructive wildfires, and help to see and report poachers (Hausser and Savary, 2009). In Kenya, establishment of a Mau Forest Complex Authority for co-management, and participatory management approach with the Ogiek (as recommended by the Prime Minister's Task Force on the Conservation of the Mau Forest Complex (2009)) would provide a way forward to re-establishing their relationships with bees, the forests, their songs, prayers and vital biocultural diversity.

5.4.5.3 Biocultural approaches to conservation

Conservation of biological diversity, cultural diversity and the links between them is referred to here as "biocultural approaches to conservation". These biocultural approaches to conservation are an emerging field of endeavor building on practice and scholarship in biocultural diversity and heritage, social-ecological systems theory, and different models of people-centered conservation (Gavin et al., 2015). Biocultural conservation is closely linked to endogenous development, that is development based on peoples' own understanding of the world, their priorities, their goals and their historical and cultural contexts (Rist, 2007). Endogenous development recognizes that biocultural actors live and link with both local and global contexts, and thus removes the focus on community-based versus top-down, and replaces it with multi-scalar collaborative practices that connect and find empowerment in both (Hill et al., 2011a). Integrated conservation and development projects, co-management and community-based conservation are examples of methods to facilitate biocultural conservation.

Gavin et al. (2015) present a set of principles for biocultural approaches to conservation (Box 5-3). They present the evidence behind the need to adopt biocultural approaches as two-fold: first that numerous international and national human-rights institutions require such approaches; and second that biocultural approaches build capacity for conservation by bringing more actors who are applying more options, with greater likelihood of long-term success.

Box 5-3. Principles of biocultural approaches to conservation
(Source Gavin et al., 2015).

1. Acknowledge that conservation can have multiple objectives and stakeholders.
2. Recognise the importance of intergenerational planning and institutions for long-term adaptive governance
3. Recognise that culture is dynamic, and this dynamism shapes resource use and conservation
4. Tailor interventions to the social-ecological context
5. Devise and draw upon novel, diverse and nested institutional arrangements
6. Prioritize the importance of partnership and relation building for conservation outcomes
7. Incorporate the distinct rights and responsibilities of all parties
8. Respect and incorporate different world views and knowledge systems into conservation planning

Biocultural approaches will have different outcomes for pollinators and pollination, as co-evolution and dynamism are usually welcomed and accepted. Exotic species often become integrated into totemic systems, and afforded the same respect, care and reciprocity as other living beings. This emphasis on accommodating evolutionary processes, rather than managing ecosystems to some past "natural" state is gaining greater support in the scientific community (Carroll, 2011; Hendry et al., 2011). For example, African honey bees and European bees are now recognised as important pollinators in degraded tropical forests of South America and fragmented dry forests of south-eastern Australia respectively (Dick, 2001; Gross, 2001).

Habitat restoration is a frequent outcome of biocultural approaches (Case example 5.21). For example, in the central Mexican states of Guerrero and Tlaxcala, Indigenous Nahuatl and Totonaco farmers from Sierra Norte of Puebla have allied with small farmers to conserve soil, water and biodiversity as they restore pollinators to hundreds of acres of smallholder farmland in their Farmer to Farmer Pollinator Restoration Project (Holt-Gimenez, 2014). Bringing traditional knowledge of bee ecology into the demarcation of tropical forest for protection in South America provides an important opportunity to protect both the critical hot-spots for pollinators and the associated biocultural knowledge of peoples like the Kawaiwete (Villas-Bôas, 2015). Rescue of stingless bee nests, and provision of these to local beekeepers, is helping to mitigate some impacts caused by deforestation in the Amazon basin, Brazil (Costa et al., 2014). Protection of biocultural refugia has been identified as an effective means of enhancing food security and biodiversity (Barthel et al., 2013a, 2013b).

Case example 5.21. Beekeeping to empower biocultural diversity and endogenous development.

Location: Southern Rio Grande do Sul State, Brazil.

Four different expressions of family farming and traditional peoples: peasant, agrarian reform settlement, quilombola community, and indigenous Guarani village (Wolff and Gomes, 2015).

Two organisations, Institute of Sociology and Peasant Studies of the University of Córdoba (ISEC), and the Temperate Agriculture Program of the Brazilian Agricultural Research Company (EMBRAPA), worked with these communities over several years toward organization and mobilization of farmers and traditional people to generate organizational structures that supported development of agro-ecological beekeeping systems. Beekeeping systems are understood by members of these communities as important for the production and sale of honey, and for pollination, and particularly because of its influence on their own strategies of organization, participation, empowerment and credibility. For example, the indigenous Gurani people undertake enrichment planting to change the forest so it has more fruits and more honey. Peasants, their representative bodies and the technicians from involved institutions of research and extension, worked together on multi-institutional articulation processes that enabled positive changes in practices used by beekeepers in the field, helping to increase production and productivity of the apiaries. This joint approach contributed to the empowerment of peasants and traditional communities, supported their aspirations for autonomy and food sovereignty, and strengthened the ability to transfer knowledge through greater understanding of the socio-political dimension of agroecology.

5.4.5.4 Strengthening traditional governance that supports pollinators

Diverse farming systems and ecosystem management practices that support pollinators critically depend on unique and complex forms of governance, involving kinship, territoriality, settlement, group membership and identity, gender relations, and leadership and political organization for decision-making (Koohafkan and Altieri, 2011). Policies, regulations and incentives can be used to strengthen these governance systems, and counter the risks posed by economic factors driving outmigration and abandonment of customary institutions. In the GIAHS initiative, although relatively recent, early results show effectiveness in countering economic risks from certification of products, tourism, research underpinning promotion, human resource development, and multi-stakeholder participation in adaptive management projects (Koohafkan and Cruz, 2011; Sun et al., 2011; Son et al., 2012; George, 2013). Endogenous development to strengthen the governance by the Hani and Yi ethnic minorities, which depend on tree worship, has been identified as critical to maintenance of the forests, villages, water channels and rice-terraced agricultural landscapes in Yunnan Province, China (Gu et al., 2012).

Protected areas, long the cornerstone of conservation, are now recognized by the International Council for the Conservation of Nature (IUCN) as existing under diverse governance types (Borrini-Feyerabend et al., 2013). Four different governance types are recognised: (1) government, where a national, provincial or local agency is in charge; (2) shared governance, where collaborative, joint or transboundary arrangements involve a range of different actors in decision-making; (3) private governance, where the protected area is run by an individual owner or organization; and (4) governance by indigenous peoples and local communities. “Indigenous and Community Conserved Areas” (ICCAs) is the term applied to the last category (Borrini-Feyerabend et al., 2013). ICCAs consist of natural and/or modified ecosystems containing biodiversity values, ecological services, and cultural values, voluntarily conserved by indigenous and other communities through local or customary laws. Such areas range in size from <1 ha sacred groves to >30,000 km^2 indigenous territories in Brazil, and are associated with the protection of links between biodiversity and wildlife that ensure pollination (Berkes, 2009; Koohafkan and Cruz, 2011; Sun et al., 2011; Son et al., 2012; George, 2013).

Recognition of ICCAs through effective means, such as inclusion in national reserve systems, can strengthen their sustainability (Berkes, 2009; Kothari et al., 2012; Davies et al., 2013). Governance evaluation and improvement provides a means to strengthen the traditional institutions (councils of elders, clan or tribal chiefs, village assemblies) that ensure ongoing protection and management of pollination and other ecosystem services (Kothari et al., 2012; Borrini-Feyerabend et al., 2013). In Australia, management of ICCAs often starts with identification of key cultural and natural assets (Hill et al., 2011b; Moorcroft et al., 2012). The Wunambal Gaambera people have focused particularly on the protection of the flying fox, an important pollinator of eucalypt trees vital for providing timber used in cultural artefacts (Birt et al., 1997; Birt, 2004; Wunambal Gaambera Aboriginal Corporation, 2011) (Case example 5.222).

Case example 5.22. Wunambal Gaambera Indigenous Protected Area and Flying Fox Pollinators.

Location: the Kimberley region of north Western Australia

Wunambal and Gaambera Indigenous peoples (Wunambal Gaambera Aboriginal Corporation, 2011)

The Wunambal Gaambera people developed their plan for health country by prioritizing 10 targets (cultural or environmental assets) for protection: Wanjina Wunggurr Law; right-way fire; Aamba (kangaroos and wallabies) and other meat foods; Wulo (rainforest); Yawal (waterholes); bush plants; rock-art; cultural places on islands; fish and other sea foods; and Mangguru (marine turtles) and balguja (dugong). Wulo (rainforest) protection highlights protection of pollinator-dependent fruits and a key pollinator, the flying fox.

Wulo has lots of different food and medicine plants, as well as other plants that we use. The main things we collect are gunu (round yam), garnmarngu (long yam) and fruit like gulangi (black plum). Wulo has more different types of plants than the moree (savanna woodland). We also hunt animals in the Wulo, like jarringgu (black flying fox) and diigu (birds) like the nyulbu (Torres Strait pigeon) and collect yinari (scrub-fowl eggs). The jarringgu (flying fox), like lots of other animals, has a special Dreaming story and song about it… Wulo is also a special place for lots of diigu (birds). Gangala (orange-footed scrub-fowl) build big nests on the ground. Mandamanda (rose-crowned fruit-dove) and jurul (emerald dove) also live there.

Wunambal Gaambera healthy country plan sets out how they are going to protect the rainforest through controlling feral animals (crazy ants and cane toads), managing fire and other practices.

In Tanzania, a proposal to exclude beekeepers from forests has been turned around through collaborative workshops recognizing the positive contributions of the local community, resulting in the creation of Bee Reserves (Case example 5.23).

Case example 5.23. Bee Reserves protected and managed by local people.

Location: Tanzania

Traditional forest beekeepers (Hausser and Mpuya, 2004; Hausser and Savary, 2009)

The forests of Inyonga area, located between the Katavi National Park, Rukwa-Lukwati Game Reserve and Ugalla Game Reserve, are some of the least disturbed, wild ecosystems in Africa. Beekeeping is traditionally practiced in the area. However, immigration and environmentally destructive activities are posing a threat to these valuable ecosystems. Those responsible for protecting the area were attempting to disallow beekeepers access to the protected area, which in the meantime was being expanded. The Association for the Development of Protected Areas (ADAP) stepped in to assist the Government of

Tanzania to tackle the problem. Through a multi-stakeholder workshop the protected area managers gained a much clearer appreciation that beekeeping is environmentally friendly and contributes directly to the effective protection of the whole ecosystem, whilst generating income for local communities, and strengthening local knowledge and skills. ‘Goldapis’, a Tanzanian company is marketing bee products and developing a highly viable income stream to local people. Bee Reserves were created within the forests that would be protected and managed by beekeepers for their purposes. This provides them with a strong incentive to maintain and manage these forests. The National Beekeeping Policy of Tanzania now includes the creation of bee reserves as a strategy to continue to promote beekeeping within the country, while strengthening forest protection.

5.4.6 Management and mitigation options most relevant to the pollinator management and beekeeping sector

5.4.6.1 Livelihoods and beekeeping

Livelihood approaches, defined here as mechanisms that support peoples’ direct utilization of pollinators and pollination resources, can overcome many economic barriers to effective pollinator protection when they are able to link: (1) customary economies (that require ongoing protection of pollinators); (2) markets (that give these products economic significance in the globalized economy); and (3) investments from government in accompanying research, market analysis and brokering, resulting in what has been termed the “hybrid economy” (Altman, 2007). Stingless beekeeping activities are clearly important in both customary and market economies, and are therefore prime examples where government investments in research and brokering can be very effective (Lyver et al., 2015). For example, obtaining organic certification, links to customers prepared to pay for high-value product in developed nations, and strengthening of traditional social organisation and knowledge have greatly improved incomes for beekeepers in Cameroon (Ingram and Njikeu, 2011) (Case example 5.24). In the coffee landscapes of Colombia, producers have obtained the designation as special coffees by Rainforest Alliance, such as the Café Reinita cerúlea produced in the Serranía de los Yariguíes, San Vicente, Santander Colombia. The name of this coffee recognizes that these ecosystems provide habitat for migratory birds such as the Reinita Cerúlea (*Dendroica cerulea*)[34]. The Mesa de los Santos coffee plantation (Santander) is internationally certified by the Smithsonian Institution as a "bird-friendly coffee plantation", because their management is based on organic agriculture practices (CENICAFÉ, 1999).

Case example 5.24. Local Zambian beekeepers gain market advantage in the EU through organic and fair trade certification.

Location: North West Province of Zambia

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[34] http://www.proaves.org/alternativas-productivas-para-la-conservacion/.

Local beekeepers (Wainwright, 2002; Malichi, 2007)

The North West Bee Products (NWBP) company of Zambia has 6,500 members, who own the company and ensure its management. In the Zambian North West province, NWBP is the largest employer after the government. All of their honey and beeswax is produced by bees housed in local- style bark hives. Their honey is organic certified (from the UK Soil Association), has fair trade certification from Germany, and meets the EU’s stringent import requirements, giving it a comparative advantage on the world market. NWBP began in 1979 with support from GTZ (German Government development organization), and subsequently received support from a variety of donors over the years. The company could not have managed without this support from donors in some years, but is now self-sustaining and successful, with beekeepers annually increasing production, confident in the market for their products. In 2003, NWBP exported 144 tons of honey to the European Union.

The success of this intervention can be attributed to the people’s access to all the types of resources needed to make their livelihoods sustainable: natural resources (strong populations of healthy bees and abundant forest); physical resources (trucks able to navigate rough forest tracks and to enable honey to be transported from the producers to the collection centre, buckets with lids allowing clean honey to be transported); social resources (the strong organization, owned and run by the producers and with access to market knowledge); human resources (the beekeepers’ skills at beekeeping and honey and beeswax harvesting); and financial resources (access by the company to credit when needed).

Across Latin America various efforts are reviving stingless beekeeping through the development of techniques to maintain and reproduce colonies efficiently, to improve the quality and marketability of products for better economic rewards, and increase the value of colonies by additional services such as commercial pollination (Cortopassi-Laurino et al., 2006). Stingless beekeeping is showing signs of recovery for various indigenous groups and local communities of Argentina, Bolivia, Brazil, Colombia, Ecuador, Mexico, and Venezuela. Key elements for the recovery of stingless beekeeping have been: teaching, since many young people have lost the experience from their ancestors and elders; respect for the local costumes and traditions; increased value of products; and development of a market niche for stingless bees products (Cortopassi-Laurino et al., 2006).

Among the “quilombola”, a traditional population of descendants of runaway slaves, or “quilombos”, the practice of meliponiculture has been carried out for generations and provides an elaborate ecological knowledge based on native bees, the melliferous flora and the management techniques (de Carvalho et al., 2014). Training courses for the “ribeirinhos”, traditional populations living near rivers (Kurihara and Cardoso, 2007; Cavalcante et al., 2009), and indigenous groups from the Amazon region have been successful in recovering and strengthening stingless bees rearing practices (Venturieri, 2008a, 2008b). In

New Zealand, the introduced European honey bee production from *Leptospermum scoparium* (mānuka trees) that are vital in the Māori pharmacopeia have resulted in a high-value medicinal mānuka honey industry (Stephens et al., 2005).

Strengthening beekeeping more generally is a key strategy for enhancing rural livelihoods (Gupta et al., 2014). FAO’s diversification tools underpin this approach by providing support for market analysis; development of equipment, standards, certification; marketing, products, packaging; and brokering relationships and trust through supply chains (Bradbear, 2009; Hilmi et al., 2011). Participatory action research has demonstrated successful outcomes from strengthening beekeeping in rural livelihoods in Cameroon (Ingram and Njikeu, 2011). A Salvation Army program around Kavwaya in the lower Congo, initiated more than 20 years ago, has established low-cost beekeeping among rural communities, with significant financial returns — for example, one harvest from five hives returned the equivalent to local average annual wages. People have been able to pay school fees and medical expenses previously beyond their reach (Latham, 2009). Nevertheless, several recent studies have noted that there is significant unrealized potential for beekeeping as a sustainable livelihood in developing world contexts, and recommend strengthening of knowledge as well as technology as key to empowering its adoption (Ubeh et al., 2011; Carroll and Kinsella, 2013; Kimaro et al., 2013; Masuku, 2013; Ja'Afar-Furo, 2014).

Non-destructive honey hunting is also recognized as useful to pollinator protection and rural livelihoods (Joshi and Gurung, 2005). The Indonesia Forest Honey Network (JHMI), a network of producers, is assisting honey hunters to market their products with a premium for their sustainable practices (Césard and Heri, 2015). Support for local honey harvesters in the Bijagos Islands of Guinea Bissau has enabled them to adopt non-destructive practices that maintain rather than damage pollination resources (Case example 5.25).

Case example 5.25. Livelihoods through beekeeping in mangroves.

Location: Guinea Bissau

Local communities in the Bijagos Islands (Hertz, 2009)

In Bijagos Islands, west of Guinea Bissau, honey hunters are attracted by the high productivity of bees in mangroves, particularly the black mangrove *Avicennia germinans*, known as the honey mangrove. It has small white flowers that produce abundant nectar. A Danish project supported local honey harvesters with protective clothing, a smoker, a knife, a bucket and some type of bee brush. Because of the protective clothing, the harvester does not have to kill the bee colony as happened previously. The beekeepers look for wild bee colonies in the mangrove and when a new one is found, it is marked as a sign that it belongs to a beekeeper. One beekeeper can in this way, without any high investment, become the owner of 30 or more bee colonies.

Beekeeping provides one of the few sustainable ways to use mangrove and with these simple protective measures can be done without harming the bees. Beekeeping may exert a positive influence on the forest, through the activities of the bees as pollinators. By ensuring the local people benefit economically from mangrove beekeeping, it is easier to protect the mangroves against total destruction from cutting and burning.

5.4.7 Management and mitigation options most relevant as integrated responses

5.4.7.1 Values and frames approaches to conservation

“Values and frames approaches to conservation” encapsulate a range of new methods that focus on framing conservation to link with peoples’ values. These new methods respond to evidence that societal concerns about pressing problems including global poverty, climate change and biodiversity loss, are relatively low compared to others such as terrorism, health care and the economy (Novacek, 2008). The response of concerned scientists has been to provide more and more factual evidence, based on a deficit model of communication that assumes this lack of concern is grounded in ignorance (Groffman et al., 2010). However, human judgements are highly influenced by overall feelings and emotions, understood through metaphors, and how these connect to their most important values and frames (Lakoff and Johnson, 1980; Lakoff, 2004; Crompton, 2010). Values and frames approaches are therefore integrated responses to the key risk identified above that people’s experiences of the causes of pollinator decline are mostly of multiple interacting threats that impact widely on their values.

Values and frames approaches are relatively new in pollination-specific context, although such organisations explicitly undertaking these approaches to promote conservation of biodiversity and ecosystem services generally are now established in 12 countries, including Australia, Sweden and Brazil. Examples in the pollination-specific context include the Faith Task Force that has produced publications on the linkages between several major religions and pollination (NAPPC Faith Task Force, 2012). Other initiatives are linking the art, literature, music and religious significance of bees and others to the scientific understanding of their roles in food production — enabling artists, writers and others to become involved in and supportive of impact management and mitigation. The Pollinator Pathway project, initiated by artist Sarah Bergman, is a good example of this type of approach, linking the values of art, design and ecology. The “Wonder of Discovery” (Figure 5-26.) similarly links people’s values with pollinators, showing engagement as bat and butterfly observers, monarch butterfly taggers, beekeepers, gardeners and through SHARE (Simply Have Areas Reserved for the Environment) (Vibbert, 2013).

62. Figure 5-26. The “Wonder of Discovery” poster showing some socio-cultural values of pollinators (Vibbert, 2013).
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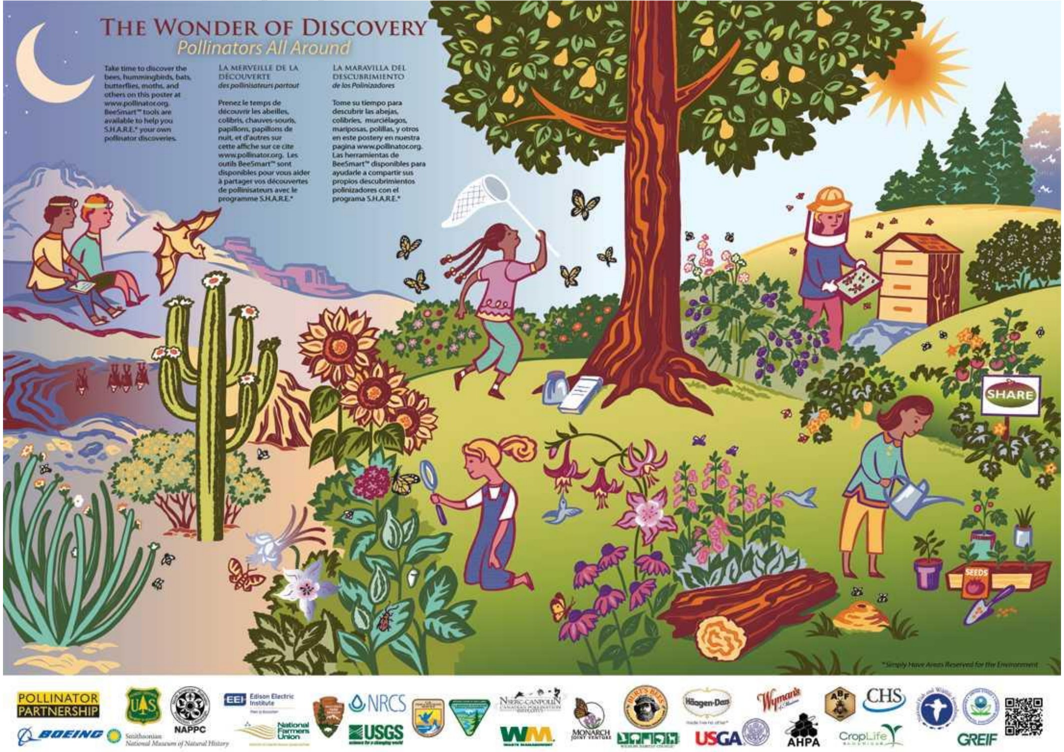

Take time to discover the bees, hummingbirds, bats, butterflies, moths, and others on this poster at www.pollinator.org. BeeSmart™ tools are available to help you S.H.A.R.E.* your own pollinator discoveries.

LA MERVEILLE DE LA DÉCOUVERTE
des pollinisateurs partout

Prenez le temps de découvrir les abeilles, colibris, chauves-souris, papillons, papillons de nuit, et d'autres sur cette affiche sur ce cite www.pollinator.org. Les outils BeeSmart™ sont disponibles pour vous aider à partager vos découvertes de pollinisateurs avec le programme S.H.A.R.E.*

LA MARAVILLA DEL DESCUBRIMIENTO
de los Polinizadores

Tome su tiempo para descubrir las abejas, colibries, murciélagos, mariposas, polillas, y otros en este postery en nuestra pagina www.pollinator.org. Las herramientas de BeeSmart™ disponibles para ayudarle a compartir sus propios descubrimientos polinizadores con el programa S.H.A.R.E.*

*Simply Have Areas Reserved for the Environment

5.4.7.2 Rights-based approaches to conservation

Rights-based approaches are founded on respecting human rights institutions, and integrating human rights norms, standards, and principles in policy, planning, implementation, and evaluation to help ensure that conservation practice respects rights in all cases, and supports their further realization where possible. Rights-based approaches have much in common with biocultural and endogenous approaches, but greater emphasis is given to global and national human rights frameworks and standards (Campese et al., 2009). The United Nations adopted a Statement on Common Understanding of on Human Rights-Based Approaches to Development Cooperation and Programming in 2003 (United Nations, 2003). For example, this statement includes recognition that people are key actors in their own development, and that development processes need to be locally owned, in common with principles for endogenous development.

In relation to nature conservation and integrated responses to risks for pollination and pollinators, rights-based approaches (RBA) in part respond to recognition that fortress conservation approaches have resulted in numerous human rights abuses, through eviction of people from their traditional lands without compensation or fair processes, and through disruption and denial of access to resources essential for their cultural practices and human well-being (Colchester, 2004). RBAs have been identified as capable of enabling actors to understand the situation of marginalized communities in a systemic manner and to address the underlying factors of vulnerability, poverty and powerlessness. They can also help attain long-term conservation while supporting local people to live in dignity (Oviedo and Puschkarsky, 2012).

RBAs can involve a range of different mechanisms, many of which are discussed above as part of biocultural approaches. Here we focus on three aspects particularly relevant to the drivers of risks to pollinators and pollination (28):

- Prior and Informed Consent for conservation, development and knowledge-exchange projects;
- Securing tenure over traditional lands;
- Strengthening governance over traditional lands.

5.4.7.2.1 Prior and Informed Consent over conservation and development projects and knowledge responses

The principle that indigenous peoples are able to give or withhold their 'free, prior and informed consent' (FPIC) to development and conservation projects that will affect them is recognised under international human rights law and as industry best practice for extractive industries, logging, forestry plantations, palm oil, protected areas and projects to reduce greenhouse gas (GHG) emissions from deforestation and forest degradation (Carino and Colchester 2010). Many of these are drivers of risks and opportunities for pollinators and pollination enabling RBA to have a positive effect (5.4.2). For example, the *Forest Rights Act* in India has secured access to forests by honey hunters, keeping alive their knowledge and practices for fostering honey and bees (Demps et al., 2012b). Application of FPIC processes for protected creation in Australia enables identification of culturally-significant pollination-dependent fruit, their bird and bat pollinators and habitats requiring protection (Case example 5.22).

In reviewing application of FPIC, however, Carino and Colchester (2010) found that relatively few national legal frameworks explicitly require respect for this right and World Bank standards have yet to be revised in line with these advances in international law. Connection is lacking between international law respecting the right to FPIC, and nation-states’ laws about resource exploitation in the 'national interest'. FPIC is poorly implemented by corporations and government agencies, reducing it to a simplified check list of actions for outsiders to follow, again removing control over decisions from indigenous peoples (Wilson and Dialogue, 2009; Lehr and Smith, 2010; Minter et al., 2012). Effective FPIC processes enable indigenous peoples' rights to represent themselves through their own institutions and make decisions according to procedures and rhythms of their choosing (Carino and Colchester, 2010).

Many potential knowledge responses to the risks and opportunities of pollination and pollinators are presented in Chapter 6. FPIC from indigenous peoples and local communities is particularly important in these responses. Legal arrangements underpinning research, for example, often transfer rights over the collected knowledge from the original knowledge holders to those who record it; prior agreements (utilising FPIC) are essential to protect ILK-holders’ intellectual and cultural rights. International best practice guidelines for FPIC in knowledge responses include the Tkarihwaié:ri Code of Ethical Conduct

(Convention on Biological Diversity, 2010) and the Code of Ethics of the International Society of Ethnobiology (International Society of Ethnobiology, 2006). The Guna General Congress found effective means of enforcing their intellectual property rights through negotiated agreements (Case example 5-26).

Case example 5.26. Guna governance, intellectual rights and pollinators.

Location: Panama

Indigenous people: Guna; Atencio López oral account, p. 44-45 (López et al., 2015)

Co-produced case example

Underpinned by direct interactions with indigenous and local knowledge-holders

"I summarise the Guna system of governance: Indigenous peoples speak of autonomy, which does not just mean the day to day administration, but also governance of resources. In February 2015, the Guna celebrated 90 years of autonomy. There are 2 systems of authority and control: 1) the communities (52 communities) make decisions on collective rights. There is no private property as it is understood in western culture; 2) the other authority is the caciques, the Guna General Congress is the political administrative organ, while the General Congress of Culture is the spiritual-religious organ, which has the priests. When it is related to natural resources, no project can be implemented in the communities without the approval of the General Congress. There are also projects that are proposed by the communities that the General Congress must approve. Within the Guna community, there is a [customary] law that the government does not officially recognize, but that is respected nevertheless."

Guna people used their governance, even though it is not government-recognised, to protect their intellectual property rights over the pollinator-dependent cacao fruit. The Congress imposed a fine on a business called CocoaWell for using Guna imagery, and negotiated an agreement that they must pay a percentage of their profit (López et al., 2015).

5.4.7.2.2 Securing tenure over traditional lands

Beekeepers and honey hunters often do not have secure tenure under nation-state legal arrangements over the land and forests where their bees forage, and their traditional management systems are being eroded by the expansion of industrial agriculture (van Vliet et al., 2012; Césard and Heri, 2015; Perez, 2015; Samorai Lengoisa, 2015). In November 2014, they argued a case in the African Court On Human and Peoples' Rights that Ogiek community's rights to life, property, natural resources, development, religion and culture were being infringed by persistent harassment and evictions from their ancestral lands in contravention of the international human rights standards of free, prior and informed consent (Samorai Lengoisa, 2015; Tiampati, 2015).The case remains pending with judgement anticipated in 2017. Forests under common property and customary law systems have been shown to produce both livelihoods and biodiversity conservation, complementing biodiversity outcomes from protected areas (Persha et al., 2010). Significant evidence that rights-based approaches work for conservation came from a study of 80 forest commons in 10 countries across Asia, Africa, and Latin America showing that larger forest size and greater rule-making autonomy at the local level are associated with livelihood benefits, and high carbon storage in trees, thereby protecting pollinator resources from the flowering of those trees and presumably also the pollinators (Chhatre and Agrawal, 2009). The authors argued that local communities restrict their consumption of forest products when they own forest commons, and that transfer of ownership to these communities would help support conservation. From this perspective, the global growth in indigenous and community reserves, territories and protected areas is likely to be making a positive contribution to the conservation of wild pollinator habitats (Berkes, 2009; Rights and Resources Initiative, 2014).

Nevertheless, the means of implementation of RBA have a critical influence on their effects. In Cambodia, simultaneous implementation of individual titles for farmers and communal title for indigenous communities has fractured forest commons management systems (Milne, 2013). Land titling in a national park in Cambodia led to a decrease in traditional practices that had maintained agro-biodiversity (Travers et al., 2015). The *Forest Rights Act* in India, promoted as a means of recognizing rights of tribes and forest dwellers, while providing positive benefits to pollinators through support honey hunters as noted above, has also undermined some common property systems and imposed a new set of external agents engaged in defining their affiliations that have been detrimental to social and cultural values (Bose et al., 2012; Kumar and Kerr, 2013). Two major lessons have emerged from these and other experiences in rights-recognition of tenure for conservation (Johnson and Forsyth, 2002). First, the nation-state's efforts to recognise rights are influenced by the broader public discourse and contest between commercial interests that opposed minority groups' rights to valuable resources, civil society interests that may negotiate rights-regimes within the wider public spheres in which rules, rights, and "community" are established, and defended (Johnson and Forsyth, 2002). Second, community-driven planning and capacity building are essential to support implementation of rights in ways that contribute to conservation of biodiversity and ecosystem services.

5.4.7.3 Knowledge co-production

ILK, in co-production with science, can be source of solutions for the present challenges confronting pollinators and pollination. Initiatives that are co-producing relevant knowledge range across classical science-driven investigations of the conditions under which diversified farming systems are underpinned by ILK protect of pollinators and pollination (Webb and Kabir, 2009; Perfecto et al., 2014), through long-term science-ILK projects involving common research design and implementation (Wolff and Gomes, 2015), to projects focused on strengthening ILK through networks. 29 summarizes the examples of knowledge of co-production presented in this sub-section.

29. Table 5.5. Knowledge co-production examples presented here and their contributions to responding to risks and opportunities associated with pollinators and pollination.

Knowledge co-production activity	**Knowledge contribution to responses to risks and opportunities associated with pollinators and pollination (Chapter 5 and Chapter 6)**
Investigating advantages and disadvantages of traditional and modern beehives	Pollinator management and beekeeping: maximizing hive design for healthy bees
Environmental education that involves recovery of traditional knowledge	Monitoring and evaluating pollinators: learning about healthy environments and respect for nature
TEK-science about the ecology of mistletoe infections leading to decline harvests of amla fruit	Habitat management: relevant to increasing health of important pollination resource (amla flowers) for bird pollinators
Community ethnoentomological collections in partnerships with scientists	Pollinator management and beekeeping; monitoring and evaluating pollinators: identifying insects that are new to both science and ILK, empowering traditional knowledge of fostering pollinators
Participatory evaluation of pollinator-friendly farming practices	Diversified farming systems: replacement of traditional shade coffee plantations with sun coffee leading to large declines in migratory bird pollinators
Sharing of traditional and agro-ecological knowledge through networks of peasant farmers	Food sovereignty and ecological intensification and diversify farming systems: promoting pollinator-friendly farming
Indonesian Forest Honey Network	Livelihoods and beekeeping; pollinator management and beekeeping: improving economic returns from forest honey as an incentive to protect forests
Environmental impact assessments incorporating ILK	Pesticides, pollutants and GMOs; landscape planning: pesticides taken up into medicinal plants
Beekeepers and scientists co-producing knowledge about the risks posed by neonicotinoids to bees	Pesticides: Moratorium on use of neonicotinoids based on precautionary approach in favour of pollinator protection
Indigenous peoples' and local communities' engagement in environmental monitoring partnerships	Monitoring and evaluating pollinators: providing baselines for analysis of future trends
Community indicators	Monitoring and evaluating pollinators: baselines for analysis of trends in biocultural diversity
Two-voices story telling about ethnobiology of bees	Biocultural conservation; monitoring and evaluating pollinators; livelihoods and beekeeping
Promoting monarch butterfly as a boundary object, bringing in multiple knowledge	Integrated social and behavioural response; Values and frames approach to conservation

Scientists and traditional beekeepers in Nepal worked together to identify the advantages and disadvantages of traditional and modern beehives, and to promote co-design that maximizes advantages of both (Joshi, 2000). Recovery of traditional knowledge in some communities of Andean countries in South America, and concerns about conservation of pollinators, is evident through different programmes of environmental education and conservation of biodiversity of ecosystems in which different members of the communities participate (Ferrufino and Aguilera, 2006; Meriggi et al., 2008; Pérez and Salas, 2008; Chicchón, 2010; Gómez, 2012; Ferrufino, 2013; Perichon, 2013; Rosso-Londoño, 2013). Although no mention is given directly and specifically to pollinators and pollination, the importance of keeping healthy environments to keep food diversity and to respect nature is emphasized.

Co-production between science and traditional ecological knowledge in the Western Ghats of India was found to fill gaps in both regarding the ecology of mistletoe infections adversely affecting harvests of amla (*Phyllanthus emblica* and *P. indofischeri*), an important source of local income (Rist et al., 2010). Kayapo people and entomologists working together in 1977-78 collected stingless bees that included 56 species recognized by the Kayapo; the entomologists identified 66 species, of which 11 were unknown or not yet described in science, thus adding to the knowledge of both ILK and science (Posey, 1983b, a). Community ethnoentomological collections are proving an effective means of empowering traditional knowledge of insects, including of how to foster pollinators, and building synergies with science in both indigenous and local communities (Aldasoro, 2003; Aldasoro and Argueto, 2013).

Participatory evaluation of pollinator-friendly farming practices in local communities has been developed by the FAO into an effective framework for co-producing knowledge between scientists and farmers for ecological intensification of farming to support improved livelihoods (Grieg-Gran and Gemmill-Herren, 2012). Knowledge co-production is critical for sustainable and ecological intensification of food production in diverse small-holder farming systems, as this type of development is knowledge-intensive (FAO, 2014b).

Knowledge co-production among ILK communities is proving effective in recovery of stingless beekeeping in Brazil (Jaffe et al., 2015). Horizontal networks that join together interdependent producers to share traditional and agro-ecological knowledge, cultivate alternate circuits of exchange, and build urban-rural partnerships, are reshaping the horizons of possibility both for peasant communities and for the broader agri-food system in Chile (Aguayo and Latta, 2015). The Indonesian Forest Honey Network (*Jaringan Madu Hutan Indonesia*, or JMHI) is bringing forest honey harvesters together to exchange expertise in order to offer honey harvested in a sustainable way (for the bees); their honey was the first forest honey in Indonesia to get organic certification, which leads to much better income potential (Césard and Heri, 2015).

Knowledge co-production is vital in environmental impact assessments (EIAs) (Athayde, 2015). Tūhoe Tuawhenuaare in New Zealand through co-production with science have identified that the pesticide ‘1080’ is taken up into their medicinal plants, with unknown effects (Doherty and Tumarae-Teka, 2015). In several Amazonian communities, the role of the indigenous environmental monitors or environmental agent has been increasingly recognized and supported through specific projects that attempt to integrate indigenous, academic and technical knowledge for biodiversity management and conservation (Athayde, 2015). Support for community indicators is emerging as an effective means of knowledge co-production to monitor trends in biocultural diversity (Verschuuren et al., 2014).

Co-production of knowledge between beekeepers and scientists in France and the European Union about the risks posed by neonicotinoids to bees has led to the adoption of moratoriums on their use, reflecting a false-positive evidence-based policy, that prefers to bear the costs of being wrong about the harm posed by these chemicals, rather than overlooking that harm (Suryanarayanan and Kleinman, 2014; Suryanarayanan, 2015). The processes of co-production were complex, involving government regulations to restrict pesticide usage, legal action, protests, compilation of evidence by beekeeper organisations, and consideration by an expert committee of scientists who identified risks that were in agreement with field observations of several beekeepers, stimulating additional research (Suryanarayanan and Kleinman, 2014). The co-produced knowledge thus formed part of collective action by farmers, environmentalists and public actors that shifted policy towards a precautionary approach in favour of pollinator protection (Suryanarayanan and Kleinman, 2014). In the United States, while beekeepers have been very active in compiling and communicating their knowledge of pesticide impacts, this on-the-ground evidence has been dismissed as anecdotal by the Environmental Protection Agency (EPA), who adopt a false-negative evidence-based policy, and will not restrict neonicotinoid use until a definitive role for neonicotinoids in causing bee harm has been proven (Suryanarayanan and Kleinman, 2011, 2013, 2014).

Emerging models for effective co-production between science and ILK emphasize building respect, trust, co-capacity and authentic relationships throughout the entire research process, from conception, through design, implementation and dissemination (Huntington et al., 2011; Adams et al., 2014). Two-voices story-telling between a scientist who moved towards understanding ILK and an indigenous person who took up studying science, reveals how their mutual interest in ethnobiology of bees allowed connections and co-production of knowledge about “bee-cultural” diversity (Rosso-Londoño and Estrada, 2015).

Knowledge co-production activities have highlighted the importance of boundary objects in communication across social groups. Boundary objects have the attributes of being plastic enough to adapt to local needs and the constraints of several parties employing them, yet robust enough to maintain a common identity across social groups. The objects may be concrete, such as a painting (Figure 5-27.) or abstract (Star and Griesemer, 1989). The monarch butterfly is a key such boundary object for linking with diverse socio-cultural values of pollination in North America: its migration has reached an iconic status, becoming a symbol of nature; environmental health; safe migrations across national borders; spiritual metamorphosis and renewal; and the souls returning to Michoacán on the Day of the Dead. These meanings have yielded a powerful story line that connects the conservation and management of the monarch butterfly to the credibility, status, and trust enjoyed by a diverse range of actors (Gustafsson et al., 2015).

63. Figure 5-27. Youth Summit for Biodiversity and Community Action participants co-producing a poster about pollination.
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5.4.7.4 Collaborative governance

Collaborative governance arrangements that support effort alignment, inclusion of local actors, scale-dependent responsibilities for all actor groups, nurturing mid-scale managers and scale-crossing brokers to link multiple actors in the network and support social learning have been identified as key to improving governance of pollinators and pollination in Sweden (Ernstson et al., 2010). Collaborative governance or co-governance is a process that articulates the context, knowledge, process, and vision of governance, linking multiple stakeholders together, and thereby connecting with their multiple socio-cultural values. Landscape and continental-scale efforts at creating habitat corridors, recognized as important to a diverse suite of pollinators, particularly migratory birds, have highlighted the need, potential and challenges in co-productive governance (Perfecto et al., 2014; Wyborn 2015). Rather than a tension between top-down and bottom-up processes, co-productive governance mobilizes institutions with scale-dependent comparative advantage for landscape-scale conservation (Hill et al., 2015a). Collaborative governance supports cross-node, cross-level linkages in polycentric systems (Brondizio et al., 2009).

In managing and mitigating impacts from pollinator decline, collaborative governance approaches offer the advantages of forging linkages across sectors (e.g., agriculture and nature conservation), across jurisdictions (e.g., private, government, not-for-profit) and among levels (e.g., local, provincial and national governments. This linkage capability overcomes many risks arising from the pollination governance deficits identified above (5.4.2.8), such as contested land use, numerous, fragmented multi-level administrative units that trigger under-valuing of pollination, marginalization of key actors oriented to protection of pollination, scale mis-matches, and networks that cross scales but do not span and low levels of flexibility for adaptation. Collaborative governance also addresses impediments such as delayed feedbacks and insufficient information flows that have recently been identified as barriers to delivery of the Aichi Targets under the Convention on Biological Diversity (Secretariat of the Convention on Biological Diversity, 2011; Hill et al., 2015b).

A number of initiatives are now underway globally, for example, the Pollinator Partnership that links corporations, universities, local, regional and national governments and communities into their collaboration across the globe. While results from this initiative are difficult to discern, analysis in the EU context suggests that social norms, habits, and motivation are the key to effective governance outcomes (Ratamäki et al., 2015). Maturation into broad social norms requires engagement of people into over long time periods, and involves several stages, including roles for social actors to challenge current practices, suggesting more time and engagement are needed for effective pollination governance to be leverage from these initiatives (Hill et al., 2013).

5.5 Methods

5.5.1 Review protocols

This review and analysis of the biocultural diversity and socio-cultural values associated with pollinators combined the strengths of systematic review (Collaboration for Environmental Evidence, 2013) with those of historical and social research methods aimed at sourcing the best and richest sources for the topic under investigation (Carr, 1961; Liamputtong, 2008). The review and analysis occurred through four main phases in the lead-up to the Second Order Draft:

- **Initial scoping literature review**: screening, selection and development of First Order Draft (FOD)
- **ILK scoping literature review**: screening, selection, review of FOD and provision of advice for the Second Order Draft (SOD)
- **ILK global and community dialogue**: selection of material from the proceedings (Lyver et al., 2015)
- **Gap-filling literature review**: response to analytical framework for SOD, review comments on the FOD and advice from the ILK scoping review

5.5.2 Initial scoping literature review and development of FOD

Systematic searches of literature databases were conducted by geographic region for South America, North America, Europe, Asia, Africa and Oceania. Search terms focused on biocultural diversity, and pollinators and their social-cultural values for indigenous and local communities. Systematic searches for relevant literature were conducted for South America, North America, Europe, Asia, Africa and Oceania, including regionally-specific terms, such as “sugar-bag” in Australia. Spanish language searches were undertaken for South and Central America. Databases accessed included the Web of Science, York University Library Database, Science Direct and others (30). Additional sources were obtained by using forward and back citations of key articles, and by contacting authors of highly-relevant articles. Material was screened and selected according to relevance, meta- and multi-case analyses, and global and regional overviews. The First Order Draft (FOD) was organized according to geographic regions that guided the literature reviews.

5.5.3 ILK scoping literature review

UNESCO, as the Technical Support Unit for the IPBES Indigenous and Local Knowledge (ILK) Taskforce, issued a call for relevant resources related to ILK and pollinators, which formed the starting point of the ILK scoping review. Systematic searches of English, French and Spanish databases and grey literature were undertaken using a variety of terms including bees, apiculture, beekeeping, flies, butterflies, birds, bats and beetles (30). Categories in the Zero Order Draft also guided the search (e.g., drivers, declines). Additional sources were obtained through personal requests from experts identified during the review. Review of the FOD guided additional searches to fill gaps. Material was screened according to the inclusion of ILK, the depth of its treatment, for more recent studies and for evidence of inclusive research methods. An excel spreadsheet of material was provided as input to the Second Order Draft (SOD).

5.5.4 ILK global and community dialogue

The ILK Taskforce convened an ILK dialogue to ensure interactions with and input from living indigenous and local knowledge systems into the pollination assessment (Lyver *et al.* 2015). Participants were selected from a global call for the global dialogue and subsequent community workshops. Members of the Taskforce also contacted specialist networks, such as the French National Museum of Natural History, to mobilize other expertise for the literature compilation and the workshop. ILK-holders from Africa, Asia, New Zealand and central America participated. Their contributions to this chapter are highlighted as ‘Co-produced case example: underpinned by direct interactions with indigenous and local knowledge-holders’.

5.5.5 Gap-filling literature review

The gap-filling literature review was commenced by the development of an analytical framework for the chapter drawing on Berkes (2012) and input from ILK experts and knowledge-holders in attendance at the second author meeting held to consider review comments on the First Order Draft (FOD). Material arranged geographically for the FOD was reorganized according to these categories, which now form the sections and sub-sections of the chapter. Some material from the FOD was removed as not relevant to the analytical framework or in response to the review comments. Additional categories were generated through consideration of the advice from the ILK scoping review, and the review comments on Chapter 5. The gap-filling literature review concentrated on Web of Science, Google scholar and Google books (30). We also examined international lists of heritage values, which adds rigor to understanding values (Tengberg et al., 2012). Material was prioritized according to relevance, evidence of inclusive processes with ILK holders, peer review, meta-analyses and multiple case studies. While our review highlighted a range of values, few studies had explicitly focused on eliciting values of pollinators and pollination through socio-cultural or holistic methods. An opportunity exists to strengthen our understanding of the values of pollinators through application of these methods; policy-relevant knowledge would be strengthened by filling this gap.

30. Table 5.6. Examples of databases and search terms in each phase of the review and analysis.

Phase	**Examples of data bases and other literature**	**Examples of search terms**
Initial scoping literature review	Web of Science, Google scholar, Springerlink, Cambridge journals, Google, Science direct	Traditional beekeeping, local community knowledge and wisdom, pollination
ILK scoping literature review	Scopus, Research Gate, SciELO, Instituto Socioambiental (http://www.socioambiental.org/pt-br); UN reports, books	TEK, ILK, ecological, knowledge ; apicultura, meliponicultura, escarabajos, savoirs locaux, savoirs traditionnels, savoirs autochtones
ILK global and community dialogue	Key experts and ILK holders identified through the global call and selection	During dialogue themes chosen were change, diversity, multiple values and knowledge protection
Gap-filling literature review	Web of Science, Google scholar, Google Books, World Heritage List, Intangible Cultural Heritage list	Diversified farming, milpa, food and pollinators, heritage, symbolic values, innovations, wax in musical instruments

5.6 Conclusions

The chapter provides the major response within the context of the pollination assessment to the IPBES goal to: *Recognize and respect the contribution of indigenous and local knowledge to the conservation and sustainable use of biodiversity and ecosystems.* UNEP/IPBES.MI/2/9, Appendix 1, para. 2 (d). The constraints of time and capacity have enabled us to interact with only a very few of the numerous indigenous and local peoples globally, to whom the global human population owes so much for their ongoing contributions to biodiversity and ecosystem services that sustain us all in forms such as clean air, sparkling waters and birds that nest and migrate across the globe. We have reduced to ‘categories’ the rich stories of these peoples that intertwine with living beings and spirits and are acutely aware of the flaws in this attempt to give a voice to ILK.

We opened the chapter with some of a story of the Guna people who kindly hosted the ILK-science dialogue for this assessment. The power of stories to communicate between the technical aspects of science and the broader life-worlds of people is gaining greater recognition in academe (Groffman *et al.* 2010); we therefore shall also close the chapter with another story from that dialogue, this one part of a poem that we think captures most what we all collectively seek from the pollination assessment.

Tororo konch logog	*god give us a generation of children*
Konech komeg	*give us honey*
Konech konyegap ongweg	*give us eyes to see the valleys in the forest*
Rpewech mosotig, poponik, murguywet	*protect our trees (mosotig, poponik…)*
Ripwech moingonigochog po mogonjog	*protect our hives of hardwood cedar*
Konech keldop kugo nimokinochiy	*give us the footstep of our forefathers that had success*
Tororo rip kotop ogiot	*god protect the house of ogiot*
Tororo tomoyon KOTOP SOGOT	*god bless our house of leaves*
Sere! Sere! Sere! Sere!	*Let it be well! Let it be well! Let it be well!* *[the word sere depicts overall goodness]*

Lines from an Ogiek prayer sung while walking in the forest on honey-hunting (Samorai Lengoisa 2015).

5.7 References

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Chapter 6: Responses to risks and opportunities associated with pollinators and pollination

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Executive Summary

Loss of diversity of wild pollinators is a worldwide problem that generates risks for food production and society (*established but incomplete*). There is evidence from some parts of the world that it is associated with crop pollination deficits at local scale, loss of wild plant diversity, and loss of distinctive ways of life, cultural practices and traditions. There is global evidence of greater crop yield instability in insect-pollinated crops than in those that don't require pollination or are wind-pollinated (*well established*). These risks are largely driven by changes in land cover and agricultural management systems, including pesticide use (*established but incomplete*) (6.2.1).

Many responses are available that can reduce these risks of pollination deficit in the short term, including land management to conserve pollinator resources, decreasing pollinator exposure to pesticides, and improving managed pollinator techniques (*well established*). These include technical, knowledge, legal, economic, social and behavioural responses that are available in literature and in the traditions of people around the world (6.4).

Modifying farming practices can benefit pollinators on farms (*well established*). Retaining or creating patches of vegetation, including small areas (e.g. patches that are only meters across) helps to retain pollinator species in agricultural areas (*well established*). For example, planting flower strips near pollinator-dependent crops increases local numbers of foraging pollinating insects (*well established*) and improves yields through increased pollination (*established but incomplete*). However, potential negative impacts, through increased exposure to pesticides when pollinator numbers are concentrated in field margins, have not been explored (*inconclusive*). Due to a lack of long-term data, there is no direct evidence yet that these responses lead to long-term increases, or stabilise pollinator populations (*inconclusive*).

Protection of larger areas of semi-natural or natural habitat (e.g., tens of hectares or more) helps to maintain pollinator habitats at regional or national scales (*established but incomplete*), but will not directly support agricultural pollination in areas that are far (> a few kms) from large reserves because of the limited flight ranges of crop pollinators (*established but incomplete*). Enhancing connectivity at the landscape scale, for example by linking habitat patches (including with road verges), may enhance pollination of wild plants by enabling movement of pollinators (*established but incomplete*), but its role in maintaining pollinator populations remains unclear. Theory and observations for other taxa suggest that when the amount of natural habitat in the landscape declines below approximately 20%, pollinator populations are at risk of becoming isolated and connectivity may play an important role in their conservation (6.4.3.1.1, 6.4.3.1.2, 6.4.5.1.6).

Organic farms support more species of wild pollinators than non-organic farms, but evidence comes mostly from Western Europe and North America (*well established*). Pollination to crops are also enhanced on organic farms (*established but incomplete*). Increases in wild pollinators are less likely to occur in response to organic farming in landscapes that are already rich in non-farmed habitats (*well established*). There is some evidence that high-yielding organic farms do not support more pollinators, which suggests that the differences usually seen between organic and conventional farms are not related to the organic status per se but to specific strategies practiced on some organic farms (*established but incomplete*) (6.4.1.1.4).

Schemes that offer farmers short-term payments for prescribed environmental management – called agri-environment schemes – can include actions known to increase numbers of foraging pollinators, or pollinator species, on land under the scheme (*well established*). For example, organic farming, and planting or retaining flower-rich habitat, are supported under many European agri-environment schemes. Financial support for such activities is important, when these activities invoke labour and opportunity costs to landholders (*well established*) (6.4.1.1, 6.4.1.3).

Three complementary strategies are envisaged for producing more sustainable agriculture that address several important drivers of pollinator decline: ecological intensification, strengthening existing diverse farming systems and investing in ecological infrastructure. These strategies concurrently address several important drivers of pollinator decline by mitigating against impacts of land use change, pesticide use and climate change. The policies and practices that form these strategies have direct economic benefits to people and livelihoods in many cases (*established but incomplete*). This is in contrast to some of the options for managing immediate risks, such as developing crop varieties not dependent on pollination, which may increase vulnerability to pests and pathogens due to reduced crop genetic diversity (*inconclusive*) (6.2.2, 6.9, 6.4.1.1.8, 6.4.1.1.12, 6.4.2.1.2, 6.4.4.1, 6.4.4.3, 6.9).

Strategies to adapt to climate change may be necessary to secure pollination for agriculture in the long term (*established but incomplete*), although the impacts of ongoing climate change on pollinators and pollination services and agriculture may not be fully apparent for several decades owing to delayed response times in ecological systems (*well established*). Adaptative responses to climate change include increasing crop diversity and regional farm diversity, and targeted habitat conservation, management and restoration. The effectiveness of these strategies at securing pollination under climate change is untested and likely to vary significantly between and within regions (*inconclusive*) (6.4.1.1.12, 6.4.3.1.2, 6.4.4.1.5, 6.5.1.10.2, 6.8.1).

Non-agricultural lands, both urban and rural, hold large potential for supporting pollinators, if managed appropriately. Increasing the abundance of nectar and pollen-providing flowering plants in urban or peri-urban green spaces such as parks, sport fields, gardens, and golf courses increases local pollinator diversity and abundance (*established but incomplete*). Many cities actively conserve and restore natural habitat for pollinators in such spaces. Other land uses including road verges, power line corridors, railway banks, and vacant land in cities hold large potential for supporting pollinators, if managed appropriately to provide flowering and nesting resources (*inconclusive*). This has been implemented in some areas, such as parts of the United States. A few studies demonstrate increased pollinator numbers on the managed areas, and one study found road verges help maintain genetic connectivity in a bird-pollinated plant (*established but incomplete*). There are possible negative impacts from pollinators feeding on road verges, such as metal contamination, which have not been fully explored (*established but incomplete*) (6.4.5.1).

Reducing risk by decreasing the use of pesticides is a central part of Integrated Pest Management (IPM) and National Risk Reduction programs promoted around the world. Many of the practices that comprise IPM, such as mixed cropping and field margin management, have co-benefits for pollinators (*well established*). Education and training for land managers, farm advisers, pesticide appliers and the public are necessary for the effective implementation of IPM, and to ensure correct and safe use of pesticides, in agricultural, municipal and domestic settings (*established but incomplete*). Exposure of pollinators to pesticides can also be reduced by a range of specific application practices, including technologies to reduce pesticide drift (*well established*) (6.4.1.1, 6.4.2.1.3, 6.4.2.4.2).

Risk assessment can be an effective tool for defining pollinator-safe uses of pesticides, and subsequent use regulations (including labelling) are important steps towards avoiding mis-use of specific pesticides that can harm pollinating insects (*well established*). Overall, the environmental hazard from pesticides used in agriculture is decreased at national level by risk assessment and use regulations (*established but incomplete*). Other policy strategies that can help to reduce pesticide use, or avoid mis-use, are supporting farmer field schools, which are known to increase adoption of IPM practices as well as agricultural production and farmer incomes (*well established*), and applying global codes of conduct (*inconclusive*). The International Code of Conduct on Pesticide Management of the Food and Agriculture Organization and the World Health Organization of the United Nations provides a set of voluntary actions for Government and industry to reduce risks for human health and environment; sixty-one per cent of countries surveyed (31 countries) are using the code, based on a survey from 2004 and 2005. Investment in independent ecological research on population-level effects of pesticides on pollinators in real agricultural landscapes would help resolve the uncertainties surrounding the risk of pesticides to pollinators and pollination. Risk assessments required for approval of genetically modified organism (GMO) crops in most countries do not adequately address the direct sublethal effects of insect-resistant (IR) crops or the indirect effects of herbicide-tolerant (HT) and insect-resistant (IR) crops, partly because of a lack of data. Extending monitoring and risk-indication of the environmental and biodiversity impacts of pesticides and GMOs specifically to include wild and managed pollinators (monitoring schemes exist in many countries) would improve understanding of the scale of the risks (*established but incomplete*) (6.4.1.5, 6.4.2.1, 6.4.2.4.1, 6.4.2.4.2, 6.4.2.2.6, 6.4.2.6.1, 6.4.2.6.2).

Preventing new invasions of species that harm pollinators (i.e., competitors, diseases, predators) and mitigating impact of established invaders can be more effective than attempting eradication (*established but incomplete*). There is case-study evidence of benefits to pollinator species or pollination of native plants from efforts to reduce numbers of invasive insect species in Japan and Hawaii (6.4.3.1.4).

Better regulation of the movement of all species of managed pollinators around the world, and within countries, can limit the spread of parasites and pathogens to managed and wild pollinators alike and reduce the likelihood that pollinators will be introduced outside their native ranges and cause negative impacts (*established but incomplete*). For example, Australia has strict biosecurity policy around honey bees and has avoided establishment of *Varroa* mites. Most countries have not regulated movement of managed pollinators other than honey bees (6.4.4.2). Movement regulation can also prevent or limit problems arising from pollinators being introduced outside their native range (*established but incomplete*).

While pollinator management by people has developed over thousands of years, there are opportunities for substantial further innovation and improvement of management practices (*well established*). These include better management of parasites and pathogens (*well established*); selection for desired traits (*established but incomplete*) and breeding for genetic diversity (*inconclusive*); pollinator symbionts, including both micro- (*established but incomplete*) and macro-organisms (*inconclusive*); and pollinator diet, including enhanced resource provision at the individual, colony, and landscape scales (*established but incomplete*). Development programs focusing on beekeeping skills, both for European honey bee and other species, can improve the value and benefits associated with these practices (*established but incomplete*) (6.4.4.1).

Disease and parasite pressures threaten managed pollinators (*well established*) and while a range of prevention and treatment options are available (*well established*) there are many opportunities to improve pollinator health outcomes through training, technology development and research. For example, there are no proven options for treating viruses in any managed pollinator species, but RNAi technology could provide one pathway toward such treatment (*established but incomplete*). *Varroa* mites, a key parasite of honey bees, have developed resistance to some chemical treatments (*well established*) so new treatment options are required (6.4.4.1, 6.4.4.5).

New managed pollinator species could contribute to agricultural pollination but incur a risk of disease transfer to wild populations and species invasions (*well established*). For example, the development of commercial bumble bee rearing and management has transformed the cultivation of several crops in glasshouse settings but there have been disease impacts on wild pollinators (*well established*) (6.4.4.1.8).

Long-term monitoring of wild and managed pollinators and pollination can provide crucial data for responding rapidly to threats such as pesticide poisonings and disease outbreaks, as well as long-term information about trends, chronic issues and the effectiveness of interventions (*well established*). Such monitoring would address major knowledge gaps on the status and trends of pollinators and pollination, particularly outside Western Europe. Wild pollinators can be monitored to some extent through citizen science projects focused on bees, birds or pollinators generally (6.4.1.1.10, 6.4.4.5, 6.4.6.3.4).

Strategic initiatives on pollinators and pollination can lead to important research outcomes and national policy changes (*established but incomplete*). Fundamental and applied research on pollinators can generate findings of real policy relevance, especially when the research is designed to answer questions posed by policy makers, land managers and other stakeholders (*well established*) (6.4.6.3.2, 6.4.6.2.2).

Education and outreach projects focused on pollinators and pollination that combine awareness-raising with practical training and opportunity for action have a good chance of generating real behaviour change, and there is direct evidence for this in a small number of cases (*established but incomplete*). There are very many pollinator-focused education and outreach projects around the world. Most are relatively new (within the last five years) and so effects on broader pollinator abundance and diversity might not be seen yet (6.4.5.1, 6.4.6.3.1).

Tools and methods are available to inform policy decisions about pollinators and pollination including risk assessment, cost-benefit analysis, decision support tools and evidence synthesis. All of those except evidence synthesis require further method development and standardisation (*well established*). Other available tools that are well developed but not yet used specifically for pollinators include environmental accounting and multi-criteria analysis. Maps of pollination seem useful for targeting interventions to areas according to service valuation or service supply, but available maps at national or larger scales may be unreliable, because they have not been tested to find out if they accurately reflect actual pollination of crops or wild flowers (*established but incomplete*) (6.5.14, 6.5.9).

There remain significant uncertainties regarding pollinator decline and impacts on agriculture and ecosystems (*well established*). Decisions about how to reduce risks can be improved if uncertainty is clearly recognised, characterised and communicated (*well established*). Some sources of uncertainty are unavoidable, because there is inherent unpredictability in natural ecosystems and human economies. Other sources of uncertainty, such as limited data availability, human preferences and lack of clarity about concepts, can be more easily reduced, once recognised, by increasing the accuracy of information at the appropriate scale (6.4.2.2.4, 6.6).

There are both synergies and trade-offs among pollinator-related responses and policy options (*well established*). An example of synergy is that creation and conservation of pollinator habitats can enhance wider biodiversity (*well established*), as well as several ecosystem services including natural pest control (*established but incomplete*), soil and water quality, aesthetics, and human cultural and psychological values (*inconclusive*). An example of a trade-off is that organic farming benefits pollinators, but in many (not all) farming systems, current organic practices usually produce lower yields (*well established*). This trade-off may be minimised by supporting research into ecological intensification to help enhance organic farm yields without losing the pollination benefits, or by encouraging organic farms in less-productive agricultural landscapes, where yield differences between organic and conventional agriculture are lower (*inconclusive*) (6.4.1.1.4, 6.4.1. 1.11, 6.7).

6.1 Introduction and outline

This chapter reviews possible responses to the risks and opportunities associated with pollinators and pollination. By responses, we mean actions, interventions, policies or strategies designed to support pollinators or mitigate against pollinator decline, carried out at any scale by individuals or organisations.

We first summarise what the risks and opportunities are, in section 6.2. Responses to these can be categorised in various ways. We have grouped them according to the type of response (technical, legal, economic, social/behavioural and knowledge), as explained in section 6.3.

The responses are organised by sector in section 6.4, and listed in a table for each sector, with a summary of relevant information. The sectors are agriculture, pesticides, nature conservation, pollinator management & beekeeping, and urban & transport infrastructure. Pesticides are separated from agriculture in our structure because these two areas are often separated in policy. Responses that cut across these sectors, such as broad policy initiatives, research, education and knowledge exchange, are presented in section 6.4.6. For each possible response, we identify whether it is proposed, tested or established, and summarise existing knowledge about whether the response is known to achieve its objectives, with a particular focus on its effects on pollinators or pollination.

Section 6.5 provides an overview of the tools and methods that have been used to understand and compare alternative responses. Section 6.6 examines the problem of uncertainty, and ways of accommodating it in decision making. Section 6.7 describes what is known about trade-offs between different possible responses. Section 6.8 identifies knowledge gaps. Appendix 6A describes the methods and approaches used to write this chapter, including how the list of considered responses was developed.

Public policy has a significant role in shaping and implementing responses. The development and implementation of policy over time is often described in terms of a ‘policy cycle’ (Figure 1). The ways in which scientific, indigenous and local knowledge are used during the policy cycle, and incorporated into policy, are complex and much discussed (for example, Juntti et al., 2009; Owens, 2012; Dicks et al., 2014). Relevant knowledge must be provided at the correct point in the policy cycle, if it is to be useful to policy makers, but the likelihood of its actual use also depends on economics, politics, governance and decision-making processes unique to each specific context. As a general guide, the scientific, indigenous and local knowledge reviewed in Chapters 4, 5 and 6 are most useful for policy **formulation, implementation** and **evaluation**. Knowledge from Chapters 2, 3 and 5 is most useful for **agenda setting**, which involves identifying problems that require a policy response.

Pollinators and pollination are relevant concerns in a range of policy areas, demonstrated by review of relevant legislation (Tang et al., 2007) and by discussion with policy makers (Ratamäki et al., 2011; Rose et al., 2014). The important policy areas, and the subsections of this chapter that discuss possible policy responses, are:

- Agriculture and public health (section 6.4.1)
- Pesticide regulation (section 6.4.2)
- Biodiversity and ecosystem services (section 6.4.3, services related to food crops in 6.4.1)
- Animal health and international trade (section 6.4.4)
- Transport and infrastructure (section 6.4.5)
- Climate change and energy (some responses reviewed in 6.4.1)

A number of theoretical frameworks have been proposed to help understand what drives policy change, but there is no clear overarching framework (Sabatier and Wiebel, 2013) and no specific research has examined the development of pollinator-related policies. Drawing on the examples collated in this report, scientific knowledge can be an important driver, as in the example of the Brazilian Pollinators Initiative (see section 6.4.6.2.2). On the other hand, pollinator-related policy could change or be developed in response to a combination of science, public opinion and political opportunity, as has perhaps been the case for pollinator strategies developed in the UK (section 6.4.6.2.2; Dicks et al., 2015).

Rose et al. (2014) suggest opportunities to ‘mainstream’ pollinator conservation and management in policy. ‘Mainstreaming’ means ensuring that impacts of policies on pollinators and pollination are considered during policy formulation and implementation in all relevant sectors (Maes et al., 2013). The Sustainable Development Goals (http://www.un.org/sustainabledevelopment/sustainable-development-goals/), the Convention on Biological Diversity (www.cbd.int) and the Committee on World Food Security (http://www.fao.org/cfs/cfs-home/en/) are highlighted as opportunities to mainstream consideration of pollinators and pollination. The Aichi targets of the Convention on Biological Diversity (www.cbd.int/sp/targets/) also demand incorporation of pollinators and pollination into policy. Target 2 on integrating biodiversity values in strategies and processes, Target 7 on sustainable agriculture and Target 14 on restoring and safeguarding ecosystem services are particularly relevant to pollinators and pollination.

64. Figure 1: A simplified representation of the 'policy cycle', the iterative decision-making process by which public policy is developed and revised.
Local stakeholders, particularly local people and businesses, are involved at every stage. See text for a discussion of how scientific and local and indigenous knowledge are incorporated.

6.2 Summary of risks and opportunities associated with pollinators and pollination

We take a scientific-technical approach to risk, from a realist and individual-level perspective. This assumes that the risks are real, and they are perceived and responded to independently by individuals, with no consideration of cultural factors or social norms. From this perspective, a risk is usually understood as the probability of a specific hazard or impact taking place. A common way to evaluate a risk is to estimate both the probability and the size or scale of the impact. We have not considered sociological or psychological understandings of risk (Taylor-Gooby and Zinn, 2006). While the cultural framing of risk perceptions and responses is clearly important in the context of pollinators and pollination, we did not find any research or relevant knowledge that would allow us to evaluate its influence critically.

An opportunity is a time or set of circumstances that make it possible to do something. The clearest opportunities associated with pollinators and management of pollination arise when there are direct economic benefits to taking action.

The potential impacts and opportunities listed in Table 6.2.1 have been defined through deliberation and discussion among the report authors (including Chapters 1 to 5).

A risk assessment for the economic, social and environmental impacts of pollinator decline would require both the probability and the scale or magnitude of each of the impacts listed in Table 6.2.1 to be assessed, and preferably quantified in some way. Given the substantial knowledge gaps regarding the status, trends and drivers of change in pollinators in most regions of the world (see Chapters 2 and 3), this has not been possible. Here we provide a brief overview of what is known about the risks posed by the direct impacts.

31. Table 6.2.1. A summary of the main potential impacts of pollinator decline, and opportunities associated with pollinators and pollination.

POTENTIAL IMPACTS OF POLLINATOR DECLINE	**OPPORTUNITIES** CREATED BY SUSTAINABLE MANAGEMENT OF POLLINATORS AND POLLINATION
PRODUCTION OF FOOD (AND OTHER PRODUCTS)	
DIRECT IMPACTS ON FOOD PRODUCTION Crop pollination deficit leading to lower quantity or visual/nutritional quality of food (and other products, such as fibre, fuel or seeds). Crop yield instability due to loss of pollinators or change in pollinator communities. Fall in honey production (and other hive products) due to declining honey/stingless bee numbers. Decline in long-term resilience of food production systems. Decline in yields of wild fruit, harvested from natural habitats by local communities. Reduced availability of managed pollinators. **INDIRECT IMPACTS ON FOOD PRODUCTION** Decline in dairy and meat production due to decline in forage quality (includes cattle feeding on sown clover or soya forage, for example, or camels browsing on legumes).	Improved or more stable yield in the long term, at lower cost. Reduced dependence on managed pollinators due to more reliable pollination delivery by natural ecosystems. Reduced financial risk due to diversified income streams through more crop types. Product premium from a more sustainable approach to farming or beekeeping. Increased production of good quality honey and other bee products. Enhancement of other ecosystem services, particularly natural pest regulation/biocontrol. More economically sustainable agriculture for the long term (for example, a more diverse pollinator community enables a broader range of responses to climate or other environmental change).

POTENTIAL IMPACTS OF POLLINATOR DECLINE	**OPPORTUNITIES** CREATED BY SUSTAINABLE MANAGEMENT OF POLLINATORS AND POLLINATION
Decline in nutritional quality of human diets (vitamin content, etc.) due to increasing prices or falling quality of animal-pollinated food products and honey. Price changes and changes in demand, in response to yield changes. More land conversion required as yields decline. Loss of income/livelihoods for growers of pollinator dependent crops.	
BIOCULTURAL DIVERSITY	
DIRECT BIOCULTURAL DIVERSITY IMPACTS Loss of wild pollinator diversity. Loss of wild plant diversity due to pollination deficit. Loss of aesthetic value, happiness or well-being associated with wild pollinators or wild plants dependent on pollinators. Loss of distinctive ways of life, cultural practices and traditions in which pollinators or their products play an integral part. **INDIRECT BIOCULTURAL DIVERSITY IMPACTS** Increased disease incidence in wild and managed pollinator populations. Increased incidence and spread of invasive species due to transport of pollinators by humans. Ecosystem instability due to loss of plant-pollinator interactions (includes, for example, reduced availability of food for other animals due to lack of fruits and seeds).	Maintenance of wild pollinator and plant diversity. Improved conditions and habitats for other species (entire ecological communities). Decreased risk of long-range disease transfer and invasion by non-native species. Maintenance of aesthetic value, happiness or well-being associated with wild pollinators or wild plants dependent on pollinators. Maintenance of distinctive ways of life, cultural practices and traditions in which pollinators or their products play an integral part. Maintenance of pollinators as biological resources for research (for example, to develop medicines based on bee products, or aerial robots based on bee flight).

POTENTIAL IMPACTS OF POLLINATOR DECLINE	**OPPORTUNITIES** CREATED BY SUSTAINABLE MANAGEMENT OF POLLINATORS AND POLLINATION
Decreased economic or dietary self-sufficiency of indigenous peoples leading to loss of sovereignty. Loss of biological resources for research (for example, medicines based on bee products, or aerial robots based on bee flight).	

6.2.1 An overview of direct risks associated with pollinator decline

Table 6.2.2 summarises the evidence included in this assessment for each of the direct impacts listed in Table 6.2.1, including whether and where the impact is known to be happening. Based on this information, we categorise the direct impacts into those that pose an immediate risk to people and livelihoods at least somewhere in the world (**immediate risk**), those that do not pose an immediate risk but could develop in the longer term (**future risk**), and those for which we do not have sufficient knowledge to assess the risk, even conceptually (**unknown**).

6.2.1.1 Linking risks to drivers

Table 6.2.3 shows the main drivers associated with the risks identified. The drivers listed are those most frequently selected as one of the ‘two or three main drivers’ by the Lead Authors and Co-ordinating Lead Authors, in an anonymous individual consultation exercise. Of the drivers discussed in Chapter 2, changes in land cover and spatial configuration (2.1.1), land management (2.1.2), and pesticides (2.2.1) are the most prominent drivers of risks associated with pollinator decline.

Kuldna et al. (2009) also found that land use practices and agrochemicals were regarded as the most significant pressures on pollinators, using a combination of literature review and expert judgement.

6.2.1.2 Other perspectives on risk

A report by the International Risk Governance Council (IRGC, 2009) identified a number of barriers, or ‘governance deficits’ that prevent effective governance of the risks related to pollination. These barriers can be summarised as: scientific uncertainty, lack of economic mechanisms, inadequate land use policies, inadequate stakeholder consultation, and lack of long-term planning. All these barriers persist to some extent, but this chapter demonstrates progress towards reducing them. Research funding has reduced scientific uncertainty (section 6.4.6), there are examples of stakeholder participation and communication around the world (6.4.1, 6.4.4, 6.4.6 and 6.5), and a range of economic methods and mechanisms have been developed, and tested or established in some regions (Chapter 4 and Section 6.5.1.5).

In 2014, the global asset management firm Schroders Investment Management Ltd. published a report on the economic and corporate significance of pollinator decline (Stathers, 2014). The report provides an insight into global business perceptions of the first two food production impacts in our list. According to the report, pollinator decline is likely to affect cash flow for some companies with exposure to agricultural produce, due to impacts on raw material prices, but it concludes that pollinator decline is more significant at national and farm levels than at the level of the global economy.

6.2.2 Opportunities to benefit pollinators and improve pollination

It is beyond the scope of this report to review evidence for the social or economic benefits that underlie many of the opportunities listed in Table 6.2.1. However, evidence for the likelihood of some of these opportunities comes from what we know about the effectiveness of the responses, and is described in the rest of this chapter.

Section 6.4.1, *Agriculture, horticulture and forestry practices*, compiles what is known about the likelihood of improved or more stable yields, reduced reliance on managed pollinators, diversified income and premium prices, and more economically sustainable agriculture in the long term, following action on pollinators. Section 6.4.2 *Pesticides and pollutants* provides information on reduced environmental hazards associated with agriculture, which could contribute to maintaining wild pollinator and plant diversity, and generate improved conditions and habitats for other species. Section 6.4.3 *Nature conservation* discusses the likelihood that better biodiversity conservation overall is associated with pollinator management. Section 6.4.4, *Pollinator management and beekeeping*, discusses what is known about the likelihood of increased production of honey and bee products from better management of pollinators. Finally, section 6.7 *Trade-offs and synergies in decisions about pollination*¸ discusses the evidence on whether mitigating pollinator decline and active management of pollination enhances other ecosystem services through synergy.

We can also use this assessment to identify responses that have been established and shown to be effective. These may represent opportunities to act in other places or contexts, if there are appropriate resources available, and suitable openings in the policy cycle. These responses are shown in bold, in Table 6.9.1.

32. Table 6.2.2. Summary of available information on the nature, magnitude and scale of direct impacts from Table 6.2.1.

Sections of the report where more information can be found are given in brackets.

Direct impact	**Evidence from this assessment**	**Immediate, future or unknown risk**
Crop pollination deficit leading to lower quantity or quality of food (and other products)	• Decreased crop yield relates to local declines in pollinator diversity, but this trend does not scale up globally (3.8). For example, pollen limitation has been shown to reduce cacao yields greatly on farms in Indonesia (2.2.2.2.4), and hand pollination is required in apple orchards of Maoxian County, China. (2.2.2.1.9) • Globally, yield growth of pollinator-dependent crops has not slowed relative to pollinator-independent crops over the last five decades (1961-2007) (3.8).	Immediate
Crop yield instability	• Globally, pollinator-dependent crops show less stable yields than non-pollinator-dependent crops (3.8).	Immediate
Fall in honey production (and other hive products)	• Globally, honey production has been increasing for the last five decades, although growth rates vary among countries. (3.3.2)	Future
Decline in long-term resilience of food production systems	• Global agriculture is becoming increasingly pollinator-dependent and the proportion of agricultural production dependent on pollinators has increased by >300% during the last five decades (3.7). • There is no specific evidence of changes in resilience of food production systems in response to pollinator decline.	Future
Decline in yields of wild fruit, harvested from natural habitats by local communities	• Our assessment contains no specific evidence for this.	Unknown
Reduced availability of managed pollinators	• The number of managed honeybee hives is increasing at the global scale, although undergoing declines in some European countries and North America (3.3.2). • The stock of domesticated honey-bee hives is growing at a much lower rate than growth in demand for pollination. Shortages of honey bee hives for crop pollination are apparent in some countries (UK, USA and China) (3.8.2). • Commercial management of a few species of bumble bee as pollinators, particularly for fruit crops, has increased dramatically since the 1980s, with an estimated 2 million colonies traded annually around the world (3.3.3). • A few other solitary bee and other pollinator species are traded around the world. There are clear opportunities to develop further species for commercial management (3.3.5, 6.4.4.1.3).	Immediate
Loss of wild pollinator diversity	• Wild pollinators are declining in abundance, species occurrence, and diversity at local and regional scales, although evidence comes mostly from NW Europe and North America. At larger spatial scales, declines in bee diversity and shrinkage of geographical ranges, e.g., of bumble bees, have been recorded in highly industrialized	Immediate

Direct impact	**Evidence from this assessment**	**Immediate, future or unknown risk**
	regions of the world, particularly Europe and North America, over the last century (3.2.2).	
Loss of wild plant diversity due to pollination deficit	• Local declines in pollinator abundance and diversity have been linked to decreasing trends in wild plant pollination and seed production in habitat fragments, and to declines in the diversity of pollinator-dependent wild plant species at regional scales (3.2.2).	Immediate
Loss of aesthetic value, happiness or well-being associated with wild pollinators or wild plants dependent on pollinators	• Pollinators are a source of multiple benefits to people, contributing to medicines, biofuels, fibres, construction materials, musical instruments, arts and crafts, and as sources of inspiration for art, music, literature, religion and technology. Loss of wild and managed pollinators will ultimately erode these benefits, but there is no specific evidence of this loss taking place yet (5.2.3, 5.2.4).	Future
Loss of distinctive ways of life, cultural practices and traditions in which pollinators or their products play an integral part	• There is a loss of indigenous and local knowledge and sustainable bee management practices within local communities. Indigenous local knowledge from Mexico suggests that numbers of stingless bee colonies and traditional meliponiculture practices are declining (3.3.4). • Shifts in social systems, cultural values, and accelerated loss of natural habitats have been associated with a decrease in the transfer of knowledge within and between generations. This has led to a decline in stingless bee husbandry in the Americas and Africa, and changes in habitat management for wild honey bee species in Asia by local and indigenous communities (3.9).	Immediate

33. Table 6.2.3. Linking direct risks to drivers and responses.
This table shows the drivers most frequently selected by the Lead Authors and Coordinating Lead Authors as one of the 'two or three main drivers' for each direct impact from Table 6.2.1, in an anonymous individual consultation exercise. It does not list all possible drivers for each impact, but indicates those for which there is strongest support.

Risk	**Main drivers (relevant section)**	**Responses described in section:**
Crop pollination deficit leading to lower quantity or quality of food (and other products)	Changes in land cover and spatial configuration (2.2.1) Land management (2.2.2) Pesticides (2.3.1)	6.4.1 Agriculture 6.4.2 Pesticides 6.4.3 Nature Conservation 6.4.4 Pollinator management and beekeeping 6.4.6 Policy, research and knowledge exchange across sectors
Crop yield instability	Changes in land cover and spatial configuration (2.2.1) Land management (2.2.2) Pesticides (2.3.1)	6.4.1 Agriculture 6.4.2 Pesticides 6.4.3 Nature Conservation 6.4.6 Policy, research and knowledge exchange across sectors
Fall in honey production (and other hive products)	Pesticides (2.3.1) Pollinator parasites and pathogens (2.4)	6.4.2 Pesticides 6.4.4 Pollinator management and beekeeping
Decline in long term resilience of food production systems	Changes in land cover and spatial configuration (2.2.1) Land management (2.2.2) Pesticides (2.3.1) Climate change (2.6)	6.4.1 Agriculture 6.4.2 Pesticides 6.4.3 Nature Conservation 6.4.6 Policy, research and knowledge exchange across sectors
Decline in yields of wild fruit, harvested from natural habitats by local communities	Changes in land cover and spatial configuration (2.2.1) Land management (2.2.2) Pesticides (2.3.1) Pollinator parasites and pathogens (2.4) Climate change (2.6)	6.4.1 Agriculture 6.4.2 Pesticides 6.4.3 Nature Conservation 6.4.4 Pollinator management and beekeeping 6.4.6 Policy, research and knowledge exchange across sectors
Loss of wild pollinator diversity	Changes in land cover and spatial configuration (2.2.1) Land management (2.2.2) Pesticides (2.3.1)	6.4.1 Agriculture 6.4.2 Pesticides 6.4.3 Nature Conservation 6.4.5 Urban and transport infrastructure 6.4.6 Policy, research and knowledge exchange across sectors
Loss of wild plant diversity due to pollination deficit	Changes in land cover and spatial configuration (2.2.1) Land management (2.2.2) Invasive alien species (plants and animals) (2.5)	6.4.1 Agriculture 6.4.3 Nature Conservation 6.4.5 Urban and transport infrastructure 6.4.6 Policy, research and knowledge exchange across sectors

Risk	Main drivers (relevant section)	Responses described in section:
Reduced availability of managed pollinators	- Pesticides (2.3.1) - Pollinator management (includes transport of managed pollinators) (2.4.2)	6.4.2 Pesticides 6.4.4 Pollinator management and beekeeping
Loss of aesthetic value, happiness or well-being associated with wild pollinators or wild plants dependent on pollinators	- Changes in land cover and spatial configuration (2.2.1) - Land management (2.2.2)	6.4.1 Agriculture 6.4.3 Nature Conservation 6.4.5 Urban and transport infrastructure 6.4.6 Policy, research and knowledge exchange across sectors
Loss of distinctive ways of life, cultural practices and traditions in which pollinators or their products play an integral part	- Changes in land cover and spatial configuration (2.2.1) - Land management (2.2.2)	6.4.1 Agriculture 6.4.3 Nature Conservation 6.4.6 Policy, research and knowledge exchange across sectors

6.3 Typology of responses

Responses can be classified according to: the driver or threat generating a need for action (e.g., habitat loss, pesticides), the actors taking the action (from private individuals to intergovernmental institutions), the type of action (e.g., policy, financial, etc.) or the scale of impact (international, regional, etc.). Most sets of responses could be variously classified according to all these different classifications, and there is no right way, but there is usually a way that seems most logical and informative for a particular subject.

Previous attempts to classify responses relating to ecosystem services include the Millennium Ecosystem Assessment (Chopra et al., 2005), the UK National Ecosystem Assessment (UK NEA; Brown et al., 2014), and a recent policy analysis carried out by the Food and Agricultural Organization of the United Nations, which classified policy responses for pollinators into six themes (FAO; Rose et al., 2014).

After reviewing these typologies, we decided classifying by type of action is the most straightforward way to group responses for pollinators and pollination. Classifications based on actors, scales or threats were less useful, as many responses involve several actors working together, operate at several scales or respond to many possible threats.

For our action-based typology, we adapted the Millennium Ecosystem Assessment model (MEA, 2011), including their technological, legal, economic and social/behavioural categories, and modifying their cognitive category to one that included not only research and indigenous and traditional knowledge, but also education and awareness-raising (see definitions in Box). Our definitions were informed also by the NEA and FAO reports.

The six thematic policy areas identified by the FAO exercise (Rose et al., 2014) are listed in Table 6.3.1. These were identified by policymakers and scientists from eleven, predominantly developing countries, as a set of successful approaches for decision makers to support. We did not use them to structure our chapter, because they represent a mix of policy sectors (e.g., pesticides, nature conservation) and action types (e.g., economic, social/behavioural and knowledge). Table 6.3.1 shows where in this chapter relevant information can be found.

6.3.1 Combining and integrating responses

A central challenge when organising and categorising responses is that sets of individual actions are often combined together in management systems, strategies or policies, but scientific research tends to test individual management actions in isolation. In this report, we compile what is known about the effects of integrated responses that cut across sectors in section 6.4.6. In the preceding sections we include combined, system-level responses where several actions within a single sector are carried out together, if they are commonly proposed or established (for example, ‘agri-environment schemes’, ‘diversified farming systems’, or ‘Integrated Pest Management’).

BOX 6.1: Types of response

TECHNICAL. These responses are tools and procedures that people use to manage pollinators or pollination, or land management approaches that could benefit pollinators. For example, they include farming or agroforestry practices such as organic farming and crop rotation (section 6.4.1), techniques to reduce the impact of pesticide use (6.4.2), creation or restoration of pollinator habitat (6.4.3) and methods of bee disease control (6.4.4).

LEGAL. These responses are mandatory rules at international, national and regional levels (‘hard’ law) and also non-legally binding treaties, guidelines, standards and codes of practice developed by law-making institutions (‘soft’ law). For pollinators and pollination, the responses include habitat or species protection through conservation designations, and controlling imports of non-native species, for example.

ECONOMIC. These responses are financial or economic actions either to either punish bad practices or provide economic incentives for good practices, related to pollinators. They include, for example, taxes on pesticides that increment their costs and reduce the benefits for the farmers (6.4.2), incentive payments to farmers for pollinator-friendly practices (6.4.1), and markets instruments such as payments for ecosystem services (6.4.3).

SOCIAL/BEHAVIOURAL: These responses focus on the informal institutions, governance and decision-making processes that shape people's choices. They include participatory processes to involve communities in decision-making (not the same as involving communities in research and knowledge gathering), adaptive management of native habitats, and voluntary codes of practice generated by community, consumer or industry groups rather than by law-making institutions.

KNOWLEDGE. Knowledge responses include actions that generate new knowledge and actions that transfer or share knowledge among groups of actors. They cover scientific research and monitoring, as well as documenting and sharing indigenous and local knowledge. They also include education, outreach, knowledge exchange and collaborative research activities. These are distinguished from social and behavioural actions because they focus on the communication or transfer of knowledge, rather than on decisions, actions and behaviour.

34. Table 6.3.1. Thematic areas for action identified by the FAO
(Rose et al. 2014).

FAO thematic area	IPBES report section
Pollinator-friendly Pesticide Policies	6.4.2 Responses to reduce impacts of pesticides
Conservation and Enhancement of Pollinator Habitats	6.4.3 Responses for nature conservation
Valuation, Incentives, and Payments for Ecosystem Services	6.4.2, 6.4.3, 6.4.4, 6.4.5 Economic responses (most well-developed in agriculture)
Participation, Knowledge-Sharing and Empowerment of Rural and Indigenous Peoples and Local Communities	6.4.1, 6.4.3, 6.4.4 Social and behavioural responses
Collaborative Research and Outreach	6.4.6 Knowledge responses
Public Awareness Raising and Knowledge Sharing	6.4.6 Knowledge responses

6.4 Options to restore and strengthen pollination

This section reviews responses in each sector that have been **proposed** in response to evidence of drivers, status and trends in pollinators (see Chapters 2 and 3 for information about drivers, status and trends).

Then we ask which, if any, have been **tested** or are already **established**, drawing on Indigenous and Local Knowledge in addition to scientific knowledge.

There is a subsection for each of five main sectors: a) agriculture, b) pesticides, c) nature conservation, d) pollinator management and beekeeping and e) urban and transport infrastructure; Subsection f) covers integrated responses that involve actions in more than one sector.

Responses are grouped according to the type of response (see section 6.3). Evidence relating to the opportunities described in section 6.2 is identified with summary statements where possible.

For each chosen response or category of response, we reviewed what is known about its **effects** on pollinators, pollination or any other measures or outcomes that relate to the risks and opportunities discussed in section 6.2.

6.4.1 Agricultural, agro-forestry and horticultural practices

This section focuses on agricultural practices, and adaptive techniques to enhance pollinator and pollination and to maintain yields in the wake of pollinator decline. These agricultural practices are commonly applied to mitigate negative impacts of agriculture, such as those identified in Chapter 2.

6.4.1.1 Technical responses

6.4.1.1.1 Conserve or sow field margins within or around crops

There is considerable evidence indicating the potential of non-crop areas within agricultural landscapes, including flower strips, permanent grassland, sown grassland, buffer strips, managed hedgerows (Kremen and M’Gonigle, 2015), set-aside fields (Greaves and Marshall, 1987), for enhancing pollinator diversity in agroecosystems (Morandin and Kremen, 2013; Garibaldi et al., 2014). These practices can benefit pollinator richness by providing suitable food and nesting resources within and across arable farms without changing cropping patterns (Nicholls and Altieri, 2013). We know of no evidence for population-level effects on pollinators, although some studies indicate that numbers of bumble bee reproductives (males or males and queens) tend to increase as flowers are added to a landscape (Williams et al., 2012, Carvell et al., 2015). Far less is known about which plant species are beneficial for bees and other pollinators in terms of quality of nectar and pollen (see section 6.8.1).

A recent review (Dicks et al., 2014) found 65 studies in Europe that focused of the effect of sown flower strips on invertebrates; 41 of the studies identified positive effects on number, diversity, or activity of invertebrates. Strips sowed with flowers, particularly those rich in nectar or pollen, support higher insect abundances and diversity than cropped habitats or other field margin types such as sown grass margins and natural regeneration (Carvell et al., 2007; Scheper et al., 2013). However, the effectiveness of these small-scale practices varied with (1) the magnitude of increase in flowering plant cover resulting from the practices, (2) farmland type, and (3) landscape context (Scheper et al., 2013). It is possible that flowering resources placed alongside crop fields increase exposure of pollinators to pesticides, however, this hypothesis has not been tested (see section 2.2.1 for a discussion of possible exposure routes).

Regional programs to increase the quality and availability of seeds from native flowering plants are important for the success of these practices (Isaacs et al., 2009). Operation Pollinator, a programme to boost numbers of pollinating insects on farms and golf courses across Europe, run by the agri-chemical company Syngenta, has developed and tested seed mixtures to provide to land managers (http://www.operationpollinator.com/).

Although some of the above studies have shown direct benefits of wildflower strips in terms of increased pollinator richness, abundance and activity on crops, there is limited evidence about the direct impact of those practices on crop yield. One study showed that floral strips surrounding crops modify the level of outcrossing within the cultivar, consequently affecting the genetic structure of the cultivar (Suso et al., 2008).

Some studies demonstrate that habitat enhancements can provide increased pollination to adjacent crops. One example of such a study was on mango production in South Africa showing that pollination was improved by planting small patches of perennial plants (Carvalheiro et al., 2012). Similar results were found in USA for blueberry, where pollination was improved after three years by the establishment of wildflower patches (Blauw and Isaacs, 2014).

Many examples of small-scale farmers maintaining habitat elements such as hedgerows and fallow areas for pollinators can be found around the world (see section 5.3.3), and there are reports from other countries of the effectiveness of these practices for increasing yields for other crops (FAO, 2008).

6.4.1.1.2 Provide nesting resources

Artificial or natural substrates, such as reed internodes and muddy spots for cavity nesters, and bare ground for soil nesters, can be enhanced at crop edges without requiring much crop area. This practice can promote the recruitment of certain bee species (Steffan-Dewenter and Schiele, 2008) and pollinator density on crops (Junqueira et al., 2013). Strategic placements of nesting cavities where abundant floral resources occur have been observed to increase population growth of pollinators (Oliveira-Filho and Feitas, 2003). Evidences that such practices lead to greater yields are few, but there are examples that such management practices increase population growth of pollinators (MacIvor and Packer, 2015). The introduction of bamboo nests for bees of the genus *Xylocopa* in Brazilian passion fruit plantations increased the yield by 781% (Camillo, 1996). In apple orchards in Canada, habitat management and placement of cavity nests for Osmiine bees resulted in increased offspring of the Osmiine bees (Sheffield et al., 2008).

6.4.1.1.3 Sow mass-flowering crops and manage the timing of blooming

Some mass-flowering crops when grown in diverse farming systems could be managed to bloom in different periods of time at a landscape scale. In Sweden, bumble bee reproduction was improved in landscapes with both late-season flowering red clover and early-season mass-flowering crops (Rundlöf et al., 2014). But the short duration of floral availability, low diversity of resources, insecticide application, and tillage may limit the capacity of mass flowering monocultures to support wild pollinator populations on their own (Vanbergen and the Insect Pollinators Initiative, 2013). In addition, studies have found strong evidence for food resource availability regulating bee populations (Roulston and Goodell, 2011) and also have revealed the critical role of resource availability on bee health (Alaux et al., 2010). Thus in heterogeneous landscapes rich in flowering species, sowing mass flowering crops can be an alternative practice to enhance wild pollinators and pollination (Holzschuh et al., 2013; Bailes et al., 2015), but more work is needed to define how this should be done.

6.4.1.1.4 Organic farming

Pollination benefits of organic practices were found in some crops such as strawberries in Sweden (Andersson et al., 2012) and canola in Canada (Morandin and Winston, 2005). Organically-farmed fields can enhance bee abundance, richness and diversity compared to conventionally-farmed fields, and also help to sustain pollination by generalist bees in agricultural landscapes (Tuck et al., 2014), but the magnitude of the effect varies with the organism group and crop studied, and is greater in landscapes with high proportions of cultivated lands (Holzschuh et al., 2007; Kennedy et al., 2013). However, the studies have been carried out mainly in Europe and North America and their applicability to other areas of the world is uncertain.

A large-scale study in ten European and two African countries showed that organic farms have much smaller effects on the diversity of habitats or species richness at farm and regional scales than at the field scale. This implies that to ensure positive benefits of biodiversity at larger spatial scales, even organic farms have to support biodiversity actively by maintaining and expanding habitats and natural landscape features (Schneider et al., 2014).

In England, a study suggested that organic farming should be mainly encouraged in mosaic (low productivity) landscapes, where yield differences between organic and conventional agriculture are lower. In less-productive agricultural landscapes, biodiversity benefit can be gained by concentrating organic farms into hotspots without a commensurate reduction in yield (Gabriel et al., 2013). This study also revealed a decrease in the abundance and diversity of some pollinator groups with increasing yield in both organic and non-organic ("conventional") wheat farms. The factors that co-vary with yield ultimately influence this pattern, and could include management practices, and management of habitats and/or cropping systems, in both conventional and organic farms.

6.4.1.1.5 No-till farming

No-till farming is a practice for soil conservation that can reverse long-term soil degradation due to organic matter loss. No-till farming has increased in the Cerrado region of Brazil from 180,000 hectares in 1992 to 6,000,000 hectares in 2002. Producers have found that no-till techniques within certain planting sequences each year, as well as longer-term crop rotations, may increase production by 10%. The estimated annual benefits of adopting no-till agriculture techniques in Brazil amount to $1.4 billion on 35% and $3.1billion on 80% of a total cultivated area of 15.4 million hectares (Clay, 2004). In contrast a global meta-analysis across 48 crops and 63 countries showed that overall no-till reduces yields, but this depends on the system. Yield difference is minimised when no-till is combined with crop residue retention and crop rotation, and no-till significantly increases rainfed crop productivity in dry climates (Pittelkow et al., 2015; see Chapter 2, section 2.2.2.1.3 for more details).

No-till coupled with the use of cover crops might be expected to enhance populations of ground-nesting bees, as many species place their brood cells < 30 cm below the surface (Roulston and Goodell, 2011; Williams et al., 2010), but there is little evidence for this. One study found an increase in squash bees *Peponapis pruinosa*, but not other bee species, on no-till squash farms in the USA (Shuler et al., 2005), while another study did not find this effect (Julier and Roulston, 2009).

6.4.1.1.6 Change irrigation frequency or type

Although there is little evidence, similarly to no-till, changing irrigation frequency or type can be a pollinator-supporting practice. In arid irrigated systems, changing from flood irrigation that may be detrimental for pollinators because of nest flooding, to drip irrigation can reduce the impact on pollinators, but in general irrigation can promote wild insect abundance through higher productivity of flowering plants or by making the soil easier to excavate (Julier and Roulston, 2009).

6.4.1.1.7 Change management of productive grasslands

Productive grasslands used for grazing or hay can be managed to be more flower-rich by reducing fertilizer inputs, or delaying mowing dates. In experimental studies in Europe, these changes usually lead to increased numbers of bees, hoverflies and/or butterflies (Humbert et al., 2012; Dicks et al., 2014a). Adding legumes and other flowering species to grassland seed mixtures is supported by some agri-environment schemes in Europe (see section 6.4.1.3) and probably benefits pollinators by supplying flowers in grassland-dominated landscapes, but this has not been clearly demonstrated (Dicks et al., 2010; Dicks et al., 2014). Two European studies have shown that avoiding use of rotary mowers and mechanical processors substantially reduces mortality of bees or butterfly larvae when cutting flowering meadows (Dicks et al., 2014b). However, studies have not been designed to look for landscape-scale, population-level effects of any of these management changes on pollinators.

6.4.1.1.8 Diversify farming systems

Diversity is the foundation of any sustainable agriculture system, and mixed crop types, crop-livestock mixtures, intercropping and cover crops bring pollinator diversity to the farm by providing floral resources and habitat for many different species of pollinators, and promote wild pollinator stability on farms (Kennedy et al., 2013). There is some evidence in Western Europe and North America suggesting that increased floral diversity achieved through diversified farming can improve pollination (Batáry et al., 2009; Kremen and Miles, 2012; Kennedy et al., 2013). Intercropping cacao with banana or plantain is correlated with an increase in the density of cacao-pollinating midges, as well as cacao fruit set, in Ghana (Frimpong et al., 2011). A recent study in Canada (Fahrig et al., 2015) suggested that reduced field size may be a more important feature of diversified farming systems than increased number of crop types, if the aim is to increase or maintain farmland biodiversity generally (including bees, hoverflies and butterflies). Recent meta-analysis suggests that two management practices that diversify crop fields – polyculture and crop rotations – increase yields in both organic and conventional cropping systems (Ponisio et al., 2015).

Diversified farming practices are an important element of the diverse cultures and practices of indigenous peoples and local communities across the globe. Scientific evidence of a benefit to pollinators or pollination in those systems is scarce but can be expected where there is increased diversity of flowering plants and habitats. For example, areas surrounding milpa systems in Central America house a wide variety of plant species that are highly attractive to insects (Lyver et al., 2015; Chapter 5, section 5.2.5.3). Indigenous Tarahumara people (Mexico) have developed an expanded cropping system that involves consuming weed seedlings (e.g., *Amaranthus*, *Chenopodium*, *Brassica*) early in the season and harvesting cucurbits, beans and maize late in the season (Bye, 1981). Similarly, small-scale farmers in the semi-arid Tehuaca´n-Cuicatla´n Biosphere Reserve (Mexico) make use of more than 90% of the 161 weed species (Blanckaert et al., 2007). Maintaining weed resources alongside local crops creates a diverse set of flowering resources for pollinators, although indigenous or rural people do not comment on the relationship between weeds or crop reproduction and pollinators (Bye, 1981; Altieri, 2003).

6.4.1.1.9 Make crops more attractive to pollinators, to enhance pollination

Spraying crops with pheromones to attract pollinators and/or enhance pollination is a well-known practice for some crops. Studies carried out in Australia (Keshlaf et al., 2013) and India (Chandrashekhar and Sattigi 2009; Nithya et al., 2012; Sivaram et al., 2013) with crop flowers sprayed with attractants significantly increased bee visitation rate, seed yield, and percent germination. In Brazil, Bee-Here^{R}, eugenol, geraniol, citral, and lemon grass extract, mainly diluted in water, were effective in attracting honeybees to sweet orange orchards (Malerbo-Souza et al., 2004).

More recently, there are ongoing studies to identify crop flower traits (e.g., brighter colours, increased scent, and increased nectar) to increase visitation by pollinators to improve the yield stability of the crop (Bailes et al., 2015).

‘Participatory Plant Breeding and Management’ is being used to develop pollinator friendly-crops that require pollinator friendly-practices (Duc et al., 2010; Suso et al., 2013). The central idea is to develop varieties to maintain open pollination, selecting flowers that can attract more pollinators. This approach aims to enhance the genetic diversity of crops, maintain pollinators and reduce chemical inputs (low-input agriculture). It requires decentralized and farmer participatory breeding methods designed to incorporate the "know-how" of farmers. There are no conclusive examples in practice yet.

6.4.1.1.10 Monitor and evaluate pollinators and pollination on farms

Systematic long-term monitoring of pollinators on farms and crop pollination deficit evaluation are still rare in literature and there are no national programmes in place. Recently FAO/GEF/UNEP has been supporting national partners in eleven countries for assessing pollinator abundance and diversity within and around crops, and for evaluation of crop pollination deficits using a standard protocol (Vaissiere et al., 2011). The projects were conducted over a five-yr-period, with studies in Brazil, Argentina, Colombia, Ghana, Kenya, Zimbabwe, India (two locations, one by an indigenous group), Nepal, Pakistan, Indonesia, and China. Results of this project, as well as of other studies can be accessed in a Special Issue on Pollination Deficits published in 2014 (volumes 12, 13 and 14) in the open Access Journal Pollination Ecology (http://www.pollinationecology.org).

More recently, a collaborative research project tested wild bees and bumble bees as part of a biodiversity indicator set at farm scale across Europe and in Ukraine, Tunisia and Uganda. The resulting toolkit is available at www.biobio-indicator.org.

6.4.1.1.11 Reduce dependence on pollinators

As global agriculture is becoming increasingly pollinator-dependent (see Chapter 3), an option to remove all the risk associated with biotic pollination is switching from dependent to non-dependent crops. This can reduce overall crop genetic diversity, thus increasing potential vulnerability to pests and pathogens (see section 6.7.1). In the USA a self-fertile variety of almond, the Independence® Almond, has been developed that needs few bees to produce numerous large nuts.

Manual or mechanical pollination can be used in high-value crops such as glasshouse tomatoes, passion fruit, kiwi or apple to compensate for deficits in pollination. In Iran, Mostaan et al. (2010) have developed a new electrical apparatus for pollinating date palms. In the absence of natural pollinators, some apple farmers in China initially adapted by using hand pollination techniques, but this has been followed by changing to fruit and vegetable crops that do not need to be cross-pollinated (Partap and Ya, 2012). However, hand pollination by human pollinators is still practiced with apples to a lesser degree, which indicates that all these farmers have yet to find satisfactory alternatives to this economically unsustainable practice (Partap and Ya, 2012).

As manual pollination represents an additional cost of production, its cost and benefits should be analysed locally. Estimates of labour costs for manual pollination of yellow passion fruit (*Passiflora edulis*), reported in studies conducted in the Brazilian states of Minas Gerais (Vieira et al., 2007) and Bahia (Viana et al., 2014), show that the cost to producers of paying workers to conduct manual pollination is equivalent to around 20% of their annual net profit.

6.4.1.1.12 Adapt farming methods to climate change

Possible adaptation strategies at the farm level include managing for a diverse pollinator community, changes in crop diversity, sowing rate, and crops/cultivars to ensure pollination in areas where pollinator populations and pollinators diversity are reduced (Reidsma and Ewert, 2008). There is evidence that biodiversity can stabilize pollination against environmental change (Rader et al., 2013). High biodiversity levels can ensure plant–pollinator phenological synchrony and thus pollination function (Bartomeus et al., 2013; Brittain et al., 2013). Greater crop diversity also can decrease crop vulnerability to climate variability, as different crops respond differently to a changing climate. But the effectiveness of adaptation efforts is likely to vary significantly between and within regions, depending on geographic location, vulnerability to current climate extremes, level of economic diversification and wealth, and institutional capacity (Burton and Lim, 2005). See section 6.4.4.1.5 for a discussion of boosting pollination by translocating native pollinators.

6.4.1.2 Legal responses

The degree to which pollination contributes to sustainable crop yields has not been addressed in agricultural policies in most countries, although China has officially recognized pollination as an agricultural input, along with other conventional inputs such as fertilizers and pesticides (FAO, 2008). At large scale, agricultural policies in Europe, (European Common Agricultural Policy (http://www.ecpa.eu/information-page/agriculture-today/common-agricultural-policy-cap) and the USA (US Farm Bill: http://www.xerces.org/wp-content/uploads/2009/04/using-farmbill-programs-for-pollinator-conservation.pdf) provide important frameworks within which specific actions to benefit pollinators have been incentivised (see section 6.4.1.3).

Most policies to increase heterogeneity in agricultural landscapes reduce intensity of land use, adopt agroecological farming practices, and prevent abandonment of agricultural land are relevant to pollinators and pollination (Smith et al., 2013). The initiative in Bhutan to eradicate chemical fertilizers and pesticides as part of its Gross National Happiness programme may have a positive impact on pollination (http://www.theguardian.com/sustainable-business/bhutan-organic-nation-gross-national-happiness-programme). Likewise, in Brazil the *National Plan for Agro-Ecology and Organic Production*, launched in 2013, with the aim to coordinate policies and actions for environmentally-friendly agriculture and organic food production may contribute to enhance pollinators and pollination (OECD, 2015). Even though the effectiveness of the regulations above is still untested, there is evidence of the positive impact of these agroecological practices on pollinators and pollination (see section 6.4.1.1). Legal responses that relate to the use of pesticides and other agrichemicals in agriculture are covered in section 6.4.2.2.

6.4.1.3 Economic responses

Financial support is often necessary to allow the farmer to switch farming practices and bear the loss in production that may result. In Europe, the USA and Australia agri-environment schemes (AES) offer farmers short-term payments for performing prescribed environmental management behaviour. Use of AES to support pollinators in Europe was reviewed by Rundlöf and Bommarco (2011), who identified three main measures that may specifically promote pollinators: creation and restoration of semi-natural habitats, establishment of flower strips, and reduction of pesticide inputs by conversion to organic farming or introduction of unsprayed field margins. Another, management of hedgerows to enhance flowering, is supported in some countries.

Effects of AES on pollinator numbers are well documented (Pywell et al., 2006; Batáry et al., 2011; http://www.conservationevidence.com/actions/700) but effects on pollinator populations are still unknown. Payment for ecosystem services (PES) is another action (e.g. Daily et al., 2009) that could promote practices to conserve pollinators on farms (see section 6.4.3.3).

More recently in the USA farmers receive financial support to diversify crops (Rose et al., 2015). The United States Department of Agriculture introduced the Whole-Farm Revenue Protection Program (http://www.rma.usda.gov/policies/wfrp.html), which offers farmers an opportunity to insure all crops on their farms simultaneously, as opposed to insuring them crop-by-crop. The lack of specific insurance programmes for fruit and vegetables in the past has been a disincentive for growers to diversify beyond commodity crops. The new way of insuring crops offers farmers enhanced flexibility and provides a greater incentive to diversify cropping systems within farming regions (USDA, 2014).

Certification schemes led by consumer or industry bodies with a price premium are a market-based instrument that can be used to encourage pollinator-friendly farm management practices. One scheme, ‘Fair to Nature: Conservation Grade’ in the UK, offers a price premium to farmers for planting flowers and managing habitat for pollinators (among other actions), as part of the licence agreement from businesses that sign up for the 'Fair to Nature' label (http://www.conservationgrade.org/conservation-farming/). One very small research project has shown that farms managed under this scheme have higher functional diversity (but not abundance) of hoverflies than conventionally managed farms (Cullum, 2014). Similar research on bees and butterflies is ongoing.

In Mexico, a proposal currently being developed is to market ‘bat-friendly mezcal’. The Mexican beverages tequila and mezcal are extracted from plants of the genus *Agave*, which are pollinated mainly by bats when they flower. Production of these drinks does not rely directly on pollination – they are extracted from vegetative parts of the plant before flowering – but agave flowers are an important food source for bats. Bat pollination is needed for seed production, which could potentially help restore agave genetic diversity for tequila production (this currently relies on clonal propagation: Colunga-GarciaMarin and Zizumbo-Villarreal, 2007; Torres-Moran et al., 2013). The Mexican endemic plant *Agave cupreata*, sometimes used for mezcal, can only be grown from seed (Martínez Palacios et al., 2011). To get this label, growers would have to leave some agave plants to flower and breed sexually through bat pollination, rather than cutting them all for production before flowering.

Financial schemes and insurance programs such as those identified above may be costly to developing countries. One alternative is where indigenous community forestry enterprises are supported by the Non-Timber Forest Products Exchange Program (NTFP-EP; http://www.ntfp.org) in South and Southeast Asia. This program empowers forestry-based communities to manage forest resources in a sustainable manner. To this end, the NTFP-EP catalyses and supports activities that strengthen the capacity of their partner organisations in their work with forest-dependent communities, particularly indigenous peoples. However, despite the great potential of this program to enhance pollinators and pollination, its efficacy is untested yet.

There is no simple relationship between financial reward and behaviour change. Payments may increase motivation, but they can also weaken motivation (Deci et al., 1999). Knowing this should make us sensitive to the way in which financial measures are applied to compensate for loss of income (Canton et al., 2009; Burton and Paragahawewa, 2011).

A recent review examining more effective instruments for changing farming social behaviour suggests switching AES for "payment by results schemes" (De Snoo et al., 2012). The latter differ from conventional agri-environmental schemes by paying farmers for outcomes rather than performing set management activities.

The intended result is that, unlike conventional schemes, farmers are encouraged to engage with conservation groups to identify common goals and to recognize the need to innovate and, in many cases, cooperate to achieve greater financial reward. There is some evidence that alternative designs for the delivery of financial rewards may also deliver environmental benefits and be associated with more enduring social and cultural changes (De Snoo et al., 2012). In Switzerland, a farmer-led initiative has successfully lobbied the government for the introduction of "bee pastures" (sown flower strips) in the national agri-environmental scheme (http://www.lobag.ch/LOBAG/Bereiche/Pflanzenproduktion/%C3%96lsaatenzuteilung/tabid/92/language/de-CH/Default.aspx)

Result-oriented schemes thus create common goals between farmers and conservationists (Musters et al., 2001), enable productivity comparisons with conventional farming products (Klimek et al., 2008; Matzdorf and Lorenz, 2010), and lead to the creation of cultural (skills and knowledge) and social capital (i.e., access to shared peer group resources) as knowledge of conservation management becomes socially valuable (Burton and Paragahawewa, 2011).

6.4.1.4 Social and behavioural responses

Conservation of ecosystem services in agricultural areas can only be effective in the long term with the active support of farming communities. Responses are required that are able not only to affect short-term changes in farmer behaviour, but also establish or re-establish group norms that will make durable changes (De Snoo et al., 2012). Effects on non-economic forms of social capital should be considered, such as how the behaviours generate status and prestige within farming communities (Burton and Paragahawewa, 2011).

For knowledge of ecosystem service conservation to have social legitimacy from the farmers' perspective, the knowledge must be generated within the farming community, rather than imposed by outsiders (De Snoo et al., 2012). Community engagement and empowerment on managing pollinators in agriculture and forestry is one broad approach to achieve this, although untested yet.

Participatory dialogue inclusive of multiple stakeholders is valuable to understand and address different perspectives and needs, and confers many benefits to policy implementation (e.g., higher-quality decisions, greater legitimacy of decisions, increased compliance (Menzel and Teng, 2009). This kind of discussion can introduce stakeholders to potential policy ideas, based on information from other regions or countries. Accounting for farmers' insights and concerns, and engaging them in change processes, is important, because they are likely to be directly impacted by laws, policies and changes to incentive schemes.

Encouraging farmers to collaborate to manage landscapes is an approach that has been tested through agri-environment schemes (see section 6.4.1.3) in some European countries (Prager, 2015). This can generate environmental, social and economic benefits, although there is no specific experience relevant to pollinators or pollination. It is more likely to be successful where there is a shared awareness among land managers of a common problem, and where schemes are flexible and can be adapted to suit local issues.

Prohibitions on behaviour, or voluntary codes of conduct, are an important social mechanism that protect and enhance pollinator presence in local communities. Farmers in Roslagen (Sweden) recognize bumble bees as important pollinators for garden and field production and afford them social protection, including restricting the cutting of trees that flower in early spring when other pollen- and nectar-producing plants are rare (Tengo and Belfrage, 2004).

6.4.1.5 Knowledge responses

Higher education and training programs for agronomists, agroecologists, veterinarians, policy-makers and farmers are important responses to support pollinators and pollination.
The Indigenous Pollinators Network promoted by the Indigenous Partnership for Agrobiodiversity and Food Sovereignty (http://agrobiodiversityplatform.org/par/2013/12/24/the-indigenous-pollinators-network/) provides a platform for scientists and indigenous people to share their ideas and best practices around pollination (see section 5.4.4.1).

Translating research into agricultural practice requires implementation, demonstration and extension work, as well as knowledge exchange between scientists and farmers, and different methodologies have been developed for promoting farmer innovation and horizontal sharing and learning (see section 6.4.6.3). In USA, the Land Grant University System, created in the mid-1800s, also provides practical knowledge and information sharing (extension), based on unbiased scientific research, to citizens everywhere, both rural and urban (National Research Council, 1995).

There are few examples where training has been demonstrated to change farmer knowledge or behaviour. The Xerces Society for Invertebrate Conservation and the US Department of Agriculture in the USA run short courses on pollinator conservation aimed at farmers and agricultural professionals. In a survey of those who participated in these short courses, 91% indicated that they would adopt bee-safe practices discussed in the course (Xerces Society, 2014), although this not does guarantee they actually did change their practice. One research project in the UK demonstrated that training farmers increases their confidence and develops a more professional attitude to agri-environmental management (Lobley et al., 2013), resulting in ecological benefits. For example, areas managed by trained farmers had more flower or seed resources and higher numbers of bees or birds than areas managed by untrained farmers (Dicks et al., 2014b).

A common approach used to transfer specialist knowledge, promote skills and empower farmers around the world is Farmer Field Schools (FFS), at which 10 million farmers in 90 countries have benefited (Waddington et al., 2014). A systematic review of FFS provides evidence that these schools are improving intermediate outcomes relating to knowledge learned and adoption of beneficial practices, as well as final outcomes relating to agricultural production and farmers' incomes (Waddington et al., 2014).

35. Table 6.4.1. Summary of evidence for responses relating to farming and agro-forestry.

Response/action (Relevant Chapter 6 section)	**Main driver(s) (Chapter 2)**	**Type**	**Status**	**Scientific evidence**
Conserve or sow field margins within or around crops (6.4.1.1.1)	Land management (2.2.2)	Technical	Established	Increases numbers of foraging pollinating insects WELL ESTABLISHED Enhances pollination ESTABLISHED BUT INCOMPLETE

Response/action (Relevant Chapter 6 section)	**Main driver(s) (Chapter 2)**	**Type**	**Status**	**Scientific evidence**
Provide nesting resources (6.4.1.1.2)	Land management (2.2.2)	Technical	Tested	Benefits to pollinator abundance and species ESTABLISHED BUT INCOMPLETE Little evidence for pollination INCONCLUSIVE
Sow mass-flowering crops and manage the timing of blooming (6.4.1.1.3)	Land management (2.2.2)	Technical	Tested	Benefits to pollinator abundance and species ESTABLISHED BUT INCOMPLETE Enhance pollination INCONCLUSIVE
Organic farming (6.4.1.1.4)	Land management (2.2.2) Pesticides (2.3.1)	Technical	Established	Supports more species of wild pollinators than non-organic WELL ESTABLISHED Enhances for pollination ESTABLISHED BUT INCOMPLETE
No-till farming (6.4.1.1.5)	Land management (2.2.2)	Technical Knowledge	Tested	Contrasting results for effects on ground-nesting bees and overall yields UNRESOLVED
Change irrigation frequency or type (6.4.1.1.6)	Land management (2.2.2)	Technical	Tested	Promotes wild insects abundance INCONCLUSIVE
Change management of productive grasslands (6.4.1.1.7)	Land management (2.2.2)	Technical	Tested	Reduced chemical inputs and delayed mowing usually increase pollinator numbers WELL ESTABLISHED Little evidence for pollination INCONCLUSIVE

Response/action (Relevant Chapter 6 section)	**Main driver(s) (Chapter 2)**	**Type**	**Status**	**Scientific evidence**
Diversify farming systems (mixed crop types, crop-livestock mixtures, intercropping, cover crops) (6.4.1.1.8)	Land management (2.2.2)	Technical	Established	Enhances pollinator abundance and species WELL ESTABLISHED Enhances for pollination ESTABLISHED BUT INCOMPLETE
Make crops more attractive to pollinators, to enhance pollination (additives or breeding strategies) (6.4.1.1.9)	Land management (2.2.2)	Technical	Tested	Increases pollinator visitation rate ESTABLISHED BUT INCOMPLETE Little evidence for pollination INCONCLUSIVE
Monitor and evaluate pollinators and pollination on farms (6.4.1.1.10)	Land management (2.2.2)	Technical Knowledge	Tested	Promotes pollinator and pollination conservation ESTABLISHED BUT INCOMPLETE
Reduce dependence on pollinators (mechanical replacement or breeding strategies) (6.4.1.1.11)	Land management (2.2.2)	Technical	Tested	Compensates pollination deficit INCONCLUSIVE
Adapt farming methods to climate change (6.4.1.1.12)	Climate changes (2.6)	Technical Knowledge	Proposed	Effectiveness at securing pollination under climate change is untested and likely to vary significantly between and within regions INCONCLUSIVE

Response/action (Relevant Chapter 6 section)	**Main driver(s) (Chapter 2)**	**Type**	**Status**	**Scientific evidence**
Establish regulatory norms and certification criteria for forest and agricultural products (6.4.1.2, 6.4.1.3)	Land management (2.2.2)	Legal Economic	Proposed	Enhances pollination and promotes pollinator conservation on farms ESTABLISHED BUT INCOMPLETE for pollinators INCONCLUSIVE for pollination
Pay financial incentives to farmers for practices that support pollinators (6.4.1.3)	Land management (2.2.2)	Economic	Established	Enhances pollinator abundance and species WELL ESTABLISHED
Engage and empower farming communities to work together to manage pollinators (6.4.1.4)	Land management (2.2.2)	Social/ Behavioural	Tested	Potential to enhance pollination and promote pollinator conservation, but no evidence of this yet INCONCLUSIVE
Translate existing research into agricultural practice through implementation, demonstration and extension (includes providing information to farmers about pollination requirements of crops) (6.4.1.5)	All	Knowledge	Tested	Enhances pollination and promotes pollinator conservation ESTABLISHED BUT INCOMPLETE

6.4.2 Pesticides, pollutants and genetically modified organisms

This section collates experience and scientific information about responses relating to pesticides, pollutants and genetically modified organisms. The impacts of these on pollinators and pollination are described in Chapter 2, sections 2.3.1 and 2.3.2. Responses are designed to reduce, eliminate or mitigate against known impacts.

Reducing the exposure of pollinators to pesticides and the toxicity of pesticides to pollinators will reduce direct risks to pollinators. Herbicides constitute the most used pesticides globally. They provide mainly an indirect risk by decreasing forb and flower availability to pollinators in the crop field, as well as in the landscape through drift and spraying of field and ditch edges, rights-of-way habitat etc. (Egan et al., 2014; see Chapter 2, section 2.2.2.1.4). The potential direct risk for pollinators from herbicides is poorly known.

6.4.2.1 Technical responses

6.4.2.1.1 Risk assessment techniques

Risk assessment of pesticides (compounds meant for controlling weeds, fungi, bacteria or animal pests) and other agrochemicals (e.g., blossom-thinners, or crop growth regulators), is an important tool to estimate the risk to insect pollinators. (Throughout this section "pollinators" refers to insect pollinators (mainly bees), as the link between pesticides and non-insect pollinators are comparatively little studied.) Risk depends on a combination of the hazard (toxicity) of a compound and the exposure of pollinators to this compound (e.g., Alister and Kogan, 2006). Risk assessment is performed at registration of a pesticide for use in a country. The honey bee was the first species in the focus of regulators, who started attending to the bee safety of pesticides a century ago. In Germany, for instance, the first ecotoxicological tests on bee safety of pesticides were conducted in the 1920s, and the first decrees to protect bees from insecticides came in the early 1930s (Brasse, 2007). Registration is since then based on ecotoxicological studies using a well-established set of methods that are being constantly developed and refined. The methods assess direct (but not indirect) lethal and sublethal threats to pollinators.

Two general techniques are used. The first basic approach (termed low tier) adopted by many countries is to test the hazard, i.e., the acute toxicity of the active compound, by estimating lethal doses in the laboratory. For pollinators, this straightforward technique is usually performed using the adult honey bee as the indicator species (also called surrogate species) for pollinators (Alix and Lewis, 2010; Anonymous, 2010). Risks to other pollinator taxa are routinely represented by, for example, rats and other mammals (for bats) and upland game birds, waterfowl or other bird species (for pollinating birds such as hummingbirds). However, because other bee species, and also the larval life stage of the honey bee, may differ substantially in their responses to a compound, guidelines have been developed to include toxicity assessments also for honey bee larvae (Oomen et al., 1992; OECD, 2013), and guidelines for toxicity tests on other bee species are under development (Fischer and Moriarty, 2014).

The second (higher tier) more resource-intensive approach is triggered by the outcome of the first tier, i.e., an intrinsic toxicity that is higher than a pre-defined threshold value that is empirically based on field incident data, and assesses the combination of toxicity and exposure under more realistic conditions in determining the likelihood on survival and sublethal effects in bees or their colonies. Techniques are becoming available for tests under semi-field or field conditions; some are standardized (e.g. EPPO 170 (http://pp1.eppo.int/getnorme.php?id=257) OECD, 2007) but the uncertainties linked to making assessments in the field are limiting their implementation in the regulatory process. These approaches are included in the regulatory registration process in some countries (see Legal responses below). For instance, guidelines for testing of pesticide impacts are internationally available for semi-field and field testing for pollinators (OECD, 2007; Anonymous, 2010; EPA, 2012; EPA et al., 2014).

There is on-going research to support the development of tools for assessing risks to pollinators, including studies for assessing sublethal effects on honey bees as well as other surrogate test species (Desneaux et al., 2007; EFSA, 2012; Hendriksma et al., 2011; EFSA, 2013b; Arena and Sgolastra, 2014; Fischer and Moriarty, 2014). Current method developments, especially in Europe and North America, focus on validating tests of chronic exposure in the laboratory, and on methods assessing impacts on bumble bees and wild bees. It has been suggested that tests need to be developed of exposure and hazards of combinations of pesticides, also combined with other stressors (Vanbergen et al., 2014). A novel approach is to consider potential impacts on ecosystem services, including pollination, in the risk assessment (Nienstedt et al., 2012).

It is not feasible to implement a full global quantitative risk assessment for all chemicals. It was estimated that there were more than 900 active substances intended for agriculture on the global market in 2009 (Tomlin, 2009). Comparative risk assessments are used with pesticide risk ranking tools as an initial screening to identify chemicals to take forward for further assessments, identify information gaps, or inform a risk management approach. Labite et al. (2011) reviewed the main 19 pesticide risk ranking tools in use in Europe and North America, categorising them according to their data needs and the specific environmental risks covered. Ten of the 19 used bee toxicity data to assess toxicity of specific chemicals as part of the risk assessment, but only one risk-ranking tool specifically evaluated the risk to pollinators (bees) – the Environmental Risk Index (ERI) developed in Chile (Alister and Kogan, 2006). This tool does not appear to have been used in practice to screen pesticides for risk assessment.

FAO and other partners have developed a risk profiling tool that assesses risk from pesticide exposures to pollinators in the field (van der Valk et al., 2013). The risk profiling is based on local information on which species provide pollination to the crop in question in the region, and a list of main factors influencing pesticide risk (e.g., pesticide type and use, phenology of crop flowering and pollinator activity). A risk profiling approach may be a cost efficient tool, particularly useful when a comprehensive risk assessment is not available. It provides a qualitative estimate of exposure, helps identify risks and knowledge gaps, and can provide a basis for education and to identify land management practices that may reduce pesticide exposure. The tool has been tested for three countries (Brazil, Kenya and the Netherlands) (van der Valk et al., 2013).

6.4.2.1.2 Risk mitigation technology

There are three general approaches to reduce exposure and thereby risk of pesticides for bees with technology: i) reduction of pesticide drift, ii) development of pollinator-friendly pesticides, and iii) application of cultivation practices that reduce exposure from or entirely avoid use of pesticides.

Reducing pesticide drift has been identified as an important action to reduce risks from pesticides use (FOCUS, 2007). Low-drift spraying equipment has been developed and tested (Felsot et al., 2010). Specific developments include sprayers with nozzles that generate larger droplet sizes, that apply the pesticide closer to the ground, or that have air wind shields mounted when spraying near the field borders. Also, changing formulation of the pesticide can reduce drift (Hilz and Vermeer, 2013). Planting buffer zones or wind breaks at field borders has been tested and recommended in several countries to reduce drift of pesticides into adjacent habitats (Ucar and Hall, 2001). However, because the buffer zone itself often contains flowers that attract pollinators, an additional in-field buffer zone can be used to protect pollinators from drifting pesticides.

Planting of pesticide-treated seeds can result in pesticide-contaminated dusts particularly in large pneumatic planters (Krupke et al., 2012; Taparro et al., 2012). Dust capture through filters and air recycling deflectors for seed-dressed neonicotinoid pesticides has been shown to reduce, but not eliminate, exposure and thereby risk from pesticides that have high acute toxicity to bees (APENET, 2011; EFSA, 2013; Girolami et al., 2013). Based on a monitoring programme of acute bee poisoning incidents in Austria 2009-2011, it was concluded that improved seed dressing quality and regulated seed-drilling equipment, reduced, but did not completely avoid incidents (Austria, 2012). Recommendations to reduce exposure during sowing of treated seed with pneumatic planters have been developed for some crops, e.g., avoid planting in windy conditions or modify the sowing equipment. However, there is a knowledge gap on dust exposure to pollinators at sowing of dressed seeds for many crops (EFSA, 2013).

These actions can substantially reduce drift and thereby exposure and risk to pollinators in the agricultural landscape. The efficiency of these techniques is normally estimated as percent reduction of drifting pesticide based on measurements and models (Felsot et al., 2010). The efficiency in terms of actual reduced impacts on pollinator individuals in the field remains scarce (e.g., Girolami et al., 2013) and even less is known for communities of pollinator (but see Brittain et al., 2010). There are no data on the extent to which drift reduction technologies have been implemented globally. A database has been set up for countries in Europe to list implemented pesticide drift reduction measures (http://sdrt.info).

Another technical response is to develop new pesticides with low toxicity to non-target organisms. These can potentially also be combined with biocontrol methods (Gentz et al., 2010). However, the number of new active ingredients being developed and introduced is limited, due to economic and environmental challenges.

6.4.2.1.3 Best management practices

Potential risks from exposure of pollinators to pesticides can be reduced by developing and encouraging use practices sometimes referred to as 'best management practices' (Hooven et al., 2013, Wojcik et al., 2014). Suggestions and training for best management and stewardship with specific reference to pollinators appear in advice to pesticide users and education material to pesticide applicators in several countries. This is mainly provided by governmental institutions and universities (e.g., http://insect.pnwhandbooks.org/bee-protection), but also by pesticide distributors and producers (https://croplife.org), universities and commodity groups. They also appear as recommendations for use on the pesticide labels.

There is no comprehensive summary of available advice internationally, but general recommendations include the following. First, to avoid applying the pesticide when the pollinators are actively foraging in the treatment area, e.g., not to apply insecticides when crops and weeds are in flower and in some cases several days before flowering, or at the time of the day when bees are foraging (Thompson, 2001). In public health efforts to reduce mosquito populations, impacts on pollinators have been minimized through timing and mode of application (Khallaayoune et al., 2013). Other recommendations include, whenever possible, to select pesticides with the lowest toxicity rating to pollinators, that rapidly detoxify via degradation and that have a as low as possible residual toxicity; to avoid tank mixing of pesticides as risks from most combined compounds are largely unknown (see Chapter 2); to remove weeds before flowering, e.g., by mowing before application; and to follow the label which may also include information on best management practices (see also Chapter 6.5). It can also be recommended not to apply pesticides when unusually low temperatures or dew are forecast as residues can remain toxic to bees much longer under these conditions. However, the toxicity can increase or decrease with temperature depending on the compound (Medrzycki et al., 2013). There are several techniques to minimize spray drift into adjacent pollinator habitats and non-target crops: spraying at calm wind conditions, adopting low-drift machinery (see above), and using in-crop buffer zones by turning off the sprayer near pollinator habitats at field margins. Other actions include to communicate to nearby beekeepers about when and which pesticide is being applied, such that honey bee hives can be removed or closed during application and a period after the pesticide treatment (Hooven et al., 2013). Obviously this measure will possibly protect honey bees but not other pollinators.

6.4.2.1.4 Reduce pesticide use (includes Integrated Pest Management)

Developing and implementing cropping systems that entail no or low use of pesticides, such as organic farming (see section 6.4.1.1.4) may reduce use and thereby exposure to pesticides. A major effort in conventional farming has been to decrease pesticide use through the adoption of integrated pest management (IPM). This entails a number of complementing pest control strategies with larger reliance on biological pest control and changed cultivation practices that decrease the need to use pesticides and to apply pesticides only when they are needed, i.e., when other measures are insufficient and pest abundances have reached the damage threshold (Desneux et al., 2007; Ekström and Ekbom, 2011; USDA, 2014; http://www.ipmcenters.org/). The cultivation practices involved include crop rotation or mixed cropping, and field margin management, with co-benefits for pollinators discussed in section 6.4.1.1. Measures have to be balanced against the risk of attracting pollinators to or near areas treated with pesticides.

6.4.2.2 Legal responses

6.4.2.2.1 Registration

The requirement to register a pesticide before use is a primary level and regulatory policy tool that in many countries has as one aim: to limit use of bee-toxic pesticides and implement pollinator-safe use of the pesticide. Pesticide products are normally registered one by one, separately for specific uses (e.g., seed dressing, by crop) and separately in each country; but national registration can also be based on internationally agreed procedures. A comprehensive global overview of registration procedures and requirements is not available. It is, however, safe to say that the principle and strictness in the rules and procedures for a pesticide registration vary enormously among countries. An indication of this variation is given by the Environmental Performance Index (EPI) that is updated annually since 2000 (http://epi.yale.edu). It gives a country-based overall assessment of environmental stress on human health and ecosystems based on agricultural land use and policies, and includes pesticide use and regulation.

Information about pesticide use is largely lacking and many countries even lack sales statistics. The EPI therefore instead scores the regulatory strength at the registration of pesticides, and tracks plans by national governments to phase out and ban a number of Persistent Organic Pollutants (POP), including nine pesticides now obsolete in agriculture. Ekström and Ekbom (2011) list the scored capacity to regulate pesticides of 11 coffee-producing countries in 2008. The scores range from 0 or 1 (e.g., Guatemala, Uganda, and Honduras) to around 20 (e.g., Brazil, Indonesia, Peru, and Vietnam), which is level with the scores of countries with internationally recognized strict registration rules (New Zealand 22, Sweden 22, US 19).

Other indications of the global variation in the regulation of pesticide use through registration is given by a regional risk assessment report for West Africa. It shows that pesticide regulation in West African countries is weak and that 50% of pesticide applications in Mali, and 8% of marketed pesticide products in Niger are reported as unregistered and therefore entirely lack risk assessments for pollinators (Jepson et al., 2014). Panuwet et al. (2012) report illegal use of pesticides, and weaknesses in the regulation and monitoring of pesticides use in Thailand. More strict registration rules not only include advanced risk assessments (with ecotoxicological studies) and rules of use (through labelling), but can also include responsibilities for the pesticide producer to mitigate risks and monitor use after registration, and allows for further restrictions of use should negative impacts on the environment and non-target organisms be observed (e.g., EC 2009, see especially Articles 6, 36 and 44). New, even more conservative, risk assessment systems are being developed for the EU and US that include measures of lethal and sub-lethal effects for several bee species in addition to the honey bee (EFSA, 2013; Fischer and Moriarty, 2014).

6.4.2.2.2 Labelling

The label provides instruction for use of the pesticide and is considered an important tool to limit risk to non-target organisms and humans. Labelling is a regulatory action that is generally part of the pesticide registration. No comprehensive summary of labelling internationally is available. A label may or may not include instructions directly related to protecting pollinators, but many pesticide labels include clear warnings about the potential risks to pollinators. In a survey on registration procedures including 20 OECD countries worldwide, all countries were found to use label mitigation to reduce risk to pollinators including approval restrictions (e.g., excluded crops, rate restrictions), use restrictions (e.g., not to be used during flowering), and advice for risk-reducing practices (e.g., avoid drift). Most countries (~80%) have a mechanism for enforcing mandatory label mitigation measures and restrictions, e.g., such that “do not” statements are legally binding. Few countries have a formal mechanism for determining the effectiveness of risk mitigation with labelling (Alix, 2013; http://www.oecd.org/chemicalsafety/risk-mitigation-pollinators/), which is typically based on incident monitoring systems.

6.4.2.2.3 Compulsory training and education

Many countries require a licence (certification) for a person to apply certain pesticides; this licence or certification is issued after a formal training course. From a survey of 20 OECD countries, training and education for pesticide applicators was mandatory in half of the countries (Alix, 2013). It is likely that such mandatory training is an efficient way to disseminate information on the responsible use of pesticides for humans and the environment, but no evaluation of the effectiveness or compliance with such measures was found. Although a country may have mandatory training for some pesticides (e.g. for 'Restricted-use' pesticides in the US http://www.epa.gov/pesticides/safety/applicators/restrict.htm), many pesticide appliers (including professionals) are not required to receive formal training for other pesticides (e.g., 'General-use' pesticides in the US).

6.4.2.2.4 Bans and moratoriums

On the global level, 72 countries have joined the Rotterdam Convention on Prior Informed Consent (http://www.pic.int), which controls trade restrictions and regulation of toxic chemicals, and many countries adhere to the Stockholm Convention of Persistent Organic Pollutants (http://chm.pops.int). The conventions aim to phase out the use of the use of chemicals meeting certain criteria in terms of persistence, bioaccumulation, and toxicity; this list currently includes 9 pesticides used in agriculture (the insecticides aldrin, dieldrin, chlordane, DDT, endrin, mirex, heptachlor, and toxaphene, and the fungicide hexachlorobenzene).

A moratorium is a regulatory action in which a temporary suspension of certain uses is imposed at a regional or national level. Such suspensions have been imposed when monitoring and/or research demonstrate negative impacts on pollinators after an accepted registration. A recent, much debated, example is the temporary moratorium in the EU of certain uses of neonicotinoids (Dicks, 2013; Gross, 2013; Godfray et al., 2014). The decision was based on identified effects and knowledge gaps in the estimated risks to wild pollinators and honey bee colonies in the field from neonicotinoid use (EFSA, 2013b; EFSA, 2013c; EFSA, 2013d; EFSA, 2013e; Godfray et al., 2014; EU Regulation 485/2013). The 2013 European regulation (No 485/2013) required manufacturers to submit information on risks to pollinators other than honey bees, and a number of other aspects of risk. The debate is ongoing whether the scientific evidence is sufficient to warrant a continuation of the moratorium. Use of four neonicotinoids has also been restricted on *Tilia* spp. trees in Oregon, US (http://www.oregon.gov/oda/programs/Pesticides/RegulatoryIssues/Pages/PollinatorIssues.asp), following a major kill of bumble bees foraging on those trees when they were sprayed. A restriction on use of neonicotinoid seed treatments for corn and soy in Ontario, Canada, is now in force and will require an 80% reduction in use by 2017.

6.4.2.2.5 Options to strengthen pesticide regulation globally

Risks of pesticides to pollinators are likely to decline if nations match risk assessment stringency and regulation of pesticides with those countries that have the most advanced registration procedures. This would raise registration standards globally. However, there are important limits to realise this policy as it will require resources that are not always available. Advanced risk assessments at registration are costly. The pesticide producers need to perform more tests, and may be reluctant to go through a costly registration for small markets. Such standards are expensive and require considerable data to support them. Also the governments setting the standards need to fund staff to handle registrations and assess risks. Sufficient experience, technical skills and specializations may be lacking within government agencies to assess studies properly.

There are several possible solutions. One option is to make registration studies more readily available worldwide such that they can be used by more than one country. A more active communication of knowledge worldwide would allow for improved risk assessments in countries with weak regulatory institutions (http://www.oecd.org/chemicalsafety/testing/oecdguidelinesforthetestingofchemicals.htm).

Several countries can also merge resources and skills for a harmonized or common registration process on a joint market. For example, in 1994, thirteen countries in West Africa developed a joint registration process for pesticides to support enhanced control of the pesticide trade (http://www.insah.org/). Seven of the countries have fully integrated this registration into their legislation. Similarly, the Southern and East African Regulatory Committee on Harmonization of Pesticide Registration (SEARCH), the East African Community (EAC), and the Economic and Monetary Community of Central African States (CEMAC) have started to harmonize their pesticide regulations, but do not yet have a common registration process. In other parts of the world, such discussions have been initiated focusing primarily on information exchange (e.g., CARICOM in the Caribbean, Comunidad Andina CAN in South America, and Secretariat of the Pacific Community SPC in the Pacific). The Organization for Economic Cooperation and Development (OECD) has compiled a guideline for joint reviews of pesticides among nations (http://www.oecd.org/chemicalsafety/pesticides-biocides/46754279.pdf).

6.4.2.2.6 Global Code of Conduct

An International Code of Conduct on Pesticide Management was adopted by member countries of the FAO in 1985, revised in 2002 and again in 2014 (http://www.fao.org/docrep/005/y4544e/y4544e00.htm; http://www.fao.org/agriculture/crops/thematic-sitemap/theme/pests/code/en/), primarily targeting voluntary actions by government and industry to reduce risks for human health and environment from pesticide use. However, only a few countries (61% of those surveyed, or 31 countries) appear to be using the code, based on a survey in 2004 and 2005 (Ekström and Ekbom, 2010), possibly because it had not been well promoted internationally. Ekström and Ekbom (2010) suggest that the Code could be used as a vehicle to promote non-chemical pest management options and the use of pesticides with low toxicity and exposure, and to phase out the use of highly hazardous pesticides as ranked by researchers, NGOs and governmental organisations (Kovach et al., 1992; WHO, 2009; PAN, 2013).

6.4.2.2.7 National risk reduction programmes

Several national pesticide risk-reduction programs have been implemented since the 1980s; examples include those in Brazil, Canada, Ethiopia, France, Sweden (e.g., Barzman and Dachbrodt-Saaydeh, 2011; Rusch et al., 2013). The efficiency of these programmes is generally evaluated based on risk indicators to health and environment, but not considering pollinators specifically (see section 6.4.2.4.1). Development of specific risk indicators from exposure of pesticides to pollinators would be useful for evaluating possible impacts of such programmes on pollinators.

6.4.2.2.8 Promoting pollinator-friendly farming and forestry practices

Promoting reduced pesticide or non-chemical pest management practices depends not only on a technical or knowledge response, but a willingness to provide resources that give continuous support to pollinator-friendly pest management research, extension and practices. It entails enacting agricultural policies that promote agricultural methods that reduce pesticide use, adopt IPM strategies, and low- or no-pesticide crop production systems (e.g., organic farming). As an example, the EU has decided that member states develop an Integrated Pest Management (IPM) action plan by 2014 (91/414 EEC).

6.4.2.3 Economic responses

There are many subsidy programs aimed to support biodiversity in agricultural landscapes that include the non-use of agrochemicals. Available evidence on the efficacy of these actions provides a mixed and complex picture of the effects of reducing agrichemical impacts on wildlife (Dicks et al., 2014b; http://www.conservationevidence.com/actions/139), but was unanimously characterised as beneficial in an expert assessment (Dicks et al., 2014c).

Another economic response is to introduce pesticide taxes and fees. These are market-based instruments that have been proposed to discourage pesticide use, and have been implemented in some European countries (Skevas et al., 2013). Important knowledge gaps remain with respect to introducing such policies broadly, e.g., related to actual efficiency in reducing risks depending on pesticide use, toxicity and productivity in a region (Skevas et al., 2013). Pedersen et al. (2012) further show that the uptake efficiency when implementing these instruments will vary depending on the farmers' motivation to maximise profits or increase the yield, implying that it is necessary to adopt an array of policy instruments to match the rationales of many farmers.

The cost and crop damage risk of an IPM approach can be minimized by a yield insurance scheme. A promising example of this is in Italy, where the program is managed as a mutual fund by participating farmer associations (Furlan and Kreuzweiser, 2015).

6.4.2.4 Knowledge responses

6.4.2.4.1 Monitoring and evaluations

Monitoring of environmental risks from pesticides is performed in many countries. It can be based on health and environmental risk indicators based on pesticide sales and use estimates, toxicity, and of measurements of residues in the environment (e.g., Labite et al., 2011, http://www.oecd.org/env/ehs/pesticides-biocides/pesticidesriskindicators.htm).

Little monitoring assesses risks on pollinators specifically. However, there is some evidence that restrictions have reduced the risk to pollinators in the UK. Based on risk indicators, Cross and colleagues found a decrease in the average environmental risk of pesticides per hectare for fruit and arable crops between the first introduction of risk-based regulations in 2002, and 2009 (Cross and Edwards-Jones, 2011; Cross, 2013). They combined pesticide usage data with a measure of hazard (toxicity) for each specific chemical, including simple scores for bee and beneficial insect toxicity. Reduced risks were largely due to removal of specific chemicals from the market, but were not consistent across crops as the risk score increased for, e.g., cider apples and pears (Cross, 2013).

There has been continuous, or time-limited, monitoring of poisoning incidents of mainly honey bees in some countries. In some EU countries and the US (http://www.npic.orst.edu/incidents.html) authorities maintain intoxication incident surveillance. No environmental monitoring of pesticide impacts on wild bees is documented except for bumble bees in the UK and in the US.

Evaluations of such monitoring programmes published in the scientific literature include incidents of honey bee and bumble bee poisoning in the UK 1994-2003. Bee death incidents attributed to pesticide poisoning declined from 23 to 5 per year in this period (Barnett et al., 2007). Similarly, the number of incidents had a decreasing tendency, but with some intermittent peaks, in the UK, the Netherlands and Germany 1981- 2006 (Thompson and Thorbahn, 2009). Very few incidents occurred in Canada 2007-2011, but with a sharp increase in 2012 in the Ontario province, where exposure to neonicotinoid dust during planting of corn was suspected to have caused the incident in up to 70% of cases (Cutler et al., 2014a). Monitoring of bee poisoning from use of neonicotinoid insecticides has taken place in Austria, Slovenia, Italy, and France. Several incidents were reported, but the direct causality between pesticide exposure and observed bee deaths is uncertain for several of these studies (EFSA, 2013).

6.4.2.4.2 Education

An important and efficient action is to educate pesticide applicators on the correct use of pesticides by following the label instructions and to adopt risk reduction practices. Many such programs exist around the world (see section 6.4.2.1). Farmer education has also been shown to result in effective implementation of IPM measures that reduce exposure and risks to beneficial organisms (van den Berg et al., 2007, Waddington et al. 2014). Studies of pesticide applicator attitudes suggest that there is potential for voluntary approaches to raise awareness among applicators of habitats sensitive to pesticide drift in rural landscapes (Reimer and Prokopy, 2012). Other important target groups are students in plant protection, agronomy and agriculture in general, and extension personnel who give pest management advice to farmers in particular. Education of extension personnel can serve as effective means of promoting pollinator-friendly practices and avoid unnecessary pollinator exposure to pesticides, as exemplified by a study from Ghana (Hordzi, 2010).

See section 6.5.12 for an example of a decision support tool designed to help farmers and advisers choose crop protection products with lower toxicity to pollinators.

6.4.2.4.3 Research

Ecotoxicology is an area of very active research (see section 2.2.1), which can have a substantial impact on policies and registration if it demonstrates unanticipated impacts of a particular pesticide on non-target species (see section 6.4.2.2.4, for example). In response to new research, regulatory authorities want to understand why non-target effects are happening and seek to impose mitigation measures.

Increased funding into research for the development of biological and agroecological methods of pest control would create opportunities for viable alternatives to pesticide uses. More information on the economic benefits (or lack thereof) of pesticide usage would improve the decision base for pesticide users.

6.4.2.5 Heavy metals and other pollutants

There is a lot of concern and monitoring of heavy metals and other pollutants in the environment. However, there are few studies addressing impacts specifically on pollinators and pollination (section 2.2.4). There are no policies to mitigate impacts of heavy metals and other pollutants specifically on pollinators. Actions employed to reduce risks for wider biodiversity (e.g., soil removal, or phytoremediation) might be useful to pollinators by removing hazards, or they might constitute risks, e.g., by providing contaminated pollen for pollinators, but this remains to be evaluated and tested.

6.4.2.6 Genetically modified organisms

6.4.2.6.1 Legal responses

In most countries, commercial release of genetically modified (GM) crops is subject to specific legislation and for those countries that are signatories to the Cartagena Protocol on Biosafety to the Convention on Biological Diversity (CBD), environmental risk assessment (ERA) is required for the regulatory approval of GM organisms (CBD 2000, Annex II; 6; 1, Annex III). The Cartagena Protocol states that ERA of GM plants should be conducted on a case by case basis, taking into account the environment where the plants will be released and the characteristics expressed by the transgene. Despite that, in general, the environmental risk assessments of GM plants have followed the toxicological model used for synthetic pesticides. Usually this model evaluates the direct toxic effects of a specific product (such as an insecticide) on surrogate species and extrapolates the results to all other species in the environment (Suter II, 2007). Therefore, the species *Apis mellifera* has been used in ERA as a representative organism of all pollinator species (Duan et al., 2008; Carstens et al., 2014). The toxicological model has been criticized when used for GM organisms for not considering the characteristics of the transformed plant for the selection of non-target species, the inserted transgene and the environment where the plant will be released (Andow and Hilbeck, 2004; Hilbeck et al., 2011; Andow et al., 2013). Furthermore, this toxicological model applied to pre-release evaluation of GM plants has focused almost exclusively on the isolated proteins produced by the GM plants (Duan et al., 2008; Wolfenbarger et al., 2008; Lövei et al., 2009) with little consideration of the whole plant. It does not adequately address the possible indirect effects of importance to pollination, such as possible changes in the bee foraging behaviour (Arpaia et al., 2011). Indirect effects through the food chain and those generated by loss of flowers in response to herbicide use, are not considered in the risk assessments for insect resistant or herbicide tolerant GM crops (see section 2.2.2.2.1 for assessment of these effects).

Possible changes in the toxicological model have been discussed and new approaches for ERA of GM plants have been proposed to match the Cartagena Protocol guidelines (Hilbeck et al., 2011; Sensi et al., 2011; Dana et al., 2012; Sanvido et al., 2012; Andow et al., 2013; Carstens et al., 2014), but there is no consensus about the exact scope of the assessment of GM plants on non-target species. Globally, there is a clear need for comprehensive, transparent, scientific guidelines for selecting the non-target species to be evaluated, and among those, different species of pollinators need to be considered, not only *Apis mellifera*. The lack of these guidelines has led to different interpretations of the risk assessment process of

GM plants among stakeholders (developer companies of GM crops, governmental regulators, and scientists) (Hilbeck et al., 2011; Andow et al., 2013; see Table 1 in Carstens et al., 2014).

In conclusion, there are no international specific policies for risk assessment of GM plants on pollinators and no specific mitigation action to deal with the possible risks. The Cartagena Protocol does not make a clear reference to pollinators, but they are in the legislation of many countries within the scope of non-target organisms, along with other beneficial species such as those used as biological control agents (Flint et al., 2012). Various species, among them *Apis mellifera*, quail and mouse have been used in the ERA of GM crops. Whether these are appropriate surrogate species for wild pollinators has been questioned for toxicological tests of synthetic pesticides (see section 6.4.2.1).

6.4.2.6.2 Knowledge responses

In Brazil, a monitoring program may be required by CTNBio (National Biosafety Technical Commission, http://www.ctnbio.gov.br), based on the results of risk analysis and it is designed on a case by case basis. Until now, this committee has not required monitoring specifically for pollinators. In Europe, post-market environmental monitoring is required for all GM crops released in the environment (EFSA, 2011), but there are few specific guidelines for pollinators (Shindler et al., 2013; Dolek and Theissen, 2013).

36. Table 6.4.2.1. Summary of evidence for responses relating to pesticides, pollutants and genetically modified organisms.

Response (relevant Chapter 6 section)	**Main drivers (Chapter 2)**	**Type of response**	**Status**	**Scientific evidence**
Globally raise standards of risk assessment and regulation of pesticide use (includes labelling) (6.4.2.1.1; 6.4.2.2.2; 6.4.2.2.5, 6.4.2.2.6)	Pesticides (2.2.1)	Technical Legal	Established	Reduces risks to pollinators WELL ESTABLISHED
Risk assessment using risk indicators based on pesticide use (6.4.2.1.1)	Pesticides (2.2.1)	Technical	Proposed	Few indicators specifically addressing pollinators available ESTABLISHED BUT INCOMPLETE
Risk profiling to assesses risk from pesticide exposures to pollinators for particular crops and regions (6.4.2.1.1)	Pesticides (2.2.1)	Technical Knowledge	Tested	Tested in three countries ESTABLISHED BUT INCOMPLETE

Risk reduction and mitigation through agricultural practices that reduce exposure to pesticides (6.4.2.1.2; 6.4.2.1.3; 6.4.2.4.3)	Pesticides (2.2.1)	Technical Legal Knowledge	Established	Reduces risks to pollinators inside and outside fields WELL ESTABLISHED
Risk reduction and mitigation through technology that reduces pesticide drift (6.4.2.1.2)	Pesticides (2.2.1)	Technical	Established	There is evidence of substantially lower drift and dust emissions with improved technology WELL ESTABLISHED
Risk reduction through the development of less pollinator-toxic pesticides (6.4.2.1.2)	Pesticides (2.2.1)	Technical	Proposed	Few new pesticides are being developed in general INCONCLUSIVE
Educate and train extension, farmers, land managers and the public on the risks and responsible use of pesticides and pollutants (6.4.2.4.2)	Pesticides (2.2.1)	Knowledge	Established in many countries	Reduces risks to pollinators WELL ESTABLISHED
Monitor and evaluate the risks and impacts of pesticides and pollutants (6.4.2.4.1)	Pesticides (2.2.1)	Knowledge	Poorly developed for pollinators	ESTABLISHED BUT INCOMPLETE
Retract registration if research shows negative impacts on pollinators from actual use (6.4.2.4.3)	Pesticides (2.2.1)	Legal Knowledge	Tested	Reduces risks to pollinators UNRESOLVED
Globally phase out obsolete chemistries that may be more persistent bioaccumulative and/or toxic (6.4.2.4.3)	Pesticides (2.2.1)	Legal	Established	WELL ESTABLISHED

Research, implement, and promote practices for pest management with non-pesticide options, or less toxic pesticides (e.g., Integrated Pest Management) 6.4.2.1.3; 6.4.2.1.4; 6.4.2.2.8; 6.4.2.4.2)	Pesticides (2.2.1) Changes in land management (2.1.2)	Technical Legal Knowledge Legal	Established	Reduces risks to pollinators WELL ESTABLISHED
Continually evaluate the efficiency of measures and programmes aimed at reducing risk from pesticide use and pollution (6.4.2.4.1)	Pesticides (2.2.1)	Technical Knowledge	Proposed	INCONCLUSIVE
Introduce national risk reduction programmes (6.4.2.2.7)	Pesticides (2.2.1)	Legal Policy	Established	Reduces risks to pollinators ESTABLISHED BUT INCOMPLETE
Subsidize non-use of pesticides (6.4.2.3)	Pesticides (2.2.1)	Economic	Tested	ESTABLISHED BUT INCOMPLETE
Market-based instruments to discourage pesticide use (taxes and fees) (6.4.2.3)	Pesticides (2.2.1)	Economic	Tested	Tested but not evaluated in some countries INCONCLUSIVE
Provide insurance against loss and damage risk linked to IPM (6.4.2.3)	Pesticides (2.2.1)	Economic	Tested	Tested in Italy INCONCLUSIVE
Consider wild bees in the risk assessment and monitoring of impacts of gene modified plants (6.4.2.6)	Genetically modified organisms (??)	Technical Legal	Proposed	Indirect and sublethal effects of GMO crops on wild pollinators are not adequately addressed in GMO risk assessments INCONCLUSIVE

6.4.3 Nature conservation

Many pollinator species are known to be vulnerable or in decline (Chapter 3). This section examines nature conservation responses that are intended to or likely to support pollinators and pollination. The nature conservation focus means that the targets are wild pollinators rather than domesticated pollinators (e.g., the European honey bee *Apis mellifera*) but may nevertheless be important to agricultural pollination. Nature conservation responses are commonly applied to mitigate negative impacts of land use change, such as those identified in Chapter 2.

6.4.3.1 Technical responses

6.4.3.1.1 Habitat management

This area has the strongest knowledge base because it has been a focus for land management practitioners and ecological scientists. The evidence that loss of habitat has been a driver of pollinator decline is very strong (see section 2.1.1). Many studies have examined the response of pollinators to on-ground actions, which inform possibilities for the future. Possible actions range from the protection or maintenance of existing natural habitat to the creation of new habitat patches by ecological restoration. At a larger spatial scale there are also actions that relate to the planning of natural habitat networks and how they spatially relate to one another to ensure that pollinators can disperse and adapt to global change, and that there is the best benefit flow into agricultural landscapes (crop pollination).

There is evidence that forage resources commonly limit wild bee populations (Roulston and Goodell, 2011), which suggests that provision of additional appropriate forage resources could have significant population effects, but most studies do not assess these, instead focusing only on activity and frequency of pollinators. Planted forage resources might be focused on native plant species, and therefore be considered part of a nature conservation strategy, but because these plantings are generally integrated into agricultural practice, we have reviewed them in section 6.4.1.1 as agricultural responses. Forest management practices also influence bee communities, and planted forests have been shown to host significant bee communities in the early stages, but declining as a more closed forest environment develops (Taki et al., 2013). In New Jersey, USA, bees were more diverse and abundant when there was less closed forest in the surrounding landscape (Winfree et al., 2007). In tropical forest successional communities in Kenya, pollinator abundance and diversity actually increased across a gradient from natural forests to cultivated areas (Gikungu, 2006). Greater generalization was found among the bee communities in more mature forests, and more specialized and rare bee species were found in the earlier successional and more open habitats. In general, bees benefit from native plants and non-farmed habitats, but increasing cover of forests with closed canopies is less likely to favour rich bee communities.

In addition to the potential to improve crop pollination (Garibaldi et al., 2014), restored patches might re-establish pollination networks of wild plant species and their pollinators (Menz et al., 2011). Some studies have shown that restored patches compare well with remnant patches in terms of diversity and identity of dominant pollinators (Forup et al., 2008; Williams, 2011; Hopwood, 2008) but the flower visitation rate for native plant species (Williams, 2011) and interactions with insect parasites (Henson et al., 2009) may take longer to recover.

Bees often require specific nesting resources that can be enriched in a nature conservation strategy. For *Osmia bicornis* (formerly *rufa*), a stem-nesting bee in Europe, the provision of nesting material (reeds) in habitat patches in an agricultural landscape led to a local population increase (Steffan-Dewenter and Schiele, 2008) and many other trials establish that appropriate artificial nesting materials are used by a range of solitary bee species (Dicks et al., 2010). In contrast, the provision of boxes intended to host bumble bees has had highly variable outcomes (Dicks et al., 2010; Williams and Osborne, 2009) with average occupation of boxes low (Lye et al., 2011). Honey bees and stingless bees prefer to nest in large old trees, so protection of such trees is important. For example, the stingless bee species *Melipona quadrifasciata* was shown to nest selectively in the legally protected cerrado tree *Caryocar brasilense* (Atonini and Martins, 2003) (further discussion of nest sites for social bees is in 6.4.4.1.9 and 6.4.4.4.).

6.4.3.1.2 Landscape planning and connectivity

Landscape planning for better pollinator outcomes has been the subject of theory and discussion (e.g., Menz et al., 2011; Viana et al., 2012) and a component of large-scale research projects, such as LEGATO (http://www.legato-project.net/). Although landscape planning has aided conservation of some species, little information is available to demonstrate the effectiveness of landscape planning strategies for pollinators and pollination specifically. Studies of existing fragmented landscapes have shown that in some biomes, the edge environments that predominate in small or linear patches tend to favour only certain pollinators (Girão et al., 2007; Lopes et al., 2009). An important theme in landscape planning is the maintenance of landscape connectivity for animal movement and gene flow. Several recent studies imply that the configuration of landscape features (the way they are arranged in the landscape) have only weak effects on bee populations or population persistence (Franzen and Nilsson, 2010; Kennedy et al., 2013, for example). However, in a review of studies examining landscape effects on the pollination, Hadley and Betts (2012) indicated that it had been very difficult to distinguish effects of landscape configuration (i.e., the shapes and position of habitat fragments) from the more general impact of habitat loss (i.e., direct effects of land clearing).

Strategically-placed replanted vegetation might increase connectivity for ecological processes, which could benefit species in fragmented landscapes and support the ability for species to move in response to climate change. There is experimental and modelling evidence that pollen flow occurs between remnant and replanted vegetation (Cruz Neto et al., 2014) and that linear features linking patches of floral resource promote movement of bees and other pollinators through landscapes (Cranmer et al., 2012; Hodgson et al. 2012), thereby enhancing pollen transfer between plants in those patches (Townsend and Levey, 2005; Van Geert et al., 2010). These patterns provide some documentation of the benefits that habitat connectivity can provide. The role habitat connectivity has in maintaining pollinator populations remains unclear, but theory and observations for other taxa suggest that when the amount of natural habitat in the landscape declines below approximately 20% populations risk becoming isolated and connectivity may play an important role in their conservation (Hanski, 2015). Increased connectivity can be achieved by making the matrix (i.e., land between the habitat patches) more hospitable to dispersing organisms (Mendenhall et al. 2014), as well as by preserving or creating “stepping stones” and corridors of habitat connection.

Climate change can impact populations in many ways, and in some cases species are expected to shift in distribution (i.e., populations move) generally poleward or to higher elevations, so that they remain within a climatically suitable environment (Chen et al., 2011). This kind of movement is only possible if suitable habitat for the species occurs at the new locations. Further, for migration to occur naturally, connectivity of habitat for the species in question may be important, keeping in mind that species vary greatly in their capacity to move long distance or cross inhospitable environments. With this in mind, adaption to climate change could include habitat improvements and increasing connectivity across landscapes, but currently there is limited evidence regarding effectiveness of this strategy.

6.4.3.1.3 Non-timber forest products

Pollinators might also be important to the productivity and maintenance of non-timber forest products (NTFPs) (Rehel et al., 2009). For example, Brazil nut is primarily harvested from wild sources (Clay, 1997) and the production of nuts depends on pollination by large-bodied wild bees (Motta Maués 2002). Another interesting example showed that Yucatec Mayan people in Central America relocate honey bees into maturing stands of secondary forest, aged 10–25 years, to aid pollination and take advantage of the many flowering plant species for honey production (Diemont et al., 2011). While there are, no doubt, many other examples of NTFP’s that are animal pollinated (e.g. guarana, Krug et al., 2014; *Euterpe* palm, Venturieri, 2006), little is known of the extent to which sustainable yield depends on pollination rates or pollinator conservation and there is little scientific knowledge available regarding the effectiveness of nature conservation strategies in protecting the pollinators of NTFPs.

6.4.3.1.4 Invasive species

Where non-native insect pollinators pose a threat to the native fauna (see Chapter 2, section 2.5), management of invasive species is likely to be an important component of a pollinator conservation strategy. However, eradication of invasive species has proven difficult in most circumstances, with successful eradication most often occurring on islands where the area to manage is limited, and re-invasion is less likely. Because of this challenge, studies of the effectiveness of invader management in terms of pollinator response are rare. Nagamitsu et al. (2010) showed that active removal of *Bombus terrestris* from sites in Japan allowed an increase in abundance of queens for two native *Bombus* species, but attempts to reduce *Bombus terrestris* numbers in the next year failed. Hanna et al. (2013) show that a reduction in invasive wasps (using poison baits) led to an increase in pollination and subsequent fruit set of a native plant in Hawaii, although interestingly in this case the primary pollinator was also an invasive species (*Apis mellifera*).

Because it is so difficult to eradicate invasive species, a focus on mitigating their impact can be the necessary alternative. There have been many examples where management has successfully contained or reduced populations of invasive species, reducing their impact (Mack et al., 2000).

6.4.3.1.5 Species-focused conservation actions

Butterflies have often been a target group for species-focused conservation actions (New et al., 1995) with a number of successful projects (e.g., Thomas et al., 2009) including *ex situ* conservation (Schultz et al., 2008). Although they have had a high profile in species conservation, relative to other insects, butterflies are considered minor pollinators relative to other insect groups, especially bees (Chapter 1). One group of wild bees has been a focus for nature conservation: the bumble bees (*Bombus* spp.). This reflects that bumble bees are large and distinctive, and some species have experienced significant declines in parts of Europe, Japan, and the Americas (Williams and Osborne, 2009). Generalising from *Bombus* to other species should be done with caution, but these studies provide a starting point for understanding the potential for species-focused conservation actions.

Most on-ground strategies for species conservation are essentially forms of habitat management (and are therefore discussed above), albeit that some habitat interventions can be more precisely targeted if single species are the focus. For example, nest preferences are quite specific, and so provision of nest resources should match the preferences of the species of concern. Beyond habitat management, conservation strategies for single species might also include *ex situ* conservation and species re-locations. For example, *Bombus subterraneus* has been extirpated from its original range in the UK, but still occurs on the European mainland and in its introduced range in New Zealand. A project has been established to restore the required habitat and then reintroduce bees (http://hymettus.org.uk/downloads/B_subterraneus_Project_report_2011.pdf accessed September 5 2014). Bees were released in 2012 and are still being sighted in 2014 (http://www.bumblebeereintroduction.org/news/news/ accessed September 5 2014).

Wild *Apis* species in Asia, such as *Apis dorsata*, have also been subject of special attention. There is a long history of traditional exploitation of these species for their honey, and as a consequence they have particular cultural significance and are the subject of traditional knowledge. Use of traditional techniques to create good nesting locations might help support their populations (Hadisoesilo, 2001).

We could find no reports of other active *ex situ* conservation actions that were specifically pollinator targeted, although some vertebrate pollinator species (especially birds and bats) that are endangered in their native range are held in captive populations in zoos and other institutions (e.g., the Rodrigues Fruit Bat, *Pteropus rodricensis*, O'Brien et al., 2007). Fruit bats are the primary pollinators of some plants on Pacific Islands but are hunted for meat and threatened by hunting and invasive species (Cox and Elmqvist, 2000). Captive populations may contribute to species re-introductions if the drivers of threat can be managed in the natural range.

Translocation of species into new locations, where they may have a better chance of survival, has been suggested as a strategy that might be increasingly called for under climate change (Seddon et al., 2014) and has recently been suggested for bumble bees in particular (Kerr et al., 2015). This strategy might also have the effect of restoring ecological function to locations that have lost species. The number of case studies for the practice of translocation is a rapidly increasing and therefore helping to reveal the logistic challenges of the strategy (Seddon et al., 2014). The knowledge base for translocation of pollinators in particular is poor because insects, the most important group of pollinators, are rarely the subject of translocations (most cases focus on birds and mammals: Seddon et al., 2014). Nevertheless there have been successful translocations of some butterfly species (Kuussaari et al., 2015) and among the important lessons is that there must be high-quality suitable habitat available in the new location. Translocation comes with considerable risk of failure to establish and could also lead to unintended harm if translocated species become invasive pests or vectors for disease in the new range (Seddon et al., 2014 and see section 6.4.3.1.4. Invasive species). Given the complexity of the task, the shortage of practical experience, and the known risks associated with translocations, evidence that translocation could play an important role in pollinator conservation remains very limited.

For plants that rely on specialised pollinators for seed production, loss of pollinators might threaten their population viability even if conditions for vegetative growth are suitable (Pauw, 2007; Vovides et al., 1997; Machado and Lopes, 2000). For these plants recovery plans may require direct action to save their pollinators also. We are not aware of any studies that have assessed the effectiveness of this strategy. One European project is testing integrated plant and pollinator conservation for the dittany (*Dictamnus albus*). This plant species is rare and protected in several European countries, pollinated by generalist medium to large bees and threatened in some populations by pollination deficit (http://www.pp-icon.eu/). Management techniques being tested include flower planting and adding artificial solitary bee nest sites.

6.4.3.2 Legal responses

Legal responses can drive on-ground change, but are not in themselves a change to the natural environment in which pollination occurs. Literature on the effectiveness of legal responses in terms of pollination outcomes is lacking. Here we review some of the policy responses that are relevant to nature conservation for pollinators and pollination, but can provide only limited insight to their effectiveness.

6.4.3.2.1 Species listing and trade regulation

A traditional mechanism for managing species facing high extinction risk is to assess them as critically endangered, endangered or vulnerable (e.g., the IUCN Red Lists, national red lists, the Convention on Biological Diversity, European Community Birds directive), which might then invoke a protected status in national or international law, or heightened community awareness. The Endangered Species Act has been credited with improving the prospect of survival of listed butterflies in in the USA (Black, 2012), where the legislation has led to specific actions and investments by the federal government that might not have happened without the Act.

The formal listing of species has traditionally been biased towards certain taxonomic groups (e.g., plants, vertebrates) whereas insects (which are overwhelmingly the most important pollinators) are grossly under-represented (Stuart et al., 2010; Winfree, 2010; Byrne and Fitzpatrick, 2009). However, the first continent-wide list Red List for bees was recently published for Europe (Nieto et al., 2014); it reports that an estimated 9% of all bees (but 26% of bumble bees) are threatened. Importantly, for 56% of species there were not enough data to assign a status, underlining the size of the knowledge gap.

Another form of species-specific protection is to limit the permitted trade in species that have commercial value, and in some cases this could influence outcomes for pollinators. Lee et al. (2005) record that the establishment of a wildlife crimes unit in Sulawesi, Indonesia reduced the trade in some protected species, but in this case fruit bats, which are threatened by exploitation and known to be significant pollinators, were not on the list for protection.

Regulations restrict the import and/or release alien pollinator species in some countries. For example, there are regulations in a number of countries to restrict the import and use of non-native bumble bees as greenhouse pollinators (see Velthuis, 2002; Velthuis and van Doorn, 2006). The Invasive Alien Species Act in Japan restricts the transport of *Bombus terrestris* (https://www.env.go.jp/en/nature/as.html). In the UK, it is illegal to release non-native species according to the Wildlife and Countryside Act 1981 (http://www.legislation.gov.uk/ukpga/1981/69). Guidance on the regulations related to importing non-native bumble bees in this context can be found in the Guidance on Importing Bees into England (Animal and Plant Health Agency's (APHA) http://www.nationalbeeunit.com/index.cfm?pageId=126). The regulation has been recently amended to take account of non-native subspecies (Natural England, 2014), such as non-native subspecies of *Bombus terrestris*. In the USA, import of certain bee species (*Bombus impatiens, B. occidentails, Megachile rotundata*, *Osmia lignaria*, and *O. cornifrons*) from Canada is possible, while import of other species is restricted (USDA APHIS, 2013). Australia has rejected the import of *Bombus terrestris* as greenhouse pollinator and one state has classified the import of this species as key threatening process for native fauna (Australian Government Media release, 2008; NSW Scientific Committee, 2004). For North American countries, there are guidelines for the petition for import and release of non-*Apis* pollinators (NAPPO, 2008). National-level regulations are not effective if neighbouring countries on a land mass do not have similar regulations. An example of this is the invasion of *B. terrestris* in Argentina after its introduction into Chile (see Chapter 3).

The European honey bee is considered an introduced species in the Americas, most parts of Asia, Australia, and Oceania. Though there are concerns that managed honey bees may be a competitor of native bees (see Chapter 2), there are relatively few regulations in place that restrict the spread of honey bees as an alien species. Regulations in most Australian states prohibit the placement of apiaries in certain natural areas (Salvin, 2015). The Africanized honey bee is considered undesirable in many countries and there are regulations in some countries to restrict its potential spread. In Mexico, for example, there are measures to control the Africanized bee (Modificación a la Norma Oficial Mexicana NOM-002-ZOO-1994, Actividades técnicas y operativas aplicables al Programa Nacional para el Control de la Abeja

Africana). In some Argentinian provinces Africanized honey bee colonies are prohibited or have to be destroyed (e.g., Neuquén: La Legislatura de la Provincia del Neuquén Sanciona con Fuerza de Ley 1796; San Luis: Legislación Apícola de la provincia de San Luis Ley Nº 4.899 / 90). In the Australian State of Victoria, and the neighbouring country of New Zealand, the Africanized honey bee is classified as an exotic notifiable disease (New Zealand Ministry for Primary Industries – Bees and Honey, 2014) (Victoria Department of Environment and Primary Industries – Notifiable Diseases in Victoria).

6.4.3.2.2 Protected areas and other area-based conservation measures

Another widely-applied policy mechanism for nature conservation is the use of protected area status to conserve habitat. This approach has been applied in many counties around the world, leading to protected status, at least in name, for significant areas of land (Gaston et al., 2008). Of course, protected area status is not usually used solely to achieve a goal as specific as pollinator conservation, but higher-level goals such as biodiversity conservation usually apply. In Indonesia, decrees to conserve Karst landscapes, their natural caves and the bats living in them (acknowledging their importance as pollinators) is contained with the Guidelines for Management of Karst areas (2000) and Regulation on the Delineation of Karst areas (2012).

In some countries protected status is conferred on certain locations on the basis of religious or spiritual belief. There is increasing recognition of the importance of protected areas of this kind, sometimes recognised as "Indigenous and Community Conserved Areas" (https://iucn.org/about/union/commissions/ceesp/topics/governance/icca/). This form of protected status might support conservation of pollinators, even if this outcome is not an explicit part of the rationale. In parts of Madagascar local people protect small forest patches and modelling suggests these patches might support a significant level of pollination for surrounding agriculture (Bodin et al., 2006).

There is some evidence that protected area status has reduced the rate of habitat loss in many locations (Joppa and Pfaff, 2011), although there are also examples where this has failed (Gaston et al., 2008). It is fair to assume that protection of habitat has benefitted pollinators or pollination interactions, but we are not aware of any studies that have specifically addressed this question. In addition to supporting populations of wild pollinators, protected areas can, in some circumstances, provide floral resources that support beekeeping (Hausser et al., 2009).

Although the value of small habitat fragments has been recognised (Tscharntke et al., 2002, Turner and Corlett, 1996), reserve design for nature conservation has typically emphasised the benefits of protecting large parcels of land where possible. Large areas of habitat (tens of hectares or more) can be effective for preserving large populations of species, but because many pollinators move over relatively short distances (Greenleaf et al., 2007) such large reserves will not generally support crop pollination on agricultural land that is more than approximately 1km from reserved land. The benefits of non-agricultural habitats in supporting pollination generally extend a few hundred meters into fields (Ricketts et al., 2008). What remnant patches exist in farmed landscapes will often be too small to support populations of the larger species of conservation concern, such as vertebrates, but can play a very important role in keeping a diversity of insect pollinators (invertebrates) to support food production (Marlin and LaBerge, 2001). In this context it is important to think of small patches (meters across) of natural and semi-natural habitat (including field margins, pasture trees, etc.) as a target for “protected status”. Even individual trees in an agricultural landscape help support farmland pollinator diversity (Lentini et al., 2012). The emerging paradigm of “countryside biogeography” seeks to address the special challenges of achieving conservation outcomes in these kinds of landscapes (Mendenhall et al., 2014).

6.4.3.3 Economic responses

Payment for ecosystem services is a market-based instrument (e.g., Daily et al., 2009; Engel et al., 2008) that could promote practices that conserve pollinators. Crop pollination is well understood to be an ecosystem service that can flow across property boundaries, creating the possibility for a payment incentive for neighbours to conserve or create pollinator habitat (Dunn, 2011; Satake et al., 2008). Some governments reward land holders for carbon sequestration benefits of certain land uses (e.g., planting woody vegetation), and there is the possibility that co-benefits could also be rewarded (e.g., crop pollination that is promoted by the new habitat; Lin et al., 2013), but the effectiveness of these incentives in terms of pollinator conservation has not been assessed.

Turning the science-based concept into market mechanism is challenging (Madoff, 2011). There can be complex economic and social tradeoffs around the values of pollinators, such as seen in conflicts among the interests of almond growers, citrus farmers, and apiarists in the San Joaquin Valley (Madoff, 2011). Small payments may not be sufficient to motivate producers, but large payments risk distorting trade in a way that affects trade agreements. Because pollinators are mobile and there is a shortage of knowledge regarding key pollinators for many crops, it can be it difficult to identify which land owners could receive a payment for supporting them.

In France, an agri-environment scheme under the European Common Agricultural Policy (i.e., dispositif apiculture API: http://www.eure.gouv.fr/layout/set/print/Politiques-publiques/Agriculture/Mesures-Agro-

Environnementales) pays beekeepers to place hives in areas of high biodiversity. Its stated aim is to enhance the pollination provided by honey bees, although the effect of this on pollination has not been measured.

6.4.3.4 Social and behavioural responses

Responses based on influencing social attitudes have occurred in many places around the world. A number of initiatives related to pollinator conservation have garnered significant public support, including citizen science data collection and on-ground actions (see section 6.4.6.3.4). However, there have not been systematic studies of their effectiveness, so that while we have identified some of the strategies for how nature conservation strategies for pollination could benefit from social and behavioural responses (Table 6.4.3), there is little to report regarding assessment of the effectiveness of these strategies.

Social action also requires an appreciation of the threats to pollinators, which might be lacking in many communities. For example, people in the Cook Islands proved to be open to the idea that hunting restrictions might be necessary to protect fruit bats, but only after they were made aware that hunting was a significant threat to these pollinators (Cousins and Compton, 2005). In Europe surveys revealed a positive attitude towards the planting of wildflower strips for pollinator conservation among both farmers and the general public (Jacot et al., 2007), indicating that some communities are inclined to support active ecological restoration options. In a similar vein, other studies have shown that people’s aesthetic preferences lean toward floral diverse areas (e.g., Junge et al., 2011).

6.4.3.5 Knowledge responses

Reviews of regional conservation needs for pollinators have identified that a shortage of taxonomic expertise is a constraint, with many regions likely to have many species not yet described and a shortage of experts to identify species even when descriptions exist (Batley and Hogendoorn, 2009; Eardley et al., 2009; Freitas et al., 2009; FAO, 2008). To address the shortage of taxonomic expertise some institutions have developed training courses. The American Museum of Natural History has conducted a training course annually since 1999, training >250 people, and while many participants are researchers some come from non-research backgrounds (http://www.amnh.org/our-research/invertebrate-zoology/bee-course-2014). Similarly, the Kenyan “Centre for Bee Biology and Pollination Ecology” parataxonomy course (http://www.museums.or.ke/content/view/153/116/), was designed to give people without formal taxonomic training some of the skills required to identify specimens. These programs have effectively delivered training, but the impact on pollinator conservation of this increased capability is, of course, difficult to assess. Provision of these courses in developing countries especially is limited by availability of funding.

Use of new DNA sequencing methods provides tools that complement and extend traditional methods of species identification (Puillandre et al., 2012). These approaches are rapidly becoming cheaper and are expected to become applied much more widely in support of monitoring and understanding pollinators.

There is an immense reserve of knowledge regarding management for nature conservation outcomes from indigenous and local knowledge. Many indigenous peoples are known to value diversity for its own sake (see Chapter 5, sections 5.3.2 and 5.3.3).

37**. Table 6.4.3. Summary of evidence for responses relating to nature conservation.**

Response	**Main Drivers**	**Type of response**	**Status**	**Scientific Evidence**
Manage or restore native habitat patches to support pollinators	Land use and its changes (2.2)	Technical 6.4.3.1.1.	Established	Increases diversity and abundance of pollinating insects WELL ESTABLISHED
Increase connectivity of habitat patches	Changes in land cover and spatial configuration (2.1.2)	Technical 6.4.3.1.2.	Tested	Some evidence that habitat connections help pollinator movement and gene flow ESTABLISHED BUT INCOMPLETE
Manage invasive species (plants, pests, predators or pollinators) that diminish pollinators or pollinator habitat	Invasive species	Technical 6.4.3.1.4.	Tested	Case study evidence of some benefits to pollinator species, but eradication is difficult to achieve ESTABLISHED BUT INCOMPLETE
Targeted conservation of specific pollinator species or groups of species (includes *ex situ* conservation of threatened species, includes species of special cultural value)	Multiple, interacting threats	Technical 6.4.3.1.5.	Tested	Examples exist for a limited range of taxa ESTABLISHED BUT INCOMPLETE

Response	**Main Drivers**	**Type of response**	**Status**	**Scientific Evidence**
Targeted conservation of pollinators associated with specific plant species threatened by pollination deficit	Multiple, interacting threats	Technical 6.4.3.1.5.	Tested	One European example known, for dittany (*Dictamnus albus*) INCONCLUSIVE
Establish protected areas or improve the quality of existing ones (including protected areas of cultural value)	Land use and its changes (2.2)	Legal 6.4.3.2.2	Established	Protected areas host species diversity, but it is difficult to determine the impact of legislation in achieving protection WELL ESTABLISHED
Payment for ecosystem services	Land use and its changes (2.2)	Economic 6.4.3.3.	Tested	Ecosystems services payments have been established for other services (watershed protection, carbon sequestration) but no examples for pollination ESTABLISHED BUT INCOMPLETE
Maintain sacred and other culturally protected areas that support pollinators	Land use and its changes (2.2)	Social/ Behavioural 6.4.3.2.2	Established	Protected areas host species diversity, but few case studies ESTABLISHED BUT INCOMPLETE (see also 5.4.2.4)
Increase taxonomic expertise on pollinator groups (formal education/training) and technology to support discovery and identification	All	Knowledge 6.4.3.5.	Tested	Significant training has been achieved in a number of countries WELL ESTABLISHED

6.4.4 Pollinator management and beekeeping

This section focus on responses associated with managed pollinators, including beekeeping for the European honey bee *Apis mellifera* as well as any other managed pollinator species, including but not limited to other honey bees (such as *Apis cerana*), social stingless bees (Apidae: tribe Meliponini), bumble bees (primarily *Bombus impatiens* and *B. terrestris*), *Osmia* species (including *lignaria*, *cornifrons, cornuta,* and *bicornis*), the alfalfa leafcutter bee *Megachile rotundata,* and the alkali bee *Nomia melanderi*. An exhaustive list of managed pollinators is given in Chapter 2.5.

Māori and the management of introduced honey bees in New Zealand

Following the introduction of the honey bee (*Apis mellifera mellifera*) into New Zealand in 1839 (Barrett, 1996), feral honey bees rapidly established and spread throughout the country (Donovan, 2007). Māori quickly recognized the value of bees and honey in the mid-19th Century and became New Zealand's first commercial honey beekeepers (Barrett, 1996; Donovan, 2007; Gillingham 2012). The first New Zealand book on beekeeping '*Ko Ngā Pi*' (Treatise on bees) was published in Māori in 1849 (Cotton, 1849). Māori also adopted the practice of harvesting honey from feral honey bee nests (Lyver et al., 2015). Honey harvest would often occur twice a year (Tahi and Morunga, 2012) and feral hives were never depleted of honey to ensure the survival of the bees and the future potential to take honey. The relocation of swarms of feral honeybees during the *heke* or 'migration' period was also a common practice used to maintain access to honey (Doherty and Tumarae-Teka, 2015). Swarms were collected in a flax woven bag at night and moved to another site in an accessible tree cavity where the hive could develop.

Since the mid-1950s however the practice of harvesting honey from feral honey bee nests in the Te Urewera region by the Tuawhenua people has been in decline and today is no longer practiced (Doherty and Tumarae-Teka, 2015). Prior to 1950, honey would be collected from 20 to 25 feral hives in an area within 1 to 5 kilometre radius around homes. By the mid-1980s the gatherers were collecting honey from 1 to 5 nests in that same 1 to 5 km radius area, and by the late 1990s the feral honeybee nests had largely disappeared from the areas searched by Tuawhenua. The reason for the decline of feral honey bees is not well understood but the simultaneous rapid expansion of the European wasp (*Vespula germanica* Fabricius) (Fordham, 1961) is thought to be a factor; these wasps were known to consume honey bee brood and rob nests of honey (Thomas, 1960; Mayer et al., 1987).

In recent years, Māori have returned to management practices which facilitated within-forest pollination and production of apicultural products from indigenous flora such as rewarewa (*Knightia excelsa*)

(Indigenous New Zealand, 2012), tawari (*Ixerba brexioides*) and mānuka (*Leptospermum scoparium*). Today beekeeping is widespread and Māori have once again developed strong commercial links to the apiculture industry, especially bee products which are derived from mānuka which are recognised for its pharmaceutical purposes. Mānuka provides a highly valued source of honey and essential oil production (Stephens et al., 2005). The highest quality mānuka honey can provide returns of up to NZD$80/kilogram (Lyver et al., 2015).

6.4.4.1 Technical responses

6.4.4.1.1 Improve husbandry of managed pollinators

The focus of this section is on the development and testing of new technologies and management techniques, and scientific evaluation / testing of existing technologies and management techniques. This section is also focused only on currently managed pollinator species, as there is a separate section (6.4.4.1.3) on development of newly managed species.

The technical responses in this section are written to be taxonomically general wherever possible, i.e., aimed at any managed insect pollinator species, though there are clearly some responses that are taxonomically specific. Generally, there is a very long and well-documented history of beekeeping with honey bees (in particular *Apis mellifera*, and to a lesser extent *A. cerana*) and thus most of the evidence in terms of improving husbandry comes from *A. mellifera*. An exhaustive review of all *A. mellifera* beekeeping management practices is beyond the scope of this section, and many management practices are relevant only to particular geographic areas. Instead, we highlight general categories of management practices that offer the possibility of addressing threats to managed pollinators, with many of them focused on *A. mellifera*.

There is a growing literature on managed bumble bees (both *Bombus terrestris* in Europe and *B. impatiens* in the USA), and on pollinators such as *Osmia*, which are increasingly being used in orchard crops in the USA (*O. lignaria*), Europe (*O. bicornis* and *O. cornuta*), and Japan (*O. cornifrons*). While there is a long history of management of social stingless bees or meliponines (Apidae: Meliponini), particularly in Mexico and Central America (see Chapter 2.5), there has been less documentation and scientific study of this group relative to other groups. Recent advances have been made in several areas including stingless bee queen rearing (Menezes et al., 2013), non-destructive honey collection and nest box construction (Cortopassi-Laurino et al., 2006).

Indigenous and local knowledge adds new information and innovation on husbandry techniques for a range of managed bee species (see Chapter 5, section 5.4.10). There is a robust body of indigenous and local knowledge on stingless bee management (see Chapter 5, Case Examples 3, 9 and 13). For example, Quilombola communities in northern Brazil have a long tradition of stingless beekeeping. They have elaborate ecological knowledge of the 12 native stingless bee species, the melliferous flora and the management techniques (de Carvalho et al., 2014). Local people recognize that patches of habitat with trees, dense vegetation and an abundance of water, are preferred. In Indonesia and southern Vietnam, people have developed a method of 'rafter' beekeeping for the giant honey bee *Apis dorsata*. Wild, migratory bee colonies nest on the artificial rafters cut from young trees, allowing people to collect up to 80% of the honey without destroying the colony. There has been some research on how to improve this practice in Vietnam by placing rafters with open space in front (Dicks et al., 2010; Tan et al., 1997).

We address improvements in bee husbandry in six broad categories: i) general management, ii) management of disease threats, iii) genetic management, iv) management of pesticide threats (at the level of the beekeeper or pollinator manager, distinct from general management of pesticide threats), v) management of pollinator symbionts and vi) combinations of different management strategies.

6.4.4.1.1.1 General management

General management is focused on multiple goals, including reducing losses of bees; maintaining bee health generally; increasing honey production; and improving beekeeper livelihoods among others. This category includes a very wide range of different actions, and it is beyond the scope of this section to review these exhaustively, especially in terms of management of *A. mellifera*. Still, management innovation in these actions can lead to significant improvements in the survival and productivity of managed bees. It is worth noting that many of these management interventions likely have trade-offs, such that increases in some desired outcomes might, in some cases, lead to reductions in other desired outcomes.

General Management techniques include:

- hive / nest design and management (especially for bees other than honey bees; but for honey bees this could include reduction of costs of nest boxes, e.g. top-bar hives)
- diet / feeding (including management of forage *in situ*, management of moving bees to specific forage, and supplemental feeding)
- management of swarming / splitting colonies / requeening / queen rearing in eusocial managed bees (honey bees, bumble bees, and social stingless bees)
- reducing robbing and absconding in honey bees and social stingless bees (e.g., through use of unique colony markings, entry orientation, height above ground, etc.)

- migration / movement: at least one managed species (*Apis cerana*) has natural seasonal migrations in parts of its range (Koetz 2013), and other managed species, especially but not exclusively *A. mellifera*, are moved extensive distances especially in the USA (Daberkow et al. 2009). At a smaller scale, populations of *Megachile rotundata* are moved between alfalfa fields. Once a field has been pollinated, populations can be moved in large trailers to a newly blooming field (Osgood 1974). We continue to know very little about ways to manage migration and movement that minimize stress to bees
- Africanized honey bees: a specific topic related to these practices is the development of strategies for managing Africanized honey bees, especially in the tropical and subtropical Americas, in order to increase human safety concerns related to management as well as colony productivity (Winston 1992)
- stocking density of managed bees in crop fields and forage areas. Maintaining appropriate stocking densities can potentially increase crop yields and reduce costs to farmers and/or pollinator managers (e.g., Eaton and Nams 2012), and preventing overstocking could potentially reduce competitive interactions with wild pollinators (e.g., Thomson 2004), the risk of pathogen spillover from managed to wild pollinators (Otterstater and Thomson 2008), and speculatively the risk of pathogen transmission in managed pollinators

6.4.4.1.1.2 Manage pathogen and parasite threats

This is a very large category, with intensive work for both honey bees and bumble bees, along with a growing body of work on other managed pollinators (see Chapter 2 for an overview of disease threats). We focus on five major categories of responses related to disease: detection/diagnosis (6.4.4.1.1.2.1); prevention (6.4.4.1.1.2.2); treatment (6.4.4.1.1.2.3); supporting social immunity mechanisms in eusocial taxa (6.4.4.1.1.2.4); and management of pathogen and parasite evolution (6.4.4.1.1.2.5).

6.4.4.1.1.2.1 Detect / diagnose disease problems

Rapid, precise detection and diagnosis of parasite and pathogen threats are critical for understanding, treating, and controlling these threats in managed bees. For many parasites and pathogens with macroscopic visual cues, detection is well established based on apiary inspection, including macroscopic mites (Sammataro et al., 2000) and some fungal pathogens such as chalkbrood (Aronstein and Murray, 2010). For other pathogens, either microscopic analysis is needed, such as in tracheal mites (Sammataro et al., 2000; Otterstater and Whitten, 2004), or molecular methods are needed, such as in the microsporidian fungal parasite *Nosema* (Fries, 2010) and many viruses (de Miranda et al., 2010). There is considerable opportunity and a research gap for improving detection and diagnosis of managed bee pathogen and parasite threats. In particular, improvements could be made in terms of speed, reliability, and accessibility of diagnostic tests, as well as reduction of costs. Rapid developments in molecular genetic technology offer considerable promise on this front.

Another opportunity is to integrate detection of disease in a legal framework with registration and inspection of managed bees, as exists in some countries, including the UK (The Bee Diseases and Pests Control [England] Order 2006, SI 2006/342). Such a framework has the potential to contribute to prevention of widespread pathogen and parasite outbreaks.

6.4.4.1.1.2.2 Prevent infections

This is a broad category, which includes: 1) management of pollinator movement; 2) general management practices; and 3) rearing facility practices. As mentioned in the previous section, detection of parasite / pathogen threats in a legal inspection framework has considerable prevention potential. We discuss country- and continental-scale preventative measures (i.e., preventing introductions of parasites and pathogens) in the "legal responses" section 6.4.4.2.

Managing pollinator movement is a key method of disease prevention. Spatial scale is a critically important consideration. At very large, within-continent scales, many pollinators are moved considerable distances for crop pollination, especially (but not limited to) honey bees in the US (Pettis et al., 2014), and alfalfa leafcutter bees from Canada to the US (Bosch and Kemp, 2005; Pitts-Singer and Cane, 2011). These operations have potential to spread diseases long distances, but limiting their movement could reduce the provision of pollination to agriculture, and also reduce beekeeper profitability.

At a smaller spatial scale, we can consider movement of *Apis mellifera* colonies among multiple apiaries managed by the same beekeeper at a landscape or regional scale, as well as movement of brood or honey frames between colonies. Movement of bees or frames again has the potential to transmit disease, but stopping such practices altogether is unlikely to be practical for most beekeepers.

General management of pollinators can also contribute strongly to disease prevention. For example, chalkbrood is a fungal disease that is highly prevalent in managed populations of the alfalfa leafcutting bee, *Megachile rotundata* in the USA, where it can reach levels as high as 20-40%. Sorting loose *Megachile* cocoons and removing those with fungal infections can be an effective way to reduce infestation (Bosch and Kemp, 2005; James and Pitts-Singer, 2005; Pitts-Singer and Cane, 2011). Several products (including bleach, methyl bromide, paraformaldehyde, various fungicides) have been used to disinfect nesting materials with irregular success (Parker 1985, 1987, 1988; James 2005, 2008, 2011). In honey bee colonies, soil management can potentially help prevent infestations of small hive beetle (*Aethina tumida*), which pupate in the soil. For example, additions of diatomaceous earth and/or slaked lime management of soil near honey bee colonies can reduce pupation success and also kill adult beetles

(Buchholz et al., 2009). Maintaining appropriate stocking density of pollinators could potentially reduce parasite and pathogen transmission among managed pollinators and/or disease spillover between managed and wild pollinators, though research is needed on this topic.

Disease prevention practices in rearing facilities are a key concern for commercial bumble bee operations, which produce very high volumes of bumble bees and colonies in close proximity. Such facilities may increasingly be used in the future to rear solitary pollinators such as *Osmia lignaria*, which are currently largely provided to commercial markets by trap-nesting in the wild (Bosch and Kemp, 2002). There is a high level of secrecy and protection of intellectual property in commercial bumble bee rearing operations, and thus any particular rearing facility practices focused on disease prevention remain speculative. Because of disease problems in managed bumble bees (Velthius and Van Doorn, 2006), improved disease prevention in rearing facilities could potentially improve colony production and even profits.

6.4.4.1.1.2.3 Treat diseases

Disease treatment in managed bees is a critical component of pollinator management given the central role of parasites and pathogens in bee health. Treatments are organized here by the taxonomic group of the parasite / pathogen, rather than the pollinator host, because treatments are largely similar within taxonomically similar parasites and pathogens. This section covers treatment of viruses (6.4.4.1.1.2.3.1), bacteria (6.4.4.1.1.2.3.2), fungi (6.4.4.1.1.2.3.3), protozoa (6.4.4.1.1.2.3.4), mites (6.4.4.1.1.2.3.5) and other colony pests (6.4.4.1.1.2.3.5). One general issue with treatment is that the impacts of parasites and pathogens on managed pollinators are context-dependent. For example, *Varroa* mites, one of the most important parasite pressures on honey bees, have different effects on colony fitness in tropical and temperate environments (reviewed in Rosenkranz et al., 2010).

6.4.4.1.1.2.3.1 Viruses

As reported in Chapter 2, more than 20 bee-associated viruses have been identified, some of which contribute to substantial bee morbidity and mortality, in honey bees, bumble bees and managed solitary bees. Treatment options for viral diseases are limited in managed pollinators, and currently preventative measures are the best protection against viral infection. One potentially promising treatment method is interference RNA, or RNAi, in which double-stranded RNA is introduced into the host in order to silence the expression of one or more viral proteins, which replicate in host cells (Fire et al., 1998). RNAi has been demonstrated to reduce viral titer, and in some cases increase bee survival, in laboratory settings in

Apis mellifera infected with Israeli Acute Paralysis Virus (IAPV; Maori et al., 2009) and Deformed Wing Virus (Desai et al., 2012), and, in *Apis cerana*, of Chinese Sacbrood Virus (Liu et al., 2010). While RNAi technology seems to have considerable promise, it has not been widely used in field beekeeping settings, even though a relatively large-scale trial showed increases in total number of adult honey bees, forager activity, and honey production in RNAi-treated vs. untreated colonies when experimentally infected with IAPV (Hunter et al., 2010). This trial was sponsored and largely conducted by a commercial RNAi producer. Given that this trial was published five years ago, it remains unclear why RNAi technology has not had broader uptake; costs and incomplete viral clearance may contribute. There has been no assessment of the risks of RNAi technology or the costs of this technology relative to its benefits.

6.4.4.1.1.2.3.2 Bacteria

The primary known bacterial pathogens of managed bees are American and European Foulbrood (“AFB”, *Paenibacillus larvae*; and “EFB”, *Melissocccus plutonius*, respectively). These bacteria impact larval-stage bees, which if infected have very high mortality rates. Both are highly transmissible and capable of re-infecting larvae in the same colony in subsequent years after an initial infection (reviewed in Forsgren, 2010; Genersch, 2010). AFB in particular is spore-forming, and the spores are highly resistant to desiccation and remain infectious >35 years after an initial infection (Genersch, 2010). A single infected larva can produce millions of spores, and the infectious dose consists of as few as 10 spores (Genersch, 2010). Foulbrood of both types is mandatorily notifiable in many countries (Forsgren, 2010; Genersch 2010), including the UK (Wilkins et al., 2007; the Bees Act [UK] 1980; The Bee Diseases and Pests Control [England] Order 2006, SI 2006/342).

Three primary treatment mechanisms exist for foulbrood diseases (reviewed in Forsgren, 2010; Genersch, 2010): 1) colony eradication and subsequent destruction or sterilization of hive body equipment; 2) the “shook swarm” method, in which adult bees are shaken out of a colony and only the infected comb is destroyed; and 3) treatment with antibiotics. The first method, colony eradication, is considered the best method for reducing potential future infections, given the high level of transmissibility, but comes at the expense of colony and equipment losses (Wilkins et al., 2007; Forsgren, 2010; Genersch, 2010). Eradication is mandatory in some countries and localities for AFB infestation, and often recommended in colonies or apiaries with high infestation levels for EFB (Wilkins et al., 2007; Forsgren, 2010; Genersch, 2010).

The shook swarm method allows for maintaining adult bees from a colony while destroying infected brood and comb. The remaining components of hive body equipment are often sterilized with bleach or localized flame application (or ethylene gas, Robinson et al., 1972). The shook swarm method is often recommended for colonies infected with EFB (or in some cases AFB) but not yet clinically diseased (Genersch, 2010). A similar method, where brood are removed but adult bees maintained, is employed and reported to be effective in controlling foulbrood in China (Duan, 1992; Du et al., 2007).

Antibiotic administration is used by beekeepers for prevention and treatment of both EFB and AFB. Antibiotics reduce the reproduction of foulbrood bacteria but do not completely "cure" a colony of infection (Forsgren, 2010; Genersch, 2010). In particular, antibiotics do not operate on AFB spores (Genersch, 2010), leaving infested colonies open to subsequent re-infection from spores. Antibiotic treatment of honey bees for foulbrood is illegal in many European countries (Generesch, 2010) and EU food regulations prohibit any detectable levels of antibiotics in commercial honey (EEC Regulation 2377/90, 26 June 1990). Still, regulations vary among countries and for example antibiotic use is permitted in the UK for EFB only (not AFB) under some conditions, depending on the level of infection and the size of the colony (Wilkins et al., 2007). Antibiotic treatment remains legal in several other countries including the USA (e.g., under several NADA—New Animal Drug Application—and ANADA—Abbreviated New Animal Drug Application—numbers under the US Food and Drug Administration: NADA 008-622, NADA 008-804, NADA 095-143, NADA 138-938, ANADA 200-026, ANADA 200-247). In addition to incomplete infection clearance, an additional issue with antibiotic use is resistance. Tetracycline-resistant AFB was first reported in the US 15 years ago (Miyagi et al., 2000), and a subsequent intensive survey has since found widespread antibiotic resistance in the gut microbiota of honey bees, including at least 10 different resistance genes (Tian et al., 2012).

6.4.4.1.1.2.3.3 Fungi

The primary fungal pathogens of managed bees are *Nosema*, chalkbrood, and stonebrood. *Nosema* includes *N. apis* and *N. ceranae*, which typically infect bees in the genus *Apis* (e.g., Fries, 2010), as well as *N. bombi*, which infects a wide range of bumble bee species (Tay et al., 2005). Chalkbrood includes: *Ascosphaera apis*, which typically infects *Apis* (Aronstein and Murray, 2010); *A. aggregata* and other species that typically infect *Megachile* (Vandenberg and Steven ,1982; Bissett, 1988); and *A. torchioi* and other species that typically infect *Osmia lignaria* (Torchio, 1992; Sedivy and Dorn, 2013). Stonebrood is caused by several *Aspergillus* species that infect honey bees (Foley et al., 2014) as well as other bee species (Goerzen, 1991).

The primary treatment for *Nosema* in honey bees in many countries, including Canada and the USA, is the antifungal treatment agent fumagillin dicyclohexylammonium (“fumagillin”; Williams et al., 2008; Fries, 2010), though its use is illegal in the EU (Fries, 2010; Botías et al., 2013) given its toxicity to mammals including humans (Huang et al., 2013). While fumagillin can reduce *Nosema* levels in honey bee colonies in some circumstances (Webster, 1994; Williams et al., 2008), it appears to have some direct toxicity to honey bees, and low levels of fumagillin may also enhance, rather than reduce, *N. ceranae* reproduction in honey bees (Huang et al., 2013). Fumagillin was not shown to be effective in controlling *N. bombi* in bumble bees at either the recommended fumagillin dose for honey bees (26 mg/L in sugar syrup) or double that concentration (52 mg/L; Whittingdon and Winston, 2003).

A single study has also shown that RNAi, using gene transcripts for an ATP/ADP transporter specific to *N. ceranae*, when fed to worker bees, reduced infection levels and parasite reproduction within adult honey bee hosts (Paldi et al., 2010). We are unaware of field implementation of RNAi therapy targeted to *Nosema*. There has been no assessment of the risks of RNAi technology or the costs of this technology relative to its benefits. The lack of proven options other than fumagillin for *Nosema* treatment (Fries, 2010) represents an important knowledge gap.

Chalkbrood and stonebrood, irrespective of host bees that are infected, also have few direct treatment options (Bosch and Kemp, 2001; Aronstein and Murray, 2010; Sedivy and Dorn, 2013). As Hornitsky (2001) noted, “A wide range of chemicals has been tested for the control of chalkbrood. However, none has proved efficacious to the point where it has been universally accepted. A chemical which is effective against chalkbrood, does not produce residues in bee products and is not harmful to bees is yet to be found.” Still, there have been some promising developments including the use of formic acid and oxalic acid (also used in the treatment of *Varroa* mites), which reduced growth of *Ascosphaera apis* chalkbrood in vitro, but was not tested in live bees (Yoder et al., 2014). Similarly, a range of essential oils showed promise in reducing stonebrood growth in *in vitro* assays, but showed challenges in translating that antifungal activity to pollinator management situations (Calderone et al., 1994). A cultural practice for chalkbrood management in alfalfa leafcutting bees, *Megachile rotundata*, is that populations are often managed as loose cells (rather than entire natal nests) to prevent emerging adults from being dusted during emergence with chalkbrood spores from infested larval cadavers (Richards, 1984).

6.4.4.1.1.2.3.4 Protozoa

The primary protozoan parasite of managed bees is *Crithidia bombi*, which infects bumble bees (Shykoff and Schmid-Hempel, 1991). There is no known treatment for *Crithidia* (Schweitzer et al., 2012). At least two lines of promising evidence point toward treatment options in the future. First, gelsamine, a nectar alkaloid, has been found to reduce *Crithidia* levels in bumble bees (Manson et al., 2009), and second, horizontally-transmitted gut microbiota also have been shown to protect against *Crithidia* (Koch and Schmid-Hempel, 2011).

6.4.4.1.1.2.3.5 Parasitic mites

Mites are among the most destructive parasites of managed bees. The primary parasitic mites of managed honey bees are in the genera *Varroa, Tropilaelaps*, and *Acarapis* (reviewed in Sammataro et al., 2000; Rosenkranz et al., 2010), while *Locustacarus* impacts bumble bees (e.g. Shykoff and Schmid-Hempel; 1991; Otterstatter and Whidden, 2004). The negative health impacts of mites are exacerbated by a range of viruses that mites vector (Sammataro et al., 2000; Rosenkranz et al., 2010). Treatment of mites is challenging because bees and mites are both arthropods, and thus compounds that are toxic to mites are likely also to be harmful to bees. A range of different mite treatment and control methods have been developed for honey bees (but not for other managed pollinators), likely due to the substantial parasite pressure that mites exert on honey bees and their economic importance. Because of the particular importance of *Varroa*, the bulk of treatment methods focus on it. *Tropilaelaps* mites have a very similar natural history and thus many of the treatments used in *Varroa* have potential for use in *Tropilaelaps* (Sammataro et al., 2000). Existing treatment classes include: 1) acaricides / miticides; 2) RNAi; 3) organic acid vapors; 4) aromatic and essential oils; 5) biological / cultural controls.

The primary groups of acaricides / miticides are the organophosphate coumaphos, two pyrethroids (tau-fluvalinate and fluvalin), and amitraz, a formamidine (Sammataro et al., 2000; Rosenkranz et al., 2010). Amitraz is illegal in the US (Sammataro et al., 2000) and many other countries. While these compounds can greatly reduce mite populations, they have several drawbacks. First, they can harm bees because these compounds have insecticidal, not just acaricidal, impacts. Second, there is the potential for these products to contaminate hive products including honey. Third, and perhaps most important, *Varroa* resistance to all of these compounds is well documented in a very widespread geographic area (reviewed in Sammataro et al., 2000; Rosenkranz et al., 2010). These compounds are lipophilic and thus can become integrated and accumulate in beeswax for long periods, which exacerbates all three of the drawbacks to their use (Rosenkranz et al., 2010).

Interference RNA (RNAi) has been targeted against *Varroa*, and injection or soaking of double-stranded RNA directly into *Varroa* strongly and specifically reduced the transcription target in a laboratory context (Campbell et al., 2010). In addition, double-stranded RNA fed to bees was found to be passed intact to *Varroa*, and then back to developing bee brood (Garbian et al., 2012). This RNAi method also reduced

Varroa counts in laboratory colonies (Garbian et al., 2012). As with other RNAi methods utilized in the treatment of managed bee parasites and pathogens (with the exception of Hunter et al., 2010, working on Israeli Acute Paralysis Virus), RNAi for *Varroa* control has not been tested in field beekeeping scenarios and there has been no assessment of the risks or the costs of this technology relative to its benefits.

The main organic acid vapors used to control *Varroa* and *Acarapis* are formic, oxalic, and lactic acids. Multiple studies have evaluated the efficacy of these acids as well as different methods for administering them, and they are effective in reducing *Varroa* and *Acarapis* populations, though they do not necessarily provide complete clearance of mites from colonies (reviewed in Sammataro et al., 2000; Rosenkranz et al., 2010). Formic acid is the only known method of *Varroa* control that kills both adult phoretic mites and developing mites within sealed honey bee brood cells. Additional advantages of organic acids are that they are hydrophilic and do not accumulate in beeswax, and that to date there is no evidence of mite resistance to them (Rosenkranz et al., 2010). Disadvantages of organic acid use include contamination of hive products, and the suggestion (for oxalic and lactic acids) of use in honey bee colonies during broodless periods, which is not possible in all geographic areas and limits use to particular times of year. In addition, results are dependent on vapour pressure and other within-hive conditions, meaning that the effects of treatment are more variable than with some other control measures (Rosenkranz et al., 2010). There is some evidence of harm to bees from use of organic acids, and they can be hazardous to human applicators if not handled properly (Sammataro et al., 2000).

The primary essential oil used in control of *Varroa* is thymol, which can reduce mite populations by up to 90% (Rosenkranz et al., 2010). Other essential oils have been tested against *Varroa* but none with the consistent success of thymol, though more research is needed (Rosenkranz et al., 2010). For *Acarapis* tracheal mites, menthol has been shown to be an effective control measure, and the only other effective treatment besides formic acid (Sammataro, 2000). As with organic acids, treatment effects are variable and vapour pressure within colonies is an important consideration. Essential oils are lipophilic and can become integrated into beeswax, heightening potential for contamination of hive products (Rosenkranz et al., 2010).

Biocontrol of *Varroa* and other parasitic mites is a control strategy with some preliminary investigations, including laboratory demonstrations of lethality to *Varroa* of several different bacterial strains (Shaw et al., 2002), but other attempts have shown less impressive results, and no commercial products or field beekeeping trials have used this strategy (reviewed in Rosenkranz et al., 2010, Meikle et al., 2012). Biocontrol of parasitic mites (and other parasites and pathogens) thus represents an important knowledge gap.

Parasitic mites, especially *Varroa*, are also controlled by beekeeping practices and other cultural controls. One such practice that has shown efficacy is the use of “trap frames”. Gravid *Varroa* females prefer to lay their eggs in drone (male) brood cells relative to worker (female) brood cells. After the drone cells are capped, the drone brood can be removed, thus greatly reducing *Varroa* populations within a colony (Sammataro et al., 2000; Rosenkranz et al., 2010). Similarly, swarming management can provide some level of *Varroa* control given that departing swarms leave infected brood behind (Sammataro et al., 2000; Rosenkranz et al., 2010). Another method involves heating colonies to 44ºC, a temperature that bee brood can survive but which kills developing mites (Sammataro et al., 2000; Rosenkranz et al., 2010). A cultural practice used in the control of *Acarapis* tracheal mites is the addition of patties of vegetable shortening and sugar to colony boxes, which may disrupt the “questing” behavior of female mites searching for new hosts (Sammataro et al., 2000). These cultural practices are often labour intensive and difficult to implement in large apiary operations (Rosenkranz et al., 2010). In solitary bees, thermal shock treatments applied during the most resistant bee stage (dormant prepupa) are used in Japan to reduce numbers of *Chaetodactylus* mites in *Osmia cornifrons* populations (Yamada, 1990).

6.4.4.1.1.2.4 Support social immunity mechanisms in eusocial taxa

These are mechanisms by which social organisms help to prevent and treat pathogens and parasite infestations at a social (not individual) level (Cremer et al., 2007; Sadd and Schmid-Hempel, 2008; Evans and Spivak, 2010; Parker et al., 2011). This is a recently emerging area of study with limited, but growing evidence that it can have a large impact on disease pressure. Management to support social immunity could include provision of resin-producing plants so that honey bees can gather propolis and not removing propolis from colonies (Simone et al., 2009; Simone-Finstrom and Spivak, 2012), and dietary management to support honey hydrogen peroxide production (Alaux, 2010). A possible trade-off is that some practices interfere with typical beekeeping practices (e.g., removal of propolis). More field-scale trials of supporting social immune mechanisms would assist pollinator managers and policy makers in evaluating their implementation.

6.4.4.1.1.2.5 Manage pathogen and parasite evolution

This category includes two broad responses. First, development of resistance to insecticides and antibiotics is a well-known phenomenon in agriculture (Brattsten et al., 1986; Perry et al., 2011) and medicine (e.g., Neu, 1992), respectively, which has also been documented in honey bees in terms of resistance of *Varroa* mites to acaricides (Milani, 1999). There is a body of evolutionary theory on managing insecticide and antibiotic resistance, and lessons from this work could be applied to treatment of disease and parasites in managed pollinators. For example, the length of treatment, treatment rotations, and treatment combinations could be applied in ways to reduce resistance (e.g., Comins, 1977; Lenormand and Raymond, 1998). Second, there is a well-described relationship in evolutionary theory between transmission of pathogens and virulence (harm to the host), such that increased transmission tends to select for increased virulence (e.g., Ewald, 2004). While there is no direct evidence of such a relationship in managed pollinators, this pattern has been detected in a broad range of other host-pathogen systems (reviewed in Alizon et al., 2009). Steps could be made to assess this relationship in managed pollinators and potentially to alter management to select for less-virulent parasites and pathogens by reducing parasite transmission rates.

6.4.4.1.1.3 Genetic management

Genetic management, similar to general management, is focused on multiple goals. There are four main methods of genetic management: 1) traditional trait-focused breeding; 2) maintenance or enhancement of genetic diversity; 3) genetic engineering, i.e. development of transgenic pollinators; and 4) high-tech breeding. The first of these is traditional breeding for desirable traits, and in *A. mellifera* there have been extensive breeding efforts, in particular (though not exclusively) focused on hygienic behavior to reduce disease and parasites (Spivak and Reuter, 1998, 2001; Ibrahim et al., 2007; Büchler et al., 2010). These objectives have been successful in terms of target trait modification, but there is limited knowledge of how bees originating from such breeding programs perform relative to other lines, in managed apiary contexts, in terms of outcomes such as colony survival and productivity. While there is at least one report of bees from "hygienic" breeding programs outperforming typical (non-hygienic) stocks in terms of both disease resistance and honey production (Spivak and Reuter, 1998), other studies have not seen consistent advantages of bees bred for *Varroa* resistance (Rinderer et al., 2014). Maintaining the traits selected for in such breeding programs may be difficult in typical apiary settings for *A. mellifera*, given high levels of polyandry (queen mating with multiple, sometimes dozens of males) in honey bee queens and relatively large-scale movement of honey bee drones, especially given that trait maintenance appears to demand primarily drones expressing the traits of interest (Danka et al., 2011). In solitary bees, there were unsuccessful attempts in the late 1970s and early 1980s to select univoltine *Megachile rotundata* strains as a means to avoid an undesired partial peak of emergence in late summer (Parker, 1979; Rank and Rank, 1989).

The second strategy is maintaining and/or increasing genetic diversity, as this is known to reduce disease threats and to promote colony health and productivity at a colony level in both *Apis* (Tarpy, 2003; Mattila and Seeley, 2007) and *Bombus* (Baer and Schmid-Hempel, 1999). By contrast, other reports show mixed effects of diversity on colony performance, depending on the origin of single versus mixed lines (Oldroyd et al., 1992; Baer and Schmid-Hempel, 2001). In addition, beyond just social taxa, all currently managed pollinators are bees (Hymenoptera: Apoidea), which are haplodiploid with a single-locus sex determination system (Beye et al., 2003); it is thought that this system might make bees particularly susceptible to deleterious effects of inbreeding (e.g., Zayed, 2009). Still, to our knowledge there are no systematic efforts to increase genetic diversity in any managed bees that have been assessed in a rigorous way. A related issue is not just genetic diversity *per se*, but maintenance of locally adapted strains. There is recent evidence that local (geographically specific) strains of honey bees outperform non-local strains, which is a distinct argument for conserving and maintaining geographic genetic diversity in managed pollinators (Büchler et al., 2014).

There are trade-offs between these strategies in that breeding and genetic engineering are typically focused on replacing, or increasing the prevalence of particular alleles at particular loci. This goal is usually in direct conflict with maintenance of diversity. Still, multiple programs could exist with different goals, such as complementing existing *A. mellifera* bee breeding efforts with a program focused on enhancing genetic diversity.

The third method is the development of transgenic pollinators, (i.e., “genetic engineering”), which has been recently shown in principle with *A. mellifera* (Schulte et al. 2014), though not yet in full honey bee colonies, or to our knowledge in any other managed pollinator. There are risks associated with such an effort, and in polyandrous species such as *A. mellifera*, transgene containment might prove to be extremely difficult. These risks should be carefully assessed in the context of potential benefits before development of such transgenic pollinators.

The fourth method, which we describe as “high-tech breeding” can be thought of as a middle-ground approach between traditional breeding and transgenic approaches. For example, marker-assisted selection is an approach where genetic, phenotypic, and other markers associated with desired traits are identified in early stages of organismal development, speeding up the process of traditional breeding (e.g., Lande and Thompson, 1990; Collard and Mackill, 2008). This approach has been proposed for honey bee breeding (Oxley et al., 2010; Oxley and Oldroyd, 2010), but has not been conducted to our knowledge.

Speculatively, additional approaches could include up- or down-regulation or particular genes already present in the genome of managed pollinators.

6.4.4.1.1.4 Reduce pesticide threats

In this section, reduction of pesticide threats is specifically focused on beekeeping management strategies; more general and holistic treatment of managing pesticide threats to pollinators (including reducing exposure of bees) is covered in section 6.4.2. Beekeeping strategies to address pesticide threats remain largely speculative, but include improved nutrition, which has been shown to reduce the negative impacts of exposure to some classes of pesticides (Wahl and Ulm, 1983; Schmel et al., 2014); and speculatively, the development of chemical antidotes or chemical (or possibly even microbial) prophylaxis against pesticides. Still, such strategies are likely to be expensive and difficult to implement compared to better management of pesticide application.

6.4.4.1.1.5 Manage symbionts and commensals

This is very much an emerging topic in pollinator management. Commensal or symbiotic macro-organisms have been documented in social bee colonies, including chelifers ("pseudoscorpions") (Gonzalez et al., 2007; Read et al., 2013) and non-parasitic mites, which could potentially have positive impacts on colony health and fitness (for example, cleaning detritus from the colony) (e.g., Walter et al., 2002). The technical development of next-generation DNA sequencing has also revealed that most macro-organisms, including pollinators, host diverse communities of endosymbiotic microorganisms, and relatively recently work has shown that different communities of such microorganisms can have important effects on the health of honey bees and bumble bees, including disease resistance (e.g., Evans and Armstrong, 2006; Hamdi et al., 2011; Kwong et al., 2014) as well as nutrient availability (Anderson et al., 2011). There is significant potential for developing ways to manage these communities to support pollinator health, including among many others, probiotics. While this is a very active area of research, there remains a poor mechanistic understanding of how different microorganisms affect pollinator health, alone and in combination, and development of effective management may take several years.

6.4.4.1.2 Improve pollination efficacy of managed pollinators (crop-focused)

In contrast to sections 6.4.4.1.1-6.4.4.1.6, this section is focused on improving crop pollination by managed bees, rather than focusing on the health and productivity of the pollinators themselves. Nearly all work in this area has been with honey bees, and to a limited extent with bumble bees, the latter especially in greenhouse / glasshouse / polytunnel contexts. This is an area with some limited evidence, with more study needed. Work to improve provision of crop pollination could include: optimization of stocking densities and configuration of colonies / nests (in conjunction with crop configuration) (Delaplane et al., 2013); floral attractants such as pheromones; (Ellis and Delaplane, 2009; Sivaram et al., 2013); feeding adjuvants such as caffeine, which can improve bee memory of particular flowers (Wright et al., 2013); and combining pollination with delivery of other materials such as biofungicidal compounds (Mommaerts et al., 2009; 2011) to plants. In a greenhouse / glasshouse / polytunnel context in particular, work could focus on optimization of lighting (Johansen et al., 2011), as well as environmental parameters such as temperature, humidity, and airflow. A particular research need is assessment of potential trade-offs between pollination activity in the short term and individual pollinator / colony lifespan or other measures of health, particularly in the case of feeding adjuvants.

6.4.4.1.3 Develop alternative managed pollinators

A very small number of pollinator species are actively managed, especially relative to the diversity of pollinator species worldwide. There is potential to develop alternative pollinators, which could help to offset ongoing declines of managed pollinators. Within this realm, there are two main categories, first the use of existing managed pollinators on crops where they have not previously been in use. There is recent evidence for this with use of managed bumble bees in crops in which they had not previously been used, e.g., blueberry (Stubbs and Drummond, 2001). Second, there is potential for developing management techniques and practices for pollinators that had not previously been managed. Bumble bees for example, have only been commercially managed relatively recently (Goulson et al., 2008). Social stingless bees (meliponines) are one taxonomic category with potential for increased domestication (e.g., Heard, 1999), along with species of *Osmia* beyond *O. lignaria, cornifrons, cornuta,* and *bicornis* (Torchio, 1990; Drummond and Stubbs, 1997; Cane, 2005), extending to other solitary leafcutter bees such as *Eumegachile pugnata* (Parker and Frohlich, 1985). For both of these categories, a potential trade-off is that it increases the density and/or distribution of newly managed species, which could lead to disease issues such as pathogen and parasite spillover to other species of pollinators (Otterstater and Thomson, 2008), as well as competition for resources with local pollinator taxa (e.g., Huryn, 1997; Thomson, 2004; see Chapter 3, section 3.3.3 and Chapter 2, section 2.4.2.2).

6.4.4.1.4 Provide resources for managed pollinators (food/nesting)

Two general limiting factors for managed pollinators are food (flowering plants) and nest sites. See sections 6.4.1.1.1 and 6.4.3.1.1 for more on provision of nesting and flowering resources for wild pollinators. There is little concrete evidence that increasing food or nesting sites leads to long-term positive effects on managed pollinator populations. Still, a major issue for large migratory beekeepers in the USA is the lack of flowering plant forage along migration routes. An additional component of forage availability is evidence that diversity of forage plant sources plays a role in bee health (e.g., Alaux, 2010). The issue of forage availability is relevant at a range of scales, from local scales surrounding sites of active pollinator management, to larger scales that could benefit from landscape/regional coordination (see section 6.4.4.3 of this chapter). A possible trade-off is that managed pollinators could usurp resources from wild pollinators in such areas, and potentially even contribute to pathogen spillover (see Chapter 2).

6.4.4.1.5 Boost native pollinators by translocation

Increasing crop pollinators by translocation (i.e., moving pollinators to an area where they are not found naturally or where their abundances are low), is distinct from migratory pollinator management practiced by migratory beekeepers in the USA, and does not include moving pollinator species to entirely new regions, which is not recommended (see Chapter 3, section 3.3.3). There are anecdotal reports of almond growers in California, USA, conducting relatively large-scale translocation of *Osmia lignaria* from states such as Utah in the interior western USA where *O. lignaria* abundances are higher. This strategy could potentially be broadened and might also be used as an adaptive response to climate change, if flowering crops and their pollinators become mismatched in space and time (see Chapter 2, section 2.6.2.3, and this chapter, section 6.4.1.1.12). We found no studies of its effects on pollination. As with any response that involves large-scale pollinator movement, two potential trade-offs are the increased risk of disease issues, including pathogen and parasite spillover, and potential for competitive effects on local pollinator taxa (see Chapter 3, section 3.3.3 and Chapter 2, section 2.4.2.2).

6.4.4.2 Legal responses

Two key policy responses are first, registration and inspection of managed pollinators, and second, regulation of managed pollinator movement, for example related to imports of hive pests and trade in managed pollinators at a single country level, or movement restrictions related to diseases. A list of such regulations around the world is included in the reference list (Annex 1). In Australia, this has so far prevented the introduction of *Varroa* mites of honey bees (Cook et al., 2007).

As an example of within-country movement, in the UK, beekeepers whose colonies are infected with American Foulbrood (caused by *Paenibacillus larvae*) are mandated with standstill orders by the 1980 Bees Act, under the UK Bee Diseases and Pests Control Orders 2006, SI 2006/342. This policy mandate thus prevents spread of this highly contagious hive pathogen.

In dealing with multiple countries, there is significant potential for regional coordination of policies surrounding movement of managed pollinators, both within and between countries. Many countries and regions have regulations in place (e.g., in the UK, The Bee Diseases and Pests Control Order 2006 [2006 No. 342]; European Union Council Directive 92/65/EEC), though a key component of their success is border enforcement infrastructure. In addition, general biosecurity, beyond specific control of managed pollinators, is necessary to limit accidental introductions of managed bees and/or their parasites and pathogens (e.g., Cook et al., 2007).

An additional policy concern is the potential for mandated registration of managed bees, which again is common in many countries and regions for honey bees (e.g., the state of Maryland, USA, under Maryland code 15.07.01.02), but could be done for bumble bees, *Osmia*, and other species. Registration would potentially assist with monitoring efforts and pathogen containment. There is very limited systematic evidence on how either regulation of pollinator movement or mandated registration of colonies affects tangible outcomes related to managed pollinators.

6.4.4.3 Economic responses

Economic responses for managed pollinators include access to markets and market building, incentives for beekeepers and other pollinator managers, and product certification. Access to markets, as well as building existing markets, is particularly relevant for alternative or newly managed pollinators. Economic incentives including supports could potentially play an important role in markets, such as for pollination contracts, where there is year-to-year variability that may discourage particular beekeepers or other pollinator managers from entering the market.

Product certification involves three areas of consideration: the targeted product; the pollinator species involved; and the certification type. Product targets currently include honey and other hive products (including wax, propolis, royal jelly), as well as bees themselves (colonies, packages, pupal cases, queens, or even bee semen for breeding purposes); for example, EU Council Regulation No 1804/1999, of 19 July 1999 includes provisions for certifying any beekeeping product. While to our knowledge there is no thorough accounting of pollinator-related certification at a global level, at a species level honey bees and their products appear to account for the vast majority of certified products. Thus, there is a particular opportunity for developing certification for other species. Meliponine honey is a good example in that it already commands a price premium for its potential/perceived medicinal effects in parts of the world (Cortopassi-Laurino et al., 2006). In terms of types of certification, these include: organic; trademark;

quality; floral source; and geographic provenance. Again, while exhaustive surveys of certification types is lacking, organic certification and monofloral honey certification are very likely (but speculatively) the largest players. Product certification could also potentially be useful to protect indirectly biodiversity and traditional knowledge (Avril, 2008).

An example of protected monofloral honey is Manuka honey, produced from *Leptospermum scoparium* trees that grow in parts of New Zealand and Australia. Manuka honey commands a strong price premium for its perceived medicinal properties. The New Zealand Ministry for Primary Industries regulates labeling of Manuka honey, and in addition there are two Manuka honey trade groups that have licensed trademarks for Manuka honey meeting particular biochemical standards, though labeling of honey in New Zealand is under review at the time of this writing (http://archive.mpi.govt.nz/food/food-safety/manuka-honey, last accessed 11 December 2014).

An example of trademark-protected bees are Buckfast$^{\text{TM}}$ honey bees, which were bred at Buckfast Abbey in the UK in an isolated, treeless moor that lacks honey bee nesting habitat, thus allowing for careful selection and breeding, in particular against tracheal mites (Osterlund, 1983). The abbey has held various UK and EU trademarks, e.g., trademark EU003089224, to the Buckfast bees (http://www.ipo.gov.uk/tmtext, search for “buckfast bees”, 13 April 2015).

While various certification schemes for products from *A. mellifera* are well established and very likely enhance beekeeper livelihoods in some contexts, there is no direct evidence to our knowledge that such certification improves colony or crop pollination outcomes. In addition, to our knowledge there is no evidence for the efficacy of market-building responses.

In France, an agri-environment scheme under the European Common Agricultural Policy provides economic support directly to beekeepers who place hives in areas of high biodiversity (le dispositif apiculture (API); see section 6.4c).

6.4.4.4 Social and behavioural responses

The two main social and behavioural responses for managed pollinators are community engagement through participatory processes, and voluntary codes of practice.

Community engagement could specifically include better coordination of growers with beekeepers and other managers of pollinators, especially in terms of pesticide use (e.g., providers of *Osmia* spp. to orchards, and alfalfa seed farmers who manage *Nomia melanderii* in the USA and Canada). It could also include provision of forage for managed bees at relatively large scales, including, for example, along beekeeper migration routes.

An example of the benefits of communities working together comes from Kenya (Rose et al., 2014). In 2009, the Kenyan Ministry of Agriculture, Livestock and Fisheries in partnership with World Neighbours, a development organization, began working with farmers to introduce beekeeping as a way to diversify livelihoods. Women were provided with new beehives and received training and technical support from Ministry of Agriculture extension workers (Atakos and Recha, 2013). Women's groups formed to support and empower each other and average honey yields doubled from about 5 kg per beehive/year to 10 kg and above (Macoloo et al., 2013). Some groups split earnings among the group or reinvest them into group functions. In addition to the economic benefits from honey production, neighbouring farmers have also experienced improved yields with their mango trees (Atakos and Recha, 2013). This case study offers an example of a government programme that not only promotes pollination, but also reduces poverty and empowers rural women.

There are examples of community-based voluntary codes of practice relating to managed pollinators. In the Mbulu highlands (Tanzania), there is a general agreement that bees and beehives should not be disturbed (Tengo & Belfrage, 2004). In the Kobo system in Ethiopia, families own groups of trees in which they can place their bee hives. These trees cannot be cut down and no one else can use these trees for beekeeping (Abebe and Lowore, 2013). The community tradition was recognized and strengthened by a forest protection agreement developed as part of participatory forest management, under the Ethiopian Government's Non-Timber Forest Product and Participatory Forest Management (NTFP-PFM) project (Abebeand Lowore, 2013). Similar practices could be enacted as part of a bio-cultural community protocol in the future (Bavikatte and Jonas, 2009).

6.4.4.5 Knowledge responses

There are four primary knowledge responses associated with managed pollinators. The first two are related to improved data on general properties of managed pollinators, first, monitoring and evaluation to give a big-picture idea of threats at large scales, and second, work to quantify the economic dimensions of managed pollinators, in particular their benefits. Previous work has shown that large-scale monitoring is very valuable in identifying threats at large spatial and temporal scales (e.g., Genersch et al., 2010; Pettis and Delaplane, 2010). Economic valuation efforts have been helpful but have tended to give very large ranges in valuation estimates, in part depending on the valuation methodology used (see Chapter 4).

A third knowledge response is improvement in technical knowledge transfer, in particular to farmers and beekeepers. While there is significant agreement that such knowledge transfer could improve pollinator management, there are few if any data on the effects of, e.g., beekeeper education on tangible outcomes such as large-scale colony health.

The fourth response is maintaining and documenting traditional and indigenous knowledge surrounding managed pollinators, including its application to modern pollinator management practices and incorporation into global markets (see Chapter 5, section 5.4.10). Such knowledge is focused on management of social stingless bees (meliponines) and honey bees (including both *A. mellifera* and *A. cerana*).

38. Table 6.4.4. Summary of evidence for responses relating to pollinator management and beekeeping.

Response (section of Chapter 6)	**Main Driver(s) (section of Chapter 2)**	**Type of Response**	**Status**	**Scientific Evidence**
Improve husbandry of managed pollinators: general management (6.4.4.1.1)	Pollinator Management (2.4.2)	Technical	Established, tested, or proposed depending on specific response	Management techniques can reduce losses of managed bees and increase production of hive products (WELL ESTABLISHED), but many specific techniques remain untested or poorly tested, especially in bees other than honey bees.
Improve managed bee husbandry: manage disease threats (6.4.4.1.1.2)	Pollinator Parasites and Pathogens (2.4.1 and 2.4.2)	Technical	Established, tested, or proposed depending on specific response	Disease management techniques can reduce morbidity / mortality of managed pollinators (WELL ESTABLISHED), but many specific techniques and treatments remain untested or poorly tested.

Response (section of Chapter 6)	Main Driver(s) (section of Chapter 2)	Type of Response	Status	Scientific Evidence
Improve managed bee husbandry: genetic management (6.4.4.1.1.3)	Pollinator Management (2.4.2)	Technical	Established, tested, or proposed depending on specific response	Successful honey bee breeding programs have been carried out for disease resistance and other traits (WELL ESTABLISHED); strong evidence that genetic diversity enhances disease resistance in social bees (WELL ESTABLISHED); some evidence that locally adapted strains can outperform non-local strains of honey bees (ESTABLISHED BUT INCOMPLETE); and preliminary work has been done in creation of transgenic honey bees (INCONCLUSIVE). Maintenance of breeding efforts in typical apiary situations is challenging and there remains no testing of management for genetic diversity or of transgenic bees.
Improve managed bee husbandry: manage pesticide threats (at the level of the beekeeper or pollinator manager, distinct from general management of pesticide threats) (6.4.4.1.1.4)	Pesticides (2.3.1)	Technical	Established	Improved diet confers some pesticide resistance to bees (ESTABLISHED BUT INCOMPLETE EVIDENCE); veterinary prophylaxis or treatment (i.e., antidotes) to limit or prevent pesticide damage could be developed. (INCONCLUSIVE)
Improve managed bee husbandry: management of pollinator symbionts (6.4.4.1.1.5)	Pollinator Management (2.4.2)	Technical	Proposed	Gut bacterial communities of bees can help to support health (ESTABLISHED BUT INCOMPLETE), and macro-symbionts such as mites and pseudoscorpions could potentially improve colony or individual pollinator health (INCONCLUSIVE). No known explicit testing of management interventions.

Response (section of Chapter 6)	**Main Driver(s) (section of Chapter 2)**	**Type of Response**	**Status**	**Scientific Evidence**
Improve pollination efficacy of managed pollinators (6.4.4.1.2)	Pollinator Management (2.4.2)	Technical	Established, tested, or proposed depending on specific response	These actions are focused on improving plant pollination outcomes, rather than on pollinator outcomes. They include optimizing pollinator stocking densities and configurations (ESTABLISHED BUT INCOMPLETE EVIDENCE); chemical attractants and feeding adjuvants (INCONCLUSIVE); and adjustment of glasshouse / polytunnel environmental parameters such as lighting, temperature, and humidity (ESTABLISHED BUT INCOMPLETE) to increase pollination and crop production.
Develop alternative managed pollinators (both existing and new) (6.4.4.1.3)	Pollinator Management (2.4.2)	Technical	Established and proposed; unclear how established information would transfer to new developments	Management strategies for several previously unmanaged pollinator species have been developed over the last 30 years. While there is high confidence that previous efforts were successful, it is unclear how that will translate to new developments. ESTABLISHED BUT INCOMPLETE
Provide resources for managed pollinators (nectar/nesting) (6.4.4.1.4)	Land Use and its Changes (2.2)	Technical Social / Behavioural	Tested	While there is strong evidence that enhanced resource provision on farms can increase pollinator diversity and abundance, and widespread agreement among migratory beekeepers for the need for greater access to floral resources, there is no direct evidence as yet that increased resource provision will improve outcomes for managed pollinators. ESTABLISHED BUT INCOMPLETE

Response (section of Chapter 6)	**Main Driver(s) (section of Chapter 2)**	**Type of Response**	**Status**	**Scientific Evidence**
Boost native pollinators by translocation (6.4.4.1.5)	Pollinator Management (2.4.2)	Technical	Proposed	Pollinators could be moved between locations to enhance plant pollination or pollinator population outcomes (distinct from migratory beekeeping). INCONCLUSIVE
Regulate import of hive pests & trade in managed pollinators (6.4.4.2)	Pollinator Management (2.4.2)	Legal	Established	Can prevent or limit the spread of parasites and pathogens of managed pollinators. ESTABLISHED BUT INCOMPLETE
Product certification for products from managed pollinators (6.4.4.3)	Pollinator Management (2.4.2)	Economic	Proposed	Certification improves livelihoods for beekeepers and other pollinator managers, but no formal assessment whether certification improves pollinator or plant pollination outcomes. ESTABLISHED BUT INCOMPLETE
Build markets for managed pollinators (6.4.4.3)	Pollinator Management (2.4.2)	Economic	Proposed	Limited assessment ESTABLISHED BUT INCOMPLETE
Community engagement through participatory processes (6.4.4.4)	Pollinator Management (2.4.2)	Social / Behavioural	Tested	Limited assessment of effectiveness, but widespread agreement that collaborative engagement would be beneficial. ESTABLISHED BUT INCOMPLETE
Voluntary codes of practice (6.4.4.4)	Pollinator Management (2.4.2)	Social / Behavioural	Tested (ILK)	Limited assessment of effectiveness. Some examples from indigenous and local knowledge. INCONCLUSIVE
Better education (farmers, beekeepers) (6.4.4.5)	Pollinator Management (2.4.2)	Knowledge	Tested	While there is widespread agreement that better education could lead to improved pollinator and pollination outcomes, this concept has not been formally tested. ESTABLISHED BUT INCOMPLETE

Response (section of Chapter 6)	Main Driver(s) (section of Chapter 2)	Type of Response	Status	Scientific Evidence
Maintain and document traditional and indigenous knowledge surrounding beekeeping and honey hunting (6.4.4.5)	Pollinator Management (2.4.2)	Knowledge	Tested	there is strong agreement of the value of such a proposition, but it needs more concrete assessment ESTABLISHED BUT INCOMPLETE
Monitor and evaluate managed pollinators (6.4.4.5)	Pollinator Management (2.4.2)	Knowledge	Established	Large-scale monitoring programs have been shown to collect and synthesize information effectively on threats to honey bees, allowing coordinated responses (WELL ESTABLISHED), but such programs remain untested in other pollinator species.
Quantify the benefits of managed pollinators (valuation incentives) (6.4.4.5)	Pollinator Management (2.4.2)	Knowledge	Proposed	Large-scale efforts to quantify the economic value of managed pollinators are useful but inherently give large value ranges. ESTABLISHED BUT INCOMPLETE

6.4.5 Urban and transport infrastructure

This section considers responses that specifically take place in urban or suburban contexts, or are associated with built infrastructure such as roads, railways and powerlines. The impacts of urbanization, and patterns of pollinator diversity and abundance in urban areas are discussed in section 6.2.1.1.

6.4.5.1 Technical responses

6.4.5.1.1 Conserving pollinators' habitat

Urbanization has been demonstrated as a threat to pollinator conservation by causing habitat loss and fragmentation (McKinney, 2008). In a 2009 review, Hernandez et al. suggested that conserving larger fragments is positive for conservation because smaller urban habitat fragments generally harboured lower bee species diversity than larger (Viana et al., 2006; Nemésio and Silveira, 2007; Hinners, 2008). This has been further supported in studies from Germany (Dauber et al., 2003), Brazil (Zanette et al., 2005; Martins et al., 2013), Sweden (Ahrné et al., 2009), UK (Bates et al., 2011), Switzerland (Sattler et al.,

2010) and USA (Tonietto et al., 2011; Hostetler and McIntyre, 2001), but there are huge remaining knowledge gaps for other countries. Restoring grasslands, even if not targeted specifically for pollinators, can provide valuable habitat (Tarrant et al., 2013). For instance, Cane et al. (2006) found that bee species diversity in Tucson, Arizona in the USA was reduced in small and older desert fragments, but bee abundance was similar to that found in continuous desert patches outside the urban area, which confirms the value to conserve remnant habitat. Also, the diversity of pollinator traits such as nesting habits, diet or body size were affected by habitat loss due to urbanization, which may alter the role of pollinators for ecosystem functioning (e.g., Banaszak-Cibicka and Zmihorski, 2012; Zanette et al., 2005; Bates et al., 2011, Sattler et al., 2010).

Little is known about how the flow of genes might be supported by maintaining habitat in urban settings. Conserving remnant habitat in urban landscapes may enhance genetic flow among pollinator populations. In a unique study, Jha and Kremen (2013) examined regional genetic differentiation of *Bombus vosnesenskii* across a landscape mosaic of natural, agricultural, urban and suburban habitats. They found that *B. vosnesenskii* regional gene flow is most limited by commercial, industrial and transportation-related impervious cover linked to urbanization. Importantly though, the effects of urbanization are not common across all studies; several show no negative impact of urbanized landscape on local pollinator communities (Bates et al., 2011), and urban areas can become important habitat for pollinators in intensively managed landscapes (Baldock et al., 2015). Also, when a statistically significant relationship has been found, some of the previously mentioned studies show that urbanization explains a low proportion of the variation in pollinator community composition compared with other local and landscape factors. Conservation of pristine habitat should, thus, be combined with other actions to support pollinators in urban landscapes (e.g., Bates et al., 2011; Sattler et al., 2010).

6.4.5.1.2 Urban landscapes

Conservation of pollinators in cities depends on the composition of the surrounding landscape. Strong relationships between landscape heterogeneity and bee species richness have been found, indicating that the availability of diverse resources for the pollinators in the landscape play a great role to maintain a rich local community (Sattler et al., 2010). Certainly, habitat connectivity can bolster a species-rich pollinator community within an urban area. For example, bee abundance on green roofs and in managed green spaces in Zurich, Switzerland was positively correlated with connectivity to surrounding habitat (Braaker et al., 2014). Managing for a less hostile “softened” matrix where some resources and habitat stepping stones are available in urban or ruderal areas, may increase conservation of pollinators in remnant high quality habitats and in the landscape. This was demonstrated in southeastern Brazil, where generalist stingless bee diversity in urban forest fragments was driven by forest composition as well as the heterogeneity and quality of the surrounding landscape (Antonini et al., 2013). In fact, several recent studies emphasize the importance of considering both the quality of local urban habitats as well as the surrounding landscape for the successful conservation of pollinators (Jules and Shahani, 2003; Bates et al., 2011; Ahrné et al., 2009). We also see reciprocal effects, with urban habitats influencing bee communities in surrounding natural areas (Hinners et al., 2012; Neame et al., 2013). For example, Hinners et al. (2012) studied diversity, abundance, and community composition of bees in remnant grassland fragments surrounded either by suburban residential areas or by extensive, continuous grassland in Colorado, USA. They found that bee species richness was positively related to grassland habitat area, and that bee species density was higher and more variable in suburban sites probably by means of habitat complementation or supplementation between grassland remnants and the surrounding suburbs.

Researchers have also begun to study how landscape context influences the pollination provided by bees in cities. Verboven et al. (2014) examined flower visitation and seed set of the obligatory outcrossing *Trifolium repens* (white clover) in public lawns in an urban-peri-urban gradient around Leuven, Belgium. They found that pollination was not compromised by urban land use. Greater abundance of *T. repens* in lawns and increasing urban area in the surrounding landscape both had a positive effect on both flower visitation rates and seed set. In this and many studies, however, a lack of mechanistic understanding of the population processes causing these patterns limits advancement in urban-focused conservation. For instance, this finding could be due to urban areas supporting an increased abundance of bumble bees, thus demonstrating a value for conservation, or due to urban sites concentrating bumble bees onto a small number of lawns due to a lack of alternative forage. The structure of landscape elements can also influence pollinator movement and directly affect plant reproductive success. Both hedgerows and artificial linear landscape features can influence the flight directions of bumble bees (Cranmer et al., 2012). Pollinator activity, pollen receipt and subsequent seed set on sentinel plants increased in patches with more connections (Cranmer et al., 2012). This knowledge has yet to be translated into specific actions.

Thus, managing the surrounding landscape to be more hospitable has potential to mitigate the negative impact of habitat loss and fragmentation. Despite the demonstrated negative impacts of urbanization, it’s important to note that relatively intact pollinator communities can be maintained in urban areas, both in boundaries between urban and rural areas such as in sub- and pen-urban landscapes (e.g., Hostetler and McIntyre, 2001; McFrederick and LeBuhn, 2006; Kearns and Oliveras, 2009; Carper et al., 2014). These ideas have not yet been widely tested or implemented, but an effort to create "Pollinator Pathways" in cities is underway, with a significant pilot study partially installed in Seattle, Washington, USA (Bergmann, 2015).

6.4.5.1.3 Urban green spaces

Urban green spaces are in focus when managing for a more pollinator-friendly landscape. Greenspaces may be privately owned yardscapes, allotments, parks, public gardens, cemeteries, golf courses, infrastructure right-of-ways, or green roofs (Kadas, 2006). They vary in their value for pollinator conservation depending on the availability of pollen, nectar and nesting resources, all of which are important factors for designing landscapes that support plant pollinator assemblages (Cane, 2005). An opportunity to maintain rich pollinator communities in urban settings lies in the appropriate management of gardens and allotments.

Increasing the abundance of flowering plants and floral area of blooms in urban green spaces can increase pollinator diversity and abundance (Dicks et al., 2010). For example, establishing a strip of meadow vegetation, a sunflower patch, or reducing weeding in small French public gardens tripled the abundance of residential butterflies and increased the abundance of other pollinators by nearly 50% (Shwartz et al., 2014). Richness of both butterflies and bees was positively related to garden floral area in New York City, New York, USA (Matteson and Langellotto, 2010). Researchers have also investigated whether the origin and structure of flowering plants influences their attractiveness. Native plants support both generalist and specialist bees (Isaacs et al., 2009; Tuell et al., 2008), but they represent only a fraction of available floral resource within a complex city landscape, often dominated by non-native weedy species and ornamentals (Gardiner et al., 2013). Addition of native or locally-adapted vegetation has given variable results. The addition of native plants to urban food gardens did not influence the pollinators in New York City gardens (Matteson and Langellotto, 2010). In Phoenix, Arizona, engaging in locally-adapted dry desert landscaping practices in residential landscapes gave a more diverse bee community than irrigated yards (Hostetler and McIntyre, 2001). Clearly, non-native plants also offer important resources to pollinators (Frankie et al., 2009; Woods, 2012; Frankie et al., 2013; Hanely et al., 2014; Garbuzov and Ratnieks, 2014). In Puebla, Mexico, local plants with many different uses are cultivated in home yards (Blanckert et al., 2004). In Moscow, Russia, lawn management for conserving pollinators has been performed recently by sowing native wild herbs as well as imitating Russian traditional meadow management with mosaic mowing about half of the lawn one time per year (Volkova and Sobolev, 2004). While not specifically for pollinators, this preserves natural habitat for pollinators.

Schemes exist to help people select appropriate plants for urban green spaces such as gardens. For example, the UK Royal Horticultural Society’s Perfect for Pollinators scheme (https://www.rhs.org.uk/science/conservation-biodiversity/wildlife/encourage-wildlife-to-your-garden/plants-for-pollinators) provides regularly-updated plant lists to help gardeners identify plants that will provide nectar and pollen for bees and other pollinating insects.

6.4.5.1.4 Retain unmanaged urban land

Retaining unmanaged areas in urban landscapes can provide important habitat for bees in cities (Tommasi et al., 2004; McFredrick and LeBuhn, 2006; Gotlieb et al., 2011; Gardiner et al., 2013). Unmanaged areas include forest, grassland or desert fragments as well as vacant land or brownfields that were formerly residential or industrial space. In a review, Gardiner et al. (2013) found that urban vacant lots or brownfields are valuable for beneficial arthropods and that these habitats also support a significant diversity of rare and threatened species including pollinators. Bumble bee abundance was positively correlated with the abundance of unmanaged undeveloped areas, or areas not actively landscaped, in the parks in the city of San Francisco, US, and there was a positive correlation with the openness of the surrounding matrix illustrating that these pollinators colonize urban parks from surrounding habitats (McFrederick and LeBuhn, 2006). Gotlieb et al. (2011) compared bee communities in natural desert and garden habitats in the Jordan Rift Valley in Israel, and found that bees in gardens were more abundant and general in their diet, whereas rarefied bee species richness was greater in the natural habitat.

6.4.5.1.5 Adding artificial nests and food

Urban residents may also add shelter and artificial food sources, and significant efforts have been made in some cities to add nesting habitat in the form of “bee hotels”. Artificial nest sites for cavity-nesting solitary bees have good occupancy rates and have been shown to enhance local populations over time (Dicks et al., 2010). The value of several types of artificial nests for solitary and social bees has been tested. Sections of bamboo, paper tubes and wooden blocks with holes ranging from 4-10 mm in diameter were added to gardens as nesting sites for bees and wasps and it was found that both design and placement influenced colonization. Nest boxes for bumble bees have much lower success rates, with underground boxes the most effective, and no evidence that they lead to increasing colony densities over time (Dicks et al., 2010). In Toronto, Canada, introduced bees occupied larger proportion of nests and were less parasitized compared with native bees (MacIvor and Packer, 2015). Bundles of twigs and plastic tubes were colonized by Megachilidae in gardens in Liege (Jacob-Remacle, 1976). Canes from *Spathodea campanulata, Ficus,* and bamboo have been found to support *Xylocopa* (carpenter bees) in urban greenspaces (Charves-Alves, 2011). Although many of these artificial nests were colonized by bees, their effects on species richness or population-level abundances of bees in the urban landscape have not been measured. It is possible that placement of artificial nests increases awareness about pollinators among citizens, but this has not been tested. Artificial nests need to be managed; otherwise, disease(s) and parasites may build up over time (Mader et al., 2010).

There is little research to date into how the addition of artificial food may influence pollinator communities. One study by Arizmedi et al. (2007) found that the addition of nectar feeders can influence visitation and subsequently the pollination of native plants by hummingbirds. Therefore, impacts of practices aimed to supplement food should be investigated further, given their ability to alter important ecological relationships.

6.4.5.1.6 Management of right-of-way infrastructure

Early successional habitat created by right-of-way management is increasingly considered valuable for pollinator conservation (Wojcik and Buchmann, 2012). The areas these habitats occupy are huge (Wojcik and Buchmann, 2012). Several studies have examined right-of-way linear elements such as road verges, power lines and railroad corridors as areas for active pollinator management, and they are often found to be valuable (Way, 1977; Bhattacharya et al., 2003; Tischendorf and Treiber, 2003; Desender, 2004, Russell et al., 2005; Noordijk et al., 2009; Osgathorpe, 2012; Berg et al., 2013). Butterflies benefit from the presence of native plants on roadsides, as shown by North American and European studies (Ries et al., 2001). Berg et al. (2013) found that power-line corridors harbored more butterfly species, higher abundances and a tendency for more individuals of red-listed species than road verges, clear-cuts, or pastures. Byrne et al. (2007) found that road verges were important in maintaining landscape-scale genetic connectivity of a bird-pollinated shrub. A replicated controlled trial in Kansas, US found that road verges planted with native prairie grasses and flowers supported a greater number and diversity of bees than paired conventionally managed verges (Hopwood, 2008). Moroń et al. (2014) found that railway embankments positively affected bee species richness and abundance, but negatively affected butterfly populations. Importantly, management efforts to encourage pollinators must also satisfy the highway engineers, and must be developed in a collaborative manner (Way, 1977). Further, the limitations of these habits should be considered as the presence of cars may disrupt or kill foragers (Hirsch, 2000). Also the potential for contamination within these habitats exists. Jablonski et al. (1995) found metal (Pb, Cd, Cu) contamination of nectar, honey and pollen collected from roadside plants. In many countries there is an interest in managing these habitats for biodiversity, but this response must be considered to be proposed but with great potential. There are right-of-way management programs for pollinator conservation underway such as the "B-lines" project in the UK (https://www.buglife.org.uk/campaigns-and-our-work/habitat-projects/b-lines), aiming to restore 150,000 ha of flower-rich habitat in the UK. In the US, Iowa installed in 1989 a program to establish roadside native vegetation funded partly by road use tax, by which 50,000 ha of roadsides have been planted with native vegetation (Brandt et al., 2011) that benefits pollinators (Ries et al., 2001). In the US state of Minnesota restored native plant habitat has been established along roadsides (The Xerces Society, 2011).

6.4.5.2 Legal responses

Some national pollinator strategies (see section 6.4.6.2.2) have specific actions to enhance pollinator habitat in towns and cities. A focus of these is on providing evidence-based guidance to local authorities, landscape planners and architects. We found no examples of strict regulations relevant to managing pollinators associated with urban areas or infrastructure developments.

Having said that, urban green space habitats are often ignored in conservation plans despite their value, an issue that must be addressed (Harrison and Davies, 2002; Muratet et al., 2007; Kattwinkel et al., 2011).

6.4.5.3 Economic responses

We know of no economic incentive programs similar to those present within agricultural landscape that support conserving habitats for pollinators and other beneficial biodiversity in cities or infrastructure.

6.4.5.4 Social and behavioural responses

6.4.5.4.1 Community engagement

Urban residents are interested in conserving and enhancing pollinators by assisting with monitoring networks, construction of pollinator gardens and addition of artificial food and nesting resources (see section 6.4.6.3.4). There are plenty of examples of NGOs that promote private and public land managers to support pollinators in the urban landscape by decreasing pesticide use and providing flowers and nests in their gardens *etc.* (e.g., http://www.xerces.org/wp-content/uploads/2008/06/Pollinator-Conservation-in-the-Portland-Metro-Area.pdf, http://www.sef.nu/smakrypsguiden/smakryp-som-hobby/skapa-din-egen-insektstradgard/), but we found no applied policies to stimulate this kind of action at the community level. Many green-space habitats are ignored in conservation plans despite their value, an issue that must be addressed (Harrison and Davies, 2002; Muratet et al., 2007; Kattwinkel et al., 2011). One step in that direction came in 2014 when the US President, Barack Obama, established the Pollinator Health Task Force. One of the key goals of this initiative is the development of plans and policy to establish or protect pollinator habitats. The U.S. government has subsequently issued a National Strategy to Promote the Health of Honey Bees and Other Pollinators (https://www.whitehouse.gov/sites/default/files/microsites/ostp/Pollinator%20Health%20Strategy%202015.pdf), which outlines actions that various federal agencies are taking as well as identifying research to address uncertainties; a key element of this strategy is the development of public/private partnerships.

Urban food production has grown rapidly worldwide with citizen groups constructing food gardens that include pollinator resource plants (Gardiner et al., 2013). Management of these small-scale gardens and farms may include the addition of managed honey bees or rely solely on existing pollinator communities for crop pollination.

39. Table 6.4.5. Summary of evidence for responses relating to urban transport and infrastructure.

Response	**Main drivers**	**Type of response**	**Status**	**Scientific evidence**
Conserve and restore pollinators' habitat in urban settings (6.4.5.1.1)	Land use and its changes (2.2)	Technical	Established in many cities	Conserving or restoring natural habitats in cities has positive effects on pollinator diversity, especially if habitat patches are large. WELL ESTABLISHED
Manage urban landscapes (6.4.5.1.2)	Urban management (2.2.2.3)	Technical	Proposed	ESTABLISHED BUT INCOMPLETE
Manage urban and recreational green space, e.g., parks, sport fields, gardens, golf courses (6.4.5.1.3)	Urban management (2.2.2.3)	Technical	Established in many cities	Managing urban green spaces by growing more flowering plants increases local abundance and diversity of pollinators. ESTABLISHED BUT INCOMPLETE
Retain unmanaged areas in cities, e.g., ruderal and vacant lands (6.4.5.1.4)	Land use and its changes (2.2)	Technical	Proposed	Ruderal and vacant land in cities has large potential to support pollinators, if managed appropriately. INCONCLUSIVE
Add nests and artificial food (6.4.5.1.5)	Urban management (2.2.2.3)	Technical	Established in many cities	Adding nesting boxes to urban green spaces possibly increases awareness but the effect on urban biodiversity is unclear. UNRESOLVED
Manage right-of-way land, e.g., road verges,	Urban management (2.2.2.3)	Technical	Proposed	Road verges, power lines and railway banks hold large potential for supporting

power lines, railway banks (6.4.5.1.6)				pollinators, if managed appropriately to provide flowering and nesting resources. ESTABLISHED BUT INCOMPLETE
Providing guidance through urban planning (6.4.5.2)	Urban management (2.2.2.3)	Legal	Proposed	INCONCLUSIVE
Community engagement (6.4.5.4.1)	Urban management (2.2.2.3)	Social & Behavioral	Tested	ESTABLISHED BUT INCOMPLETE

6.4.6 Policy, research and knowledge exchange across sectors

This section explicitly reviews responses that cut across sectors, such as large-scale land use planning, education and engagement, and community engagement through participatory processes. It compiles global experience of developing broad pollinator policy or actions and considers how research and monitoring needs have been met, and could be met in the future.

6.4.6.1 Summary of experience across sectors

Across the policy sectors in this section (agriculture, pesticides, nature conservation, managed pollinators and urban/transport infrastructure), some common themes emerge about available responses and the evidence for their effectiveness.

Technical responses are the most widely established and the most scientifically tested. For many of those relating to land management, such as planting flowers, or restoring semi-natural habitat, there is high confidence in positive effects on pollinators themselves, with many studies showing that pollinators make use of new resources provided for them (biodiversity). There is much less evidence of longer-term effects on pollinator populations, and limited evidence of effects on pollination.

Economic and legal responses tend to be established, with some evidence of impacts on pollinators and pollination. Regulatory control through obligatory registration and standards (legal responses) are most strongly established in the pesticides sector (6.4.2), and there is evidence they reduce risks to pollinators. Among economic market-based instruments, voluntary incentives such as certification or agri-environment schemes are established in some regions in the agriculture and managed pollinator sectors

(6.4.1 and 6.4.4). Taxes, which are obligatory market-based instruments, have been proposed to discourage pesticide use, but not tested.

Social/behavioural responses, even those that are established, seldom have robust evidence of effectiveness. Many examples come from indigenous and traditional knowledge, such as voluntary codes of practice among farming and beekeeping communities and community groups working together (6.4.1 and 6.4.4).

Knowledge responses related to ongoing research are generally known to be effective in enhancing knowledge and improving responses, whereas those related to education and awareness-raising usually have limited evidence to demonstrate effectiveness. Exceptions to this are the evidence on ability of Farmer Field Schools to change pest management practices (see section 6.4.2.4.2) and evidence that outreach programmes led by the Xerces Society for Invertebrate Conservation in the USA have created pollinator habitats (Xerces Society, 2014).

Indigenous and local knowledge particularly enhances scientific knowledge in the area of diversified farming systems (5.2.8 and 6.4.1.1.8), knowledge responses in agriculture (6.4.1.5), non-timber forest products (6.4.3.1.3), species-focused conservation actions (6.4.3.1.5), and protected areas and conservation (6.4.3.2.2). It also complements scientific knowledge by adding significantly to scientific information on husbandry techniques and habitat management for managed pollinators other than *Apis mellifera* (sections 5.3.4, 5.3.6 and 6.4.4.1.1, 6.4.4), such as adding artificial nests and food for pollinators (6.4.5.1.5), or related to social and behavioural responses (6.4.3.4 and 6.4.4.4).

6.4.6.2 Legal integrated responses

6.4.6.2.1 Large-scale land-use planning

There is an extensive literature regarding how an understanding of ecosystem services in general could be used to improve land-use planning (for example, Chan et al., 2011; Goldstein et al., 2012). There are a few examples where an understanding of ecosystem services has been used to influence land use planning outcomes, such as the often cited example of the New York City water management (Kremen and Ostfeld, 2005). We were unable to find an implemented example where pollination or pollinator protection has been one of the primary drivers in land-use planning. There are, however, a number of research projects that have used pollination as one of the key ecosystem services in analyses of the cost impact of different land-use change scenarios (Olschewski et al., 2006; Olschewski et al., 2010; Ricketts and Lonsdorf, 2013).

Land-use planning is more likely to build on an understanding of multiple overlapping benefits (and costs) associated with different land-use scenarios rather than a single ecosystem service, such as crop pollination. This approach is also more likely to detect economic advantages associated with habitat protection, because the sum of multiple benefits will be greater than that from any single service unless there are strong trade-offs between services (Olschewski et al., 2010) (see section 6.8 for a discussion of the evidence for specific trade-offs). Whereas some land-use analyses have applied a total valuation approach, decision making is generally guided by the marginal change in value associated with an action (i.e., the value added or lost for each small piece of land changed). Ricketts and Lonsdorf (2013) show that some patches of habitat have a much higher value under marginal valuation (i.e., assessing stepwise loss in cover) than they would in an average or total valuation across the whole landscape.

6.4.6.2.2 High-level initiatives, strategies and policies focused on pollinators

The North American Pollinator Protection Campaign (NAPPC; http://pollinator.org/nappc), was established in 1999. This initiative focuses on North America, including Canada, USA, and Mexico. It has members and 120 partner organizations from all three countries, and is co-ordinated by The Pollinator Partnership. The biggest achievements of the NAPPC so far have been the 2007 Status of Pollinators report (National Academy of Sciences, 2007), the production of 31 Web-based regional planting guides covering the entire US, to help farms, schools, parks and businesses grow pollinator-friendly landscapes, and the 11 major pollinator-protection agreements signed between the Pollinator Partnership and federal government agencies responsible for land management.

The International Pollinators Initiative, facilitated by the Food and Agriculture Association of the United Nations (FAO), was formally established by the Convention on Biological Diversity in 2000 (Convention on Biological Diversity, 2012), as part of a Programme of Work on Agricultural Biodiversity developed in 1996. Its aim was to coordinate action worldwide to: monitor pollinator decline; address the lack of taxonomic information on pollinators; assess the economic value of pollination; and promote the conservation and sustainable use of pollinator diversity. It has developed a number of useful tools and guidance, including a protocol for detecting and measuring pollination deficit in crops tested in at least eighteen countries (Vaissiere et al., 2011, see section 6.4.1.1.10)), a guide to help farmers evaluate the costs and benefits of applying pollinator-friendly practices (Grieg-Gran and Gemmill-Herren, 2012), and a spreadsheet-based tool for assessing pollination value and vulnerabilities to pollinator decline at national scale (Gallai and Vaissiere, 2009). The International Pollinators Initiative also maintains the Pollination Information Management System (see *Decision Support Tools* in section 6.4).

Several national or regional pollinator initiatives have been established under the umbrella of the FAO International Pollinators Initiative (http://www.fao.org/pollination/en/). One that preceeded it was these include the African Pollinator Initiative and the Brazilian Pollinators Initiative. The Brazilian Pollinators Initiative was started in 2000 by scientists. It became an official Government initiative in 2009, led by the Brazilian Ministry of the Environment, and established research networks focused on 11 valuable crops including cashew, Brazil nut and apple. These networks were funded by the Brazilian Research Council (CNPq; costing US $2 million in total) and supported by a range of international institutions (http://www.polinizadoresdobrasil.org.br/index.php/pt/). In 2010, the African Pollinator Initiative published a guide for the identification of tropical bee genera and subgenera of sub-Saharan Africa, in both English and French. This is available free to download at http://www.abctaxa.be/volumes/vol-7-bees, and hardcopies are freely available for people in developing countries. Between 2010 and 2014, 349 free copies of the book were distributed to people in 16 countries, including Cameroon, Ethiopia, Sri Lanka and Malaysia.

More recently, several countries have initiated strategic policy initiatives on pollinators at the national level. They include the Welsh Pollinator Action Plan, the National Pollinator Strategy for England, and the US National Pollinator Health Strategy.

There is no doubt that these integrated actions and strategies can lead to policy change with the potential to influence pollinator management on the ground. There are examples of both non-Governmental Pollinator Initiatives (the North American Pollinator Protection Campaign) and national pollinator strategies (The National Pollinator Strategy for England) leading to specific consideration of pollinators in agricultural policy. In the US, the NAPPC worked with other organisations to ensure that the 2008 Farm Bill included pollinator programs. In England, a new agri-environmental scheme being designed for the latest reform of the Common Agricultural Policy, to start in 2016, will include an optional package of measures targeted to pollinators, as a direct result of Government signing up to a National Pollinator Strategy for England. In both cases this was possible because action on pollinators was demanded at the appropriate time, during a development stage in the agricultural policy cycle (Dicks et al., 2015; see section 6.1 for explanation of the policy cycle).

6.4.6.3 Integrated knowledge responses

6.4.6.3.1 Changing behaviour through engagement and education

Education and outreach programs focused on pollinators and pollination have increased in recent years globally, in both school curricula and informal settings (museums, websites, conservation programs, entertainment media such as TV and radio). For example, in Mexico, scientific information on pollination and the role of bats is included in a fourth-grade text book issued by the Government to all 9-10 year old school children (Secretaría de Educación Pública, Mexico, 2014).

We found no published evidence of pollinator education programs leading to impacts on pollinator populations through behaviour change.

Environmental education (EE) research, drawing on the fields of environmental psychology and sociology, provides evidence of particular outreach and education strategies that result in behaviour changes in the audience. The early and persistent assumption that environmental knowledge leads to environmental attitudes, which then lead to pro-environmental behaviour, is no longer accepted (Kollmuss and Agyeman, 2002). Instead, numerous evidence-based theories involving meta-analyses of existing studies have identified variables associated with pro-environmental behaviour. Some of these variables are relevant to the specific behaviours necessary to enhance pollinator populations: knowledge of the issue and action strategies to address it, perception of one's own ability to affect change (internal locus of control), pro-environmental attitudes, verbal commitment to the behaviour, sense of personal responsibility for the environment, and social and institutional constraints to the desired behaviour. Key behaviour-change strategies that influence these variables can be drawn from standard techniques in social marketing (Monroe, 2003). They include: tailor the message and the types of information provided to the audience, including understanding barriers and benefits to the behaviours for that audience; use methods that create commitment to the behaviours, including providing vivid, meaningful procedural information about the action desired (Monroe, 2003).

Pollinators, unlike many targets of environmental education, allow the public to make a direct link between learning and specific behaviours. The two main strategies of pollinator education campaigns expected to be effective in producing behaviour change are: 1) *Building awareness and concern* about the declines in populations of some pollinator species and their role in food production; 2) *Practical training and real opportunities for action,* such as planting a garden or reducing pesticide use.

Many public programs around the world use these education strategies. Conservation organizations such as the Xerces Society (USA), Bumblebee Conservation Trust (UK), and the Pollinator Partnership (USA) offer conferences, workshops and/or training that specifically provide information and hands-on practice with pollinator habitat enhancement techniques, as well as online educational materials, for landowners, farmers, teachers and the broader public. University programs aimed at post-graduates and professionals in agriculture and environmental sciences provide courses on pollinator biology, management and conservation. For example, a two-week Pollination Course is provided by government, university and

NGO partners in Brazil. This has run every year since 2008 (every other year 2003-2008), and has intensively trained nearly 300 professionals (http://pollinationcourse.wix.com/2014english). Pollinator citizen science programs are numerous (see Citizen Science section) and in addition to producing monitoring data, are also effective education programs, engaging thousands of volunteers by providing information about the role of pollinators in ecosystems and food production, and providing an opportunity for action by monitoring the pollinators in their local area (Toomey and Domroese, 2013).

6.4.6.3.2 Research and monitoring

There are funding programmes dedicated to pollinators or pollination research in Australia, the UK, USA, Brazil, India, Kenya and South Africa. For example, between 2003 and 2009, the Brazilian Government invested US$ 3.3 million in development of management plans for native pollinators of plants of economic value, including West Indian cherry, guava, tomato, mango, passion fruit, cashews, Brazil nuts, melons, and cotton (http://www.cnpq.br/web/guest/chamadas-publicas;jsessionid=22C71C12E78764DEB8534068636DF7AC?p_p_id=resultadosportlet_WAR_resultadoscnpqportlet_INSTANCE_0ZaM&idDivulgacao=76&filtro=resultados&detalha=chamadaDetalhada&exibe=exibe&id=116-16-938&idResultado=116-16-938 and http://www.mma.gov.br/biodiversidade/projetos-sobre-a-biodiveridade/projeto-de-conserva%C3%A7%C3%A3o-e-utiliza%C3%A7%C3%A3o-sustent%C3%A1vel-da-diversidade-biol%C3%B3gica-brasileira-probio-i/processos-de-sele%C3%A7%C3%A3o-finalizados).

The Australian Honey Bee and Pollination Programme is a joint Government and industry program that invests over US$1 million a year in research on sustainable beekeeping and crop pollination. Analyses of its research investments showed that it provided positive returns, with benefit: cost ratios ranging from 2.05 to 28.61 (Rural Industries Research and Development Corporation, 2012). These numbers were based on economic, environmental and social benefits accrued, relative to a scenario without the research, for three case study projects. Potential societal benefits included the maintenance of rural livelihoods through beekeeping, and reduced impacts of chemical handling through biological control of chalkbrood. The AmericanHort Bee and Pollinator Stewardship Initiative http://americanhort.org/AmericanHort/Shop/Be_In_The_Know/AmericanHort/Knowledge_Center/beespoll.aspx is a similar collaborative funding scheme for the US horticulture industry. The UK Government, through its National Bee Unit (www.nationalbeeunit.com), and the US Department of Agriculture (www.ars.usda.gov/main/site_main.htm?modecode=80-42-05-40) dedicate research funding to honey bee health and monitoring. The USDA Colony Collapse Action Plan (http://www.ars.usda.gov/is/br/ccd/ccd_actionplan.pdf) directed $1 million USD per year from 2008-

2012, which contributed to understanding the causes of Colony Collapse Disorder, and the programme was continued in 2015 (USDA, 2013; USDA, 2015; see section 2.3).

The UK Insect Pollinators Initiative invested a total of £9.65 million in nine projects through a partnership of six research funders between 2009 and 2014. The research covered the health, ecology and conservation of both managed and wild pollinators, as well as crop pollination. It led to a number of important new findings, including spatial evidence for pathogen transfer between wild and managed bees (Furst et al., 2014), empirical evidence of negative interactive effects between pesticides (Gill et al. 2012), and maps of current and future pollination for the UK (Polce et al., 2013; Polce et al., 2014). The final outcomes and impact of this research effort are yet to be reported.

The European Commission has funded a series of international research projects focused at least partly on pollinators (ALARM http://www.alarmproject.net/, STEP http://www.step-project.net/) and more recently on pollination as an ecosystem service (LIBERATION http://www.fp7liberation.eu/TheLIBERATIONproject; QUESSA http://www.quessa.eu) or measuring farmland biodiversity (BIO-BIO http://www.biobio-indicator.org). Each cost several million euros. These projects either have generated, or are expected to generate, globally important findings and datasets. The ALARM project, completed in 2009, compiled the first detailed quantitative assessment of pollinator decline (Biesmeijer et al., 2006) and a Europe-wide climate change risk atlas for butterflies (Settele et al., 2008). The STEP project is continuing this work, with greater focus on mitigation. It has produced, for example, a meta-analysis on the effects of agri-environmental management for pollinators (Scheper et al. 2013) and new analyses of the pollinator decline data for Europe (Carvalheiro et al., 2013). The BioBio-project identified wild bees and bumble bees as one of 23 indicators for measuring farmland biodiversity (Herzog et al., 2013).

These examples demonstrate that dedicated funding for pollinator research is effective at delivering robust, peer-reviewed scientific evidence and societal benefits.

6.4.6.3.3 Centres of information, research and knowledge exchange

Knowledge exchange must take place alongside research to ensure that the research answers the right questions and has a chance to be incorporated into policy and practice quickly. See Chapter 5 (section 5.2.4.7) for a discussion on co-production of knowledge across different knowledge systems.

Cook et al. (2013) described four institutional frameworks to achieve effective knowledge exchange in conservation science – i) boundary organisations spanning science and management, ii) scientists embedded in management agencies, iii) formal links with decision-makers at research-focussed institutes and iv) training programmes for practitioners. At least three of these approaches can be identified in one or more of the many networks or centres for information and knowledge exchange on pollinators that have been established around the world. Prominent examples are shown in Table 6.4.6.2. All examples are providing information or resources to a broad set of target audiences, usually including researchers, beekeepers, farmers, policymakers and members of the public. The effectiveness of this activity is hard to quantify. Most of the centres have not actively reported performance indicators, or direct or indirect measures of their impact. Even so, some of the resources they have produced, even very recently, are widely used and well known.

Several international biodiversity information centres carry information on pollinators although their remit is far broader. For example, the International Union for the Conservation of Nature (IUCN; www.iucn.org) holds a number of conservation databases, including the Red List of threatened species, which has assessed the threat status of all European bee species (Nieto et al., 2014). The Global Biodiversity Information Facility (GBIF; http://www.gbif.org/) collates global biodiversity data for over 1.5 million species and has been used to investigate spatial patterns in plant-pollinator interactions, such as oil-collecting bees in the genus *Centris* and flowers that produce oil (Giannini et al., 2013). The Integrated Taxonomic Information System has a checklist of the world's bee species, providing details of all synonyms and subspecies (ITIS; http://www.itis.gov/beechecklist.html).

Ensuring transfer of indigenous and local knowledge, or biocultural traditions, from elders to new generations is a different challenge. In New Zealand, the Tuhoe Tuawhenua Trust (http://www.tuawhenua.biz/index.html) publish online videos of elders demonstrating traditional knowledge, such as methods for gathering honey, as if in conversation with younger people.

40. Table 6.4.6.2. Centres of pollinator-related information, research and knowledge exchange around the world.

Name	**Purpose**	**Location**	**Institutional framework**	**Website**
International Bee Research Association IBRA	Provides information and educational material on bee science and beekeeping worldwide.	UK	Boundary organisation	http://www.ibra.org.uk/

International Commission for Plant Pollinator Relationships (ICPPR)	Promotes and coordinates research on plant-pollinator interactions by organising meetings and networks.	International	Formal link between researchers and decision makers	http://www.uoguelph.ca/icpbr/index.html
Apimondia	The International Federation of Beekeeper's Associations. Organises international meetings for scientists, beekeepers, honey traders, regulators and development professionals.	Italy	Boundary organisation	http://www.apimondia.org/
COLOSS Network	A network of over 350 scientists from 64 countries. To coordinate research efforts and facilitate transfer of scientific information about honey bee health. It was initially funded as a European COST Action (COST FA0803).	Switzerland	Formal link between researchers and decision makers	http://www.coloss.org
SuperB	A new research network, SuperB (Sustainable	Netherlands	Formal link between researchers and decision makers	http://www.superb-project.eu/

	pollination in Europe) set up in 2014, also funded by COST (COST Action FA1307). Already has members from 30 countries.			
Centre for Pollination Studies in India	A Government-funded field research station focused on capacity building and making use of pollinator research (see case study box).	India	Formal link between researchers and decision makers Training for practitioners	http://cpscu.in/
Bee Health eXtension network	An online 'learning environment', linking research users directly with the American Land Grant Universities. Bee health is one of many resource areas.	USA	Formal link between researchers and decision makers Training for practitioners	http://www.extension.org/bee_health
Honey and Pollination Centre, University of California, Davis	Exchanging knowledge between pollination researchers and the wider community of research users.	USA	Formal link between researchers and decision makers	http://honey.ucdavis.edu

CASE STUDY: Farmers, researchers and Government working together in Tripura, India

As part of a Darwin Initiative project '*Enhancing the Relationship between People and Pollinators in Eastern India'* the Centre for Pollination Studies, based at University of Calcutta, established a field station for researchers in the north eastern state of Tripura (http://cpscu.in/). This was initially funded by the UK and Indian Governments and the University of Calcutta, with ongoing support from the local Government of Tripura. Local field staff joined the project to support researchers and facilitate engagement with farmers. In the first year a network of 15 long-term monitoring stations was established. Many farmers have been keen to engage by running long-term monitoring on their farms, sharing their local knowledge or taking part enthusiastically in training events. The project has run a series of well-attended farmer events, referred to as 'festivals' because they include a celebratory meal and some cultural events. At festivals, project staff provide training on pollinators and their role in agriculture. Local officials and prominent community members have increasingly lent their support, attending and speaking at these events. From the outset the Tripura State Department of Agriculture was very supportive, providing staff at no charge and helping to keep farmers informed. Recently a Memorandum of Understanding was signed between the Centre for Pollination Studies and the Tripura State Department of Biotechnology to mainstream the findings of the project research programme and to work together to engage and build capacity in local communities. The first jointly-run festival event attracted 150 people. The next joint venture will be to create exhibits in a public space.

6.4.6.3.4 Use of citizen science for pollinator research and monitoring

Long-term monitoring of pollinator populations, and pollination, is greatly needed all over the world (see Chapter 2). Appropriate methods and costs of a global monitoring scheme have been discussed (Lebuhn et al., 2013) and the UK Government is currently funding research to design a cost-effective pollinator monitoring programme for the UK, as part of the National Pollinator Strategy for England (Defra, 2014).

Citizen science projects to monitor pollinator populations have been established in many regions. We have gathered some prominent examples in Table 6.4.6.3.

As an indication of the scale of citizen science activity for pollinators, the Xerces Society (USA) provides a catalogue of 15 pollinator citizen-science projects in the US (http://www.xerces.org/citizen-science/pollinator-citizen-science/). A database of biodiversity monitoring projects across Europe collected by the EU MON project (http://eumon.ckff.si/index1.php; accessed 22 October 2014) lists 34 different butterfly, moth or wild bee monitoring schemes involving volunteers, in 18 different European countries. Most of these monitor butterflies (30 of the 34 schemes), ranging from single species (*Maculinea rebeli*) annual egg counts on a few sites by a single volunteer in Italy, to 2000 volunteers doing standardised weekly transect counts of 64 species at 1,200 sites in the UK.

Kremen et al. (2011) tested the quality of citizen-science data by comparing the results of flower visitor monitoring between trained citizens and professional insect ecologists. Overall coarse trends in pollinator abundance, richness and community structure matched between citizens and scientists. Citizens could reliably distinguish between native bees and honey bees (which are not native in the US), allowing them to provide important data on the overall abundance of wild bees, for example. Such data could potentially be used as proxies to track trends in pollination, or ecosystem health (Munoz-Erickson et al., 2007) as required by policy makers, although their correlations with actual pollination or measures of ecosystem resilience are untested. In Kremen et al.'s study, the citizens missed over half the groups of bees collected. The authors concluded that citizen science data collected by inexperienced members of the public could not reliably reflect patterns in occurrence of specific pollinator species or groups.

Some citizen science projects have generated globally important datasets. For example, data from long-running insect recording schemes in the UK, Belgium and the Netherlands are the basis of important analyses of pollinator trends in Europe (Biesmeijer et al., 2006; Carvalheiro et al., 2013). The data held by these insect recording schemes (see Table 6.4.6.3) are usually validated for obvious anomalies and verified by experts to check species identities. While there is often no information on sampling effort, and a possibility of bias towards attractive, unusual or easy to find species (Ward, 2014), statistical techniques have been developed to account for these issues (Morris, 2010; Hill, 2012; Carvalheiro et al., 2013).

National-level trends and spatial patterns are discernible from citizen-science data. Here we highlight a few studies to illustrate this. Deguines et al. (2012) found degraded insect flower-visitor communities in urban areas across France, relative to agricultural or natural areas, based on data from the SPIPOLL project. Hiromoto et al. (2013) are using a participatory monitoring project to gather information about the numbers of invading *Bombus terrestris* in Hokkaido, Japan. Stafford et al. (2010) showed that photographic records collected via popular social media sites could quickly generate records from across the UK, which could be used for species identification if clear instructions were given on important body parts to include in the photo. Trained members of the public in New South Wales, Australia monitored the extent of a small invading non-native bee species, *Halictus smaragdulus* (Ashcroft et al., 2012). Data from the North American Bird Phenology Program were used to show that ruby-throated hummingbirds (*Archilochus colubris*) are arriving 11-18 days earlier from their migration in the Eastern USA than in the early to mid-twentieth century (Courter et al., 2013). There are many other examples, covering pollinators in general, or specific to bees, moths or birds.

Where citizen science data have been systematically collected with standard methods, they can also enable scientists to begin to distinguish the relative importance of possible drivers of decline. For example, Bates et al. (2014) showed a negative effect of degree of urbanization on the diversity and abundance of moths in gardens, based on the citizen science Garden Moth Scheme in the UK (www.gardenmoths.org.uk).

41. Table 6.4.6.3. Global examples of citizen science projects that monitor pollinators.
This Table gives examples to illustrate the range of possibilities. It is not exhaustive (see text for indication of the number of pollinator monitoring schemes that involve volunteers).

Project name	Geographic scope	Number of participants	Brief description and reference
The Great Sunflower Project	US	Over 100,000 people signed up. Data submitted from 6,000 sites.	Volunteers count insects and birds visiting flowers in their back gardens, following a standard methodology. Data are used to map urban pollination. www.greatsunflower.org
Insect recording schemes. Example: Bees, Wasps and Ants Recording Society (BWARS)	Schemes in several countries, including the UK, Netherlands, Belgium.	BWARS (UK): About 50 regular recorders	Volunteer recorders, often highly skilled amateur entomologists, submit ad-hoc records of species, which are validated and verified by experts, and collated in national distribution maps. www.bwars.com
New Zealand Nature Watch Hymenoptera project	New Zealand	25 members in the first year. (Ward 2014)	Online community of volunteer recorders. Identifications are open to be validated and queried by others; anyone can be an expert. http://naturewatch.org.nz/
Seiyou status	Hokkaido, Japan	Over 140 participants in the years 2007-2011.	Participants monitor and destroy spring queens of the invasive bumblebee *Bombus terrestris*. Scheme running 2006-2014. (Horimoto et al. 2013). http://www.seiyoubusters.com/seiyou/en/
Social wasp and bumblebee monitoring in Poland	Poland	50 volunteers	Standard transect counts to monitor bumble bee and wasp community composition (50 species) at 40 agricultural or garden sites, every 20 years. Operating 1981 – 2020.
SPIPOLL (France)	France	1,137 volunteers	Following a standard protocol, volunteers photograph all insects visiting a flower of their choice over a 20 minute period. Pictures are identified online by volunteers (Deguines et al. 2012) www.spipoll.org.
Monarch Larva Monitoring Scheme	USA, Canada, Mexico	Over 1000 sites since inception in 1996, multiple volunteers per site	MLMP volunteers collect data on monarch egg and larval densities, habitat characteristics, and parasite infection rates. (Oberhauser and Prysby 2008). http://www.mlmp.org/

Iingcungcu Sunbird Restoration Project	City of Cape Town, South Africa	Eight schools	The aim is to relink broken migration routes for sunbirds across nectar-less urban areas by planting bird-pollinated plants on school grounds and involving learners in restoration and bird monitoring. http://academic.sun.ac.za/botzoo/iingcungcu/
Earthwatch: Butterflies and bees in the Indian Himalayas	Kullu Valley, Himachel Pradesh, India.	Three expeditions a year since 2012. So far 88 volunteers have taken part.	Volunteers monitor bees and butterflies visiting fruit crops at different elevations and the diversity of other flower resources. http://earthwatch.org/expeditions/butterflies-and-bees-in-the-indian-himalayas
People, Plants and Pollinators: Uniting Conservation and Sustainable Agriculture in Kenya	Kenya: Kerio Valley, Kakamega Forest, Taita Hills	> 50 farmers and >100 schoolchildren involved in direct monitoring	Volunteers document and monitor flower-visiting insects on specific crops and plants that of high value to the community and/or for pollinators. Over 1000 pollinator species documented on some farms.
“Guardiões da Chapada” Chapada Guadians	Brazil: Chapada Diamantina, Bahia	>50 tour guides and > 100 volunteers in 2015 (the first year)	Volunteers upload pictures of flower-visitor interactions to the project Webpage and/or identify the species. The information will be used to build a database on the distribution of plants and flower visitors in the Chapada Diamantina region. http://www.guardioesdachapada.ufba.br/ https://www.facebook.com/Guardi%C3%B5es-da-Chapada-486135114871905/timeline/

42. Table 6.4.6.1. Summary of evidence relating to policy, research and knowledge exchange across sectors.

Response (Chapter 6 section)	**Main driver(s) (Chapter 2)**	**Type**	**Status**	**Scientific evidence**
Large scale land use planning (6.4.6.2.1)	Land use change	Legal	Proposed	No specific evidence of use.
High-level initiatives, strategies and policies focused on pollinators (6.4.6.2.2)	All	Policy	Established	Some evidence of direct influence on policy, but not of actual impacts on biodiversity, food production or cultural value. (ESTABLISHED BUT INCOMPLETE)
Outreach and education	All	Knowledge	Established	Well-designed activities can change practices, although there is no evidence yet of direct effects on pollinators, or food production.

(6.4.6.3.1)				(ESTABLISHED BUT INCOMPLETE)
Fund scientific research on pollinators (6.4.6.3.2)	All	Knowledge	Established	Dedicated funding delivers high-quality scientific outputs (WELL ESTABLISHED) and societal benefits (ESTABLISHED BUT INCOMPLETE).
Knowledge exchange between researchers or knowledge holders and stakeholders (6.4.6.3.3)	All	Knowledge	Established	Many examples around the world. Effectiveness for pollinators and pollination unknown. (INCONCLUSIVE)
Employ citizen science for pollinator monitoring (6.4.6.3.4)	All	Knowledge	Established	Can discern trends and spatial patterns for some pollinator species or groups. (WELL ESTABLISHED) No specific evidence of use.

6.5 Experience of tools and methodologies for assessing responses

This section describes the available tools and methods for mapping, modelling and analysing options for action on pollinators and pollination, and reviews experience of their use.

6.5.1 Summary of tools, methods and approaches

Many of these tools and methods aim to incorporate existing knowledge and stakeholder or policy preferences into environmental decisions. Often, they can be applied in conjunction with one another. For example, models can be used to build maps that are used in participatory assessments or decision support tools. Evidence synthesis can be used to identify best practice, to define parameters in models or to quantify performance criteria for multi-criteria analysis. Some, not all, of these tools employ economic valuation methods discussed in Chapter 4 (Sections 4.4, 4.5 and 4.6).

6.5.1.1 Case study/best practice approach

Case studies are often used to exchange knowledge and experience, or communicate best practice. An advantage of case studies is that they can be a quick, low-resource option providing localised guidance. For example, the International Pollinators Initiative has collected online written case studies, including reports on pollination requirements of particular crops, monitoring methods and data recording sheets (www.internationalpollinatorsinitiative.org).

The FAO published an initial survey of best pollination practices for at least eight crops in Africa, Asia, North America and South America (FAO, 2008), including mango, papaya and cardamom. This resource is currently being updated. Costs and benefits of the practices are described, but not quantified.

The Pollinator Partnership in the US has published a set of Best Management Practices for four US crops: almond, apple, melon and corn (Wojcik et al., 2014). 'Best' practices were identified by reviewing scientific literature, printed and online resources available to growers and interviews with farm advisers and producers. Some identified best practices were commonly promoted across the industry, such as night spraying and providing outreach material to growers. Others were not mentioned or missing from practice. For example, 'pesticide label instructions in Spanish' was identified as a best practice, but missing from industry practice for all four crops.

Strictly, best practices should be identified by benchmarking, based on outcome metrics that compare practices carried out in a similar context, to find out which perform best. We do not know any examples of this involving pollinators or pollination.

6.5.1.2 Evidence synthesis

Systematic, hierarchical synthesis of evidence is the basis of evidence-informed policy and practice (Dicks et al., 2014a). For pollinators and pollination, a number of systematic reviews, meta-analyses and systematic maps have analysed relevant evidence (Humbert et al., 2012; Randall and James, 2012; Scheper et al., 2013).

In 2010, global evidence on the effects of interventions to conserve wild bees (all species) was summarised in a collated synopsis, covering 59 different responses to a range of threats, with 162 scientific studies individually summarised (Dicks et al., 2010). These summaries are available in an open-access online resource (www.conservationevidence.com). The synopsis has been used for reference in developing the National Pollinator Strategy for England (Defra, 2014) and the FAO International Pollinators Initiative (Convention on Biological Diversity, 2012).

This resource needs updating to cover all pollinators, pollination and evidence from 2011 onwards. The approach has been applied to other ecosystem services, such as pest regulation and soil-related services (www.conservationevidence.com).

The evidence in the bee conservation synopsis was scored for certainty by a group of experts (Sutherland et al., 2011) and their scores used to identify research priorities considered important by conservationists but with little scientific certainty about effects. Research priorities included investigating effects on wild bees of restoring species-rich grassland, and increasing the diversity of nectar and pollen plants at landscape scale. A similar assessment of summarised evidence on interventions to enhance farmland biodiversity (Dicks et al., 2014c) recommended one action specific to pollinators – planting nectar flower mixtures – on the basis of existing evidence.

We know of no examples where this unbiased synthesis of evidence has been employed in decision-support systems relevant to pollinators or pollination (see *Decision support tools* below).

Scanning for alternative options, or solutions, is an important element of organising synthesized evidence to link it with decision-making approaches (such as *Multi-criteria analysis* below). Thirty-one management actions for enhancing biodiversity-mediated pollination were listed by Sutherland et al. (2014), and incorporated in the list of responses developed for this report.

6.5.1.3 Risk assessment

Risk assessment is a way of quantifying the likelihood of specific threats or hazards, and is used to help decide whether mitigation is needed. Risk assessment uses a well-established and constantly developing set of methods, and is widely used to support decision making in policy and business. For pollination and pollinators, risk assessment is most widely used in the context of predicting the risk from pesticides and GMOs. It is discussed as a Technical response in section 6.4.2.

The Causal Analysis/Diagnosis Decision Information System (CADDIS; http://www.epa.gov/caddis/) is a formal approach to elicit and organize expert opinions on risk factors, designed by the US Environmental Protection Agency for environmental problems where multiple causes are suspected. It was used to identify '*Varroa* mites plus viruses' as the probable cause of reduced survival in honey bee *Apis mellifera* colonies in California almonds orchards (Staveley et al., 2014).

6.5.1.4 Multi-criteria analysis

Multi-criteria Analysis (MCA; also called multi-criteria decision analysis MCDA, multi-criteria decision-making – MCDM, or multi-criteria evaluation – MCE) is an approach to decision-making that evaluates multiple objectives against multiple attributes or performance criteria (see section 4.2.7.5). MCA is designed to take account of trade-offs. It often involves participatory engagement with stakeholders (42% of examples included stakeholders in a recent review by Estevez *et al.* 2013) and was strongly advocated over purely economic valuation for making decisions about ecosystem services (Spangenberg and Settele, 2010). It has very frequently been applied to environmental decision domains such as land-use planning, biodiversity conservation, water resource management, and energy systems, and a range of methods and approaches are well developed (see Moffett and Sarkar, 2006; Hajkowicz and Collins, 2007; Huang et al., 2011; Estevez et al., 2013).

Multi-criteria evaluation was used to derive a map of suitability for honey bee hives in La Union Island, the Philippines (Estoque and Murayama, 2011). Criteria for good hive placement were suggested and weighted by experts. The results showed high correlation between the landscape suitability index and real honey yields. We could find no cases where pollination was explicitly considered as part of a Multi-Criteria Analysis.

A broader approach advocated for environmental decisions is called Structured Decision Making (SDM) (Gregory et al., 2012). This expands on Multi-Criteria Analysis with more focussed effort and guidance on defining the initial objectives and performance measures with stakeholders, as well as monitoring and review stages to incorporate learning into the ongoing decisions. SDM practitioners employ various Multi-Criteria analysis tools, when formal quantitative analysis of trade-offs is required to make a decision.

6.5.1.5 Cost-benefit analysis

Cost-benefit and cost-effectiveness analyses (section 4.1.1.4) have both been used to address decisions about pollinators (Morandin and Winston, 2006; Olschewski et al., 2007; Breeze et al., 2014a). A range of valuation methods can be employed (see Table 4.2).

Marginal Abatement Cost (MAC) curves are a popular tool to illustrate cost-effectiveness information. They show the cost associated with the last unit (marginal cost) for varying amounts of reduction in something bad for the environment (such as greenhouse gas emissions), or supply of an environmental good (such as clean water or pollination). They are used to select a cost-effective set of responses to an environmental problem and have mostly been employed to inform climate change mitigation policy (Kesicki and Strachan, 2011). MAC curves have not yet been employed to inform decisions on actions to enhance pollination, or other ecosystem services, because the analysis required to do so it still at an early stage. Ricketts and Lonsdorf (2013) estimated marginal losses of pollination value from removal of forest patches in a Costa Rican landscape, and showed that the marginal pollination value of a hectare of forest is highest when the density of surrounding forest cover is low. To develop a MAC curve, this marginal value information would be combined with the cost associated with keeping each hectare of forest, the amount of forest available to keep, and then compared to similar marginal pollination values generated by other responses, such as retaining or restoring other habitat types.

6.5.1.6 Environmental Impact Assessment

Environmental Impact Assessment (EIA) is a set of well-defined methods for evaluating the environmental impact of particular projects or activities. In Europe, regulation requires that EIAs be carried out on all projects involving certain defined process. When applied to policies, it is called Strategic Environment Assessment.

We found no examples of EIA taking explicit account of pollinators or pollination. A review of Environment Impact Assessment methods applied to the fruit sector doesn't mention pollination (Cerutti et al., 2011). Crist et al. (2013) describe a process for assessing the likely impacts of a development on regional ecosystem services, which focuses on the process of consultation and decision-making around major projects. The guidance does not mention pollination as a possible service.

6.5.1.7 Vulnerability assessment

Vulnerability Assessment, or vulnerability analysis, describes an analytical exercise in which the goal is to identify areas, sectors or groups of people particularly vulnerable to adverse effects of environmental change (see definition of vulnerability in Chapter 4, Section 4.6.1). It might be thought of as a broader, generic form of risk assessment. Several different approaches and frameworks have been used. Indicator-Based Vulnerability Assessment (IBVA) is a widely used method that combines quantitative and qualitative indicators, and has been used to inform climate change adaptation in the contexts of public health and water management (Tonmoy et al., 2014). These authors warn that methodological problems such as inappropriate scales and aggregation methods are frequent.

Given the emerging ability to identify areas of potential pollination deficit, vulnerability analysis could be a useful tool for policy on pollinators and pollination. A spreadsheet-based tool developed by the Food and Agriculture Organisation of the United Nations (Gallai and Vaissiere, 2009) allows a simple economic vulnerability assessment for a national economy (see *Decision support tools*, 6.5.1.12).

6.5.1.8 Environmental accounting

In environmental accounting, pollinators can be considered as a natural capital asset, and pollination as an input to production (see Chapter 4, Section 4.4). The recently developed System of Environmental-Economic-Accounting (European Commission et al., 2012) accounts for 'environmental goods and services', which are flows of products within the economy, rather than flows of services from the environment to the economy. This system is designed to accord with the established System of National

Accounts (an international statistical standard for compiling national accounts). It treats pollination as an input to the growth of a mature crop, flowing in fixed proportion to the quantities of harvested product, therefore assuming that the production function is stable (European Commission et al., 2013). The level of pollination can be accounted for as a function of the abundance of pollinators.

We found no example of pollination actually being accounted for in a national accounting framework, but steps have been taken towards doing so. For example, Dickie et al. (2014) assessed which characteristics of pollination need to be understood to allow its appraisal as a natural capital asset in national accounts. They identified a need to monitor common wild pollinators for ongoing trends, given the option value (possible future value) provided by diversity in the stock of wild pollinators.

Bateman et al. (2013) outline a different approach to taking account of ecosystem service values in national decision-making, based on welfare changes as a consequence of specific scenarios. These authors did not illustrate their approach with pollination as an example.

6.5.1.9 Mapping pollination

Most maps of ecosystem services so far produced do not consider pollination as a service, focusing instead on services with clearer links to spatial data such as land use on a regional or larger scale, such as recreation, or primary production. For example, in a 2012 review, Martinez-Harms and Balvanera (2012) identified just five studies that had mapped pollination at that time, from a total of 41 studies mapping ecosystem services.

A blueprint for mapping and modelling ecosystem services published by the thematic working group on mapping ecosystem services of the Ecosystem Services Partnership (ESP) in 2013 (Crossman et al., 2013) suggests pollination is not often mapped because it is delivered at small scale. Table 6.5.1 summarises all the published maps of pollination that we identified based on our searches (see *Methods* section). It serves to illustrate the range of methods that have been used. Where pollinators themselves (estimates or probability of abundance, for example) have been used to derive maps, only bees have been considered. We know of no pollination maps that take account of other (non-bee) pollinators.

As demonstrated by Table 6.5.1, all the currently available maps of pollination are based on relative measures or proxies of the pollination and most lack empirical validation. Whilst these studies represent good steps along the way to developing a validated tool for mapping pollination services, most overplay their utility, in the way they are presented in the primary literature. Using these maps as tools for decision-making poses serious problems if they are not accurate.

Eigenbrod et al. (2010) warned against the use of secondary proxy data, demonstrating that such maps provided a poor fit to primary data for three services – biodiversity, recreation and carbon storage. The estimates of bee abundance in the InVEST pollination module have been validated against empirical field data for some sites (see section on *Modelling* below), but the relationship between bee abundance and pollination is not straightforward (see Aizen et al. (2014), for an example where over-abundant bees reduced fruit set in raspberries).

Most maps of pollination supply or demand have not been validated against empirical (primary) data. Only two of the seventeen pollination maps in Table 6.5.1 have been validated. Some of the proxy measures used are very indirect, such as land cover variables. The 'supply' of pollination services map in Figure 2, for example, does not really show the pollination, but the distribution of habitat types such as grassland and forest edge assumed to support wild bees (Schulp et al., 2014). This map implicitly assumes that habitat is the only driver of wild bee abundance (see Chapter 2 for discussion of other possible drivers), and that wild bees are the only pollinators.

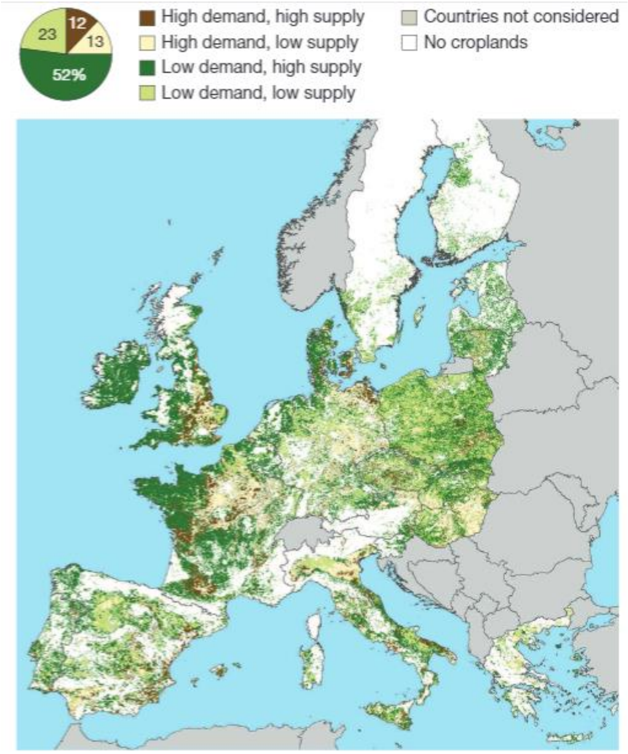
43. Table 6.5.1. Maps of pollination according to the methods used.
The validation column shows whether the maps were validated with empirical data from mapped landscapes. Scale categories are as defined in Chapter 4, with maps encompassing the whole of Europe classed as 'Global'. References marked * mapped other ecosystem services as well as pollination. The Lonsdorf index and InVEST model are described in section 6.5.10.

Method to map ecosystem services	Proxy data used to represent or derive pollination estimates	Validation	Scale	Study area	Reference
Index of bee abundance based on the availability of nest sites and floral resources (from land cover data) and bee flight ranges (Lonsdorf index).	Land cover	No	Regional	The Baiyangdian watershed. China	Bai et al. (2011)*
	Land cover	No	Global. 25x25m pixel size.	Europe	Maes et al. (2012)*
	Land cover	Yes	Regional. 30x30m pixel size.	California, Costa Rica and New Jersey	Lonsdorf et al. (2009)
Pollination service value, estimated using an index of pollination service based on proportion of pollinator habitat, and quantity and pollination dependence of crops grown in each pixel.	Land cover Crop areas	No	Regional. 30x30m pixel size.	California	Chaplin-Kramer et al. (2011)

Method to map ecosystem services	Proxy data used to represent or derive pollination estimates	Validation	Scale	Study area	Reference
Functional diversity of wild bees.	Bee distribution data (presence/absence)	No	National 10x10km pixel size.	Great Britain	Woodcock et al. (2014)*
Probability of presence for ten pollinating bee species (from species distribution models) for field bean *Vicia faba*.	Bee distribution data (presence/absence)	No	National. 1x1km pixel size.	Great Britain	Polce et al. (2013)
Changes to expected crop yield based on index of bee abundance (InVEST model) per hectare of deforested land.	Land cover Crop areas	Yes	Regional. 30x30m pixel size.	Costa Rica	Ricketts and Lonsdorf (2013)
Economic value of crops weighted by the value of animal-pollinated crops and total agricultural area.	Land cover Crop areas	No	Regional. 500 ha pixel size.	Central Coast ecoregion of California. USA	Chan et al. (2006)*
Area of pollinator-dependent crops, potential wild bee habitat and the visitation probability based on distance from nesting habitats.	Land cover Crop areas	No	Regional. 10x10m pixel size.	Leipzig, Germany	Lautenbach et al. (2011)*
	Land cover Crop areas	No	Global. 1x1km pixel size.	Europe	Schulp et al. (2014)
Modeling onset of flowering plants with explanatory variables (soil, climate and land-use data).	Soil, climate, land cover	No	Regional. 20x20 m pixel size.	Central French Alps, France	Lavorel et al. (2011)*
Percentage fruit set based on the distance of crops to forest.	Forest cover	No	Regional. 250m pixel size.	Central Sulawesi, Indonesia	Priess et al. (2007)*
Model exponential decline in pollination (pollinator species richness) as a function of distance from nearest natural habitat.	Land cover	No	Global	Global	Ricketts et al. (2008)
Model spatial relationship between the diversity of nectar-providing plants and explanatory variables (soil, climate and land-use data).	Soil, climate, land cover	No	National. 1x1km pixel size.	Temperate ecosystems of Great Britain	Maskell et al. (2013)*
Crop yield per area considering crops depending on pollination.	Crop yield	No	Global. 10x10km pixel size.	Global	Lautenbach et al. (2011)
Number of honey bee colonies divided by the total number of colonies demanded.	Honey bee colony numbers Crop areas	No	Global	Europe	Breeze et al. (2014b)
Landscape suitability for bees based on the quantification of desired land cover types (grasslands) within foraging distance from potential nesting sites.	Land cover	No	National. 100x100m pixel size.	North Dakota, USA	Gallant et al. (2014)

6.5.1.9.1 Indicators of pollination, as a basis for mapping

One approach to mapping ecosystem services is to define indicators of service status that can be estimated spatially. Layke et al. (2012) evaluated ecosystem service indicators from over 20 ecosystem assessments at multiple scales and many countries. They did not find any indicators for pollination, and considered that "regulating or cultural services such as pollination [and others] …. were not assessed by enough … assessments to draw or permit an analysis of indicators" (Layke et al., 2012). A 2011 report on ecosystem service indicators published by the Convention on Biological Diversity (CBD) Secretariat proposes three possible indicators of pollination that could be mapped (UNEP-WCMC, 2011) – percentage of planted crop area dependent on (wild) pollinators, status of pollinating species and landscape configuration, and suitability for pollinators. It does not include evidence that these have been used, either for mapping or any purpose, for actual policy decision or in sub-global ecosystem assessments. As pointed out above, all three indicators suggested by the CBD rely on secondary proxies that have never (crop areas; status of pollinating species), or seldom (landscape configuration) been validated against empirical data to check whether they reliably represent pollination delivery.



65. Figure 2. Estimated pollination supply and demand for Europe.

WARNING: this map, and others like it, use proxy measures of the *potential* for landscapes to generate pollination. Such measures are unvalidated, and may not reflect real pollination supply. Source: Schulp et al. (2014).

Maskell et al. (2013) used the number of species of nectar-rich plants preferred by bees and butterflies from a UK Countryside Survey dataset as indicators of pollination. A decision-support tool developed by a partnership of agricultural co-operatives in France (see section on *Decision support tools* below: 6.5.1.12) has also used pollinator forage plants as a proxy for pollination.

6.5.1.10 Modelling pollinators and pollination

For this report, modelling is the process of making an abstract, usually mathematical, representation of an ecosystem or socioeconomic system, in order to understand and predict the behaviour and functioning of the modelled system.

6.5.1.10.1 Spatially explicit models of pollinators and pollination, as a basis for mapping

A range of quantitative, spatially-explicit modelling approaches have been used to quantify and map the supply or demand of pollination (Table 6.5.1). The most widely used is part of The Integrated Valuation of Ecosystem Services and Trade-offs (InVEST) suite of models (Sharp et al. 2015).

The InVEST pollination module uses modelled estimates of wild bee abundance as a proxy for the supply of pollination. It employs the ‘Lonsdorf model’, in which different land use or cover types are assessed, using expert judgement, for their nesting and forage potential for wild bees (Lonsdorf et al., 2009). Each land cover type is mapped and a wild bee abundance index (the Lonsdorf Index) derived for every pixel, based on the foraging and nesting potential of the surrounding cells and the foraging ranges of the local bee species. The model must be implemented at scales within the foraging ranges of individual bees. Pixels of 30 x 30 m have been used in the cases where the model has been validated with empirical wild bee abundance data (Lonsdorf et al., 2009; Kennedy et al., 2013). A value of the pollination supplied to agriculture from each pixel is calculated as the economic impact of pollinators on crops grown in pixels within the relevant foraging ranges of each pixel in the pollinator source map, using dependence ratios and a simple saturating crop yield function, which assumes that yield increases as pollinator visitation increases, but with diminishing returns (see Chapter 4 for more on production functions). This model is well documented here: http://ncp-dev.stanford.edu/~dataportal/invest-releases/documentation/3_0_0/croppollination.html. The model provides relative, not absolute, abundance estimates and economic values, but these can be calibrated with real data on bee abundance data and effects on crop yield.

Other well-documented modelling platforms for spatially-explicit assessment of ecosystem service trade-offs (at least 15 identified by Bagstad et al. (2013)) have not yet incorporated alternative pollination modules, although some use the InVEST pollination module (see *Decision support tools*). This would be a valuable development, as some of the other modelling platforms place more emphasis on non-economic values and different groups of beneficiaries. For example, the Artificial Intelligence for Ecosystem Services modelling framework (ARIES; http://www.ariesonline.org) maps ecosystem service flows with an emphasis on the beneficiaries of each service. Pollination is suggested as a service suitable for ARIES modelling (Villa et al., 2014), but to our knowledge this has not been developed.

Spatially-explicit modelling of bee nesting and foraging resources in agricultural landscapes was used by Rands and Whitney (2011) to show that increasing the width of field margins would provide more food resources to wild bees whatever their foraging range.

6.5.1.10.2 Other modelling techniques

Various modelling techniques have been used to predict effects of future land-use change and climate change and on pollinators or pollination demand (see sections 2.1.1 and 2.5.2.3 respectively). These could provide information to inform crop management or conservation decisions, but we know of no specific examples where they have. For example, Giannini et al. (2013) showed a substantial reduction and northward shift in the areas suitable for passion fruit pollinators in mid-Western Brazil by 2050. This information could be used by the passion fruit industry to target conservation effort for these pollinators and their food plants, although there is no evidence it has been used for this purpose.

Population dynamic models have been built for honey bees (for example, DeGrandi Hoffman et al., 1989). An integrated model of honey bee colony dynamics that includes interactions with external influences such as landscape-scale forage provision has recently been developed (Becher et al., 2014), which accurately generates results of previous honey bee experiments. Bryden et al. (2013) used a dynamic bumble bee colony model to demonstrate multiple possible outcomes (success or failure) in response to sublethal stress from exposure to neonicotinoids, while a spatially-explicit model of individual solitary bee foraging behaviour has recently been developed (Everaars and Dormann, 2015). All these models have great potential to be used for testing effects on bees of different mitigation options, such as enhancing floral resources in the landscape, or reducing pesticide exposures.

A stochastic economic model was employed to quantify the potential cost of *Varroa* mites arriving in Australia, in terms of lost crop yields to due reduced pollination (Cook et al., 2007). This model has been used as a guide to how much the Government should spend trying to delay the arrival of *Varroa* (Commonwealth of Australia, 2011).

6.5.1.11 Participatory integrated assessment and scenario building

Participatory Integrated Assessment involves a range of stakeholders in scenario building or use of models to consider and decide on complex environmental problems. Its techniques have been extensively used in climate-change policy development at local and regional levels (Salter et al., 2010) and are sometimes used to develop scenarios for multi-criteria analysis. The underlying assumption is that participation improves the assessment, and the final decision. Salter et al. (2010) provide a review of methods and issues.

Future scenarios were built using a deliberative approach by the Millennium Ecosystem Assessment and UK National Ecosystem Assessment (Haines-Young et al., 2011). Those from the UK NEA were used to develop pollination futures to 2025 in a recent assessment of evidence for the UK Government (Vanbergen et al., 2014).

6.5.1.12 Decision support tools

Decision support tools are increasingly being used in environmental management to help decision-making (Laniak et al., 2013). They are distinct from the analytical mapping and modelling tools discussed above because they are designed around a particular decision or decision-making context, and ideally developed collaboratively with end-users. Most decision support tools are software based, and assist with decisions by illustrating possible outcomes visually or numerically, or leading users through logical decision steps (see section 4.6.3 for an example of stepwise decision trees). Some rely on complex models, only operable by their developers (see *Modelling pollinators and pollination*). Others have simple interfaces designed to be used by non-experts. Costs are variable, but can be relatively high (Dicks et al., 2014a).

A variety of decision support tools have emerged for systematic assessment of ecosystem services, in order to examine trade-offs and assist policy decisions. Bagstad et al. (2013) identified 17 different tools, ranging from detailed modelling and mapping tools (including InVEST, discussed in *Models for mapping the pollination* above) to low-cost qualitative screening tools developed for business, such as the Ecosystem Services Review (Hanson et al., 2012), and others have been developed since then. Many include carbon storage, sediment deposition, water supply and the scenic beauty of landscapes, among other services. Only a few such tools currently include pollination (for example, InVEST, Envision [using the InVEST pollination module (Guzy et al., 2008)] Ecometrix and the Ecosystem Services Review).

The Ecosystem Services Review includes pollination as one of a list of 31 possible goods and services, and business dependence on pollination is assessed qualitatively by stakeholders. Sandhu et al. (2012) developed this further into a risk analysis tool for three land-based businesses, but the case studies did not include a company with dependence on any pollination.

A great range of decision support tools can be applied in agriculture, agroforestry, pollinator management and land management. For example, the Danish decision support tool Crop Protection Online, sold commercially, presents users with relative risk quotients for bees and other beneficial insects, to help them choose crop protection products according to their toxicity (Gyldenkaerne and Secher, 1996). At least one commercial decision support tool in development uses field-scale estimates of pollinator food sources to generate advice on honey bee management for commercial farms (pers. comm., Jeremy Macklin, Hutchinson's Ltd, UK).

A spreadsheet-based tool developed by the Food and Agriculture Organization of the United Nations (Gallai and Vaissiere, 2009) has been used to assess the vulnerability of several countries to pollinator decline, based on the proportion of GDP dependent on pollination. This highlighted, for example, a dependence of over 7% of Ghana's GDP on pollinators, as a result of the high value and high dependence of cocao (Convention on Biological Diversity, 2012).

6.5.1.12.1 Accessible data sources

There are at least three online sources of data specific to pollinators and pollination that could be used for decision support tools, mapping, modelling and accounting. The Pollinator Information Network of the Americas (http://pollinator.org/PINA.htm) provides digitized pollinator records, contacts, and other plant-pollinator interaction datasets from across the Americas. Other more general sources of biodiversity data are discussed in the integrated responses section, under *Centres of information, research and knowledge exchange* (6.4.6.3.3).

The Pollination Information Management System managed by the FAO is an online database of pollination studies and basic crop dependence information based on Klein et al. (2007) (http://www.internationalpollinatorsinitiative.org/pims.do). The crop dependence information requires updating to take account of developments in the literature since 2007. For example, its entry on papaya does not identify the importance of hawkmoths (Sphingidae), demonstrated to be the primary pollinators of papaya in Kenya (Martins and Johnson, 2009).

Finally, there are accessible databases of toxicology information for specific pesticides. For example, the US Environmental Protection Agency maintains a database of ecotoxicology information (http://cfpub.epa.gov/ecotox/).

6.5.1.13 Ecosystem Approach

An 'Ecosystem Approach' is the primary framework for action under the Convention on Biological Diversity. It is defined as "the integrated management of land, water and living resources to promote conservation and sustainable use", with a priority to maintain ecosystem services (COP 5, Decision V/6 http://www.cbd.int/decision/cop/default.shtml?id=7148). In practice, this means taking account of the stocks and flows of ecosystem services, including pollination. Potschin & Haines-Young (2013) classify three major ecosystem assessment frameworks – habitat-based, system- or process-based, and place-based. The pollination examples they use fall into systems- or process-based (using the InVEST model to map supply and value of pollination, for example). They argue that all ecosystem assessments could be place-based at some scale, overlain with habitat, system- or process-based assessments.

6.5.2 Building an effective toolkit

Table 6.5.2 summarises the global experience of use of all these tools and methods for assessing responses and making decisions about pollinators and pollination. In general, we see that while many tools are available or in the process of being developed, only some have been used, and very few incorporated into real decisions in policy or practice. There is great potential to enhance the consideration of pollinators and pollination in environmental decisions through increased use of these tools.

The following tools and methods are well developed and appropriate for application to policy decisions about pollinators and pollination: evidence synthesis, environmental accounting, modelling, multi-criteria analysis and participatory integrated assessments.

For other tools, methods relevant to pollinators and pollination are not yet well developed enough for immediate application to decisions, but there is strong potential: identifying best practice, risk assessment, vulnerability assessment, mapping pollination, and decision support tools.

Enhancing the consideration of pollinators and pollination in policy requires **engaging** and **communicating** with people from all relevant sectors, so they understand the importance and value of pollinators to them (Cowling et al., 2008; Maes et al., 2013). It also requires **designing** and **resourcing** appropriate responses at appropriate scales. The tools discussed here can enable these different elements of mainstreaming pollination in policy, as shown in Table 6.5.3.

The literature on environmental decision support systems is informative on how to increase the use of particular tools and methods (McIntosh et al., 2011). The importance of involving end users in design and implementation is repeatedly emphasized, and the development of agricultural DSSs has tended to shift towards participatory approaches to both design and implementation (Jakku and Thorburn, 2010; Valls-Donderis et al., 2013).

44**. Table 6.5.2. Comparison of tools and methods.**

	Purpose	**Use for pollinators**	**Strengths**	**Weaknesses**
Case study/best practice approach	To exchange knowledge and guide practice.	Many organisations share case studies online. Best pollinator management practices identified for some crops.	Relatively quick. Relatively cheap. Easily understood. Can be locally relevant.	Performance metrics for identifying best practice not quantified.
Evidence synthesis	To inform decisions with the best available evidence.	Systematic reviews and synopses of evidence have informed decision-making on wild bees and agricultural interventions.	Systematic, explicit review and meta-analysis methods are well established. High confidence in conclusions. Demonstrates knowledge gaps.	Relatively expensive (Dicks et al. 2014). Interpretation in decisions requires judgement. Evidence may not be relevant locally.
Risk assessment	To identify and prioritise risks of a product or activity.	Established on several continents for pesticide regulation. Has led to restrictions of chemicals identified as a risk to the environment. Some evidence that it reduces overall environmental toxicity of pesticide use in agriculture over time.	Well established in many countries. Relatively quick and cheap if relevant data are available. Can be done at a range of scales.	Established methods only consider direct toxicity to honey bees and/or aquatic invertebrates. Rigorous methods specific to non-*Apis* pollinators, and sublethal effects, still under development. Relevant data are not always available.
Multi-criteria analysis	To evaluate multiple objectives against multiple attributes or performance criteria.	Very little used for decisions about pollinators. Could be used to address trade-offs between pollination and other services.	Effective at addressing trade-offs. A range of methods well developed. Involves stakeholders. Can be locally relevant.	Can be time-consuming.
Cost-Benefit Analysis	To compare the costs and benefits of different responses, and provide a single	A few simple examples have compared actions to benefit pollinators.	Compares costs and benefits. Can account for non-use values. Relatively quick and cheap if relevant data are available.	Standard methods to calculate costs and benefits not established for pollinators. Data on costs of alternative

	Purpose	Use for pollinators	Strengths	Weaknesses
	indicator of net benefit.			responses usually not available. Discount rates used to actualize future cost and benefit flows are a source of controversy.
Environmental Impact Assessment	To evaluate impacts of a project or activity.	None found.	Methods well established. Always locally relevant.	Only applies to specific projects.
Vulnerability Assessment	To identify areas, sectors or groups vulnerable to adverse effects of environmental change.	None found. Could be used to identify areas with pollination deficit.	Can be done at regional, national and global scales. Takes economic and ecological information into account.	Varied methods, not well developed and often mis-used.
Environmental Accounting	To monitor stocks and flows of environmental goods and services.	Pollination not included in ‘environmental footprint’ calculations, but included in international Environmental-Accounting Guidance. No experience of use yet.	Potential for high impact, by incorporating pollination into national accounts.	Recommended accounting method depends on a static production function uniform across crop varieties, extrapolated from empirical evidence. Requires a lot of data.
Mapping pollination	To visualise pollination supply and/or demand for a specific area, or set of conditions.	Many maps of pollination drawn around the world. A range of methods used. None incorporated directly into policy or practice decisions yet.	Estimates of wild bee abundance underlying one method (the Lonsdorf model, used in InVEST) have been validated empirically. Most useful on a regional scale (several farms or a landscape)	No validated measures of actual pollination. Validated measures are data intensive and time-consuming.

	Purpose	**Use for pollinators**	**Strengths**	**Weaknesses**
Modelling	To quantify and/or visualise the possible behaviour of environmental systems in response to sets of conditions or variables.	Various approaches to modelling pollinators and pollination supply demonstrated, including future effects of environmental change. Global scale models not yet developed. None incorporated directly into policy or practice decisions yet.	Most modelling approaches for pollinators and pollination are validated, tested for sensitivity and explicit about sources of uncertainty.	Methods are complex, with many assumptions that must be understood by users. Usually expensive.
Participatory Integrated Assessment and scenario building	For experts and stakeholders to consider and decide on complex environmental problems.	Some pollinator scenarios developed in the UK.	Enables alternative futures to be considered. Involves stakeholders. Can be done at a range of scales.	Based largely on judgement. Appropriate methods of consultation must be documented.
Decision support tools	To assist with decisions by illustrating possible outcomes, or leading users through logical decision steps.	Few decision support tools assessing ecosystem services or supporting land management decisions have incorporated pollination so far. Two examples of these being incorporated directly into policy or practice decisions.	Tools may refer to empirical data sets, such as toxicity data or crop dependence ratios. Specific to a decision-making context, can be at any scale.	Can be expensive. Link to evidence or real data is seldom explicit.
Ecosystem Approach	To maintain ecosystem services through integrated management of land, water and living resources.	Pollination can be included, using any of the above methods. No specific experience identified.	Considers multiple ecosystem services and trade-offs. Locally relevant. Works best at regional scale (landscape or catchment).	Can be an expensive and time-consuming. Requires large amounts of data.

45. Table 6.5.3. Utility of tools and methods for decision-making on pollinators at different levels of governance

An example for the food industry. ENGAGE = a tool to engage and communicate with users of the pollinator-related services. DESIGN = a tool to design or select appropriate responses.

Scale	**Farm**	**Regional**	**National**	**Global**

ACTORS (examples from the food industry)	Farmers	Suppliers Processors	Retailers Manufacturers Government	International agri-businesses Government
Case study/best practice approach	ENGAGE	ENGAGE	DESIGN	ENGAGE
Evidence synthesis		DESIGN	DESIGN	DESIGN
Risk assessment			DESIGN	DESIGN
Multi-criteria analysis	ENGAGE	ENGAGE + DESIGN	ENGAGE + DESIGN	ENGAGE + DESIGN
Cost-Benefit Analysis	ENGAGE + DESIGN	ENGAGE + DESIGN	ENGAGE + DESIGN	ENGAGE + DESIGN
Environmental Impact Assessment			DESIGN	DESIGN
Vulnerability Assessment		ENGAGE + DESIGN	ENGAGE + DESIGN	ENGAGE + DESIGN
Environmental Accounting			ENGAGE + DESIGN	ENGAGE + DESIGN
Mapping pollination	ENGAGE + DESIGN	ENGAGE + DESIGN	ENGAGE + DESIGN	ENGAGE + DESIGN
Modelling	DESIGN	ENGAGE + DESIGN	ENGAGE + DESIGN	ENGAGE + DESIGN
Participatory Integrated Assessment and scenario building			ENGAGE + DESIGN	ENGAGE + DESIGN
Decision support tools	DESIGN	DESIGN	DESIGN	DESIGN
Ecosystem Approach		DESIGN	DESIGN	DESIGN

6.6 Dealing with ecological uncertainty

Knowledge about the natural world and its complex relationships is inherently uncertain. Decision-makers faced with uncertain information need to know as much as possible about how much uncertainty there is and why it exists, in order to choose a course of action.

For scientific information, there has been considerable effort to clarify and manage uncertainty across different research fields (e.g., Elith et al., 2002; Regan et al., 2002; Walker et al., 2003; Norton et al., 2006; Li and Wu, 2006; Beale and Lennon, 2012; Kujala et al., 2013; Riveiro et al., 2014). Among the proposed taxonomies, frameworks, and modelling approaches, there is neither a commonly shared terminology (Walker et al., 2003) nor a comprehensive framework (see Mastrandrea et al., 2011 and Moss, 2011 for general uncertainties guidance). We therefore take a pluralist view and use all the available information to suggest how to improve the treatment of uncertainty in pollination research and management strategies.

Uncertainty assessment is not something to be added only *a posteriori* to interpret scientific results, management decisions or policy options. It is better to recognize it from the outset (Refsgaard et al., 2007). Perceiving, defining and analysing different sources of ecological uncertainty can increase the accuracy of risk estimation, improve models and predictions, and consequently improve control over the system. Although future drivers, effects or events cannot always be anticipated, environmental management or restoration of pollinators and pollination services can be performed in ways that tolerate ecological and economic uncertainty.

Table 6.6.1 summarises a general view of uncertainty. It is divided into four main sources: *linguistic, stochastic*, *scientific* and *epistemic*. Two or more types of uncertainty are identifiable within each source. This list of sources and types of uncertainty is not exhaustive.

For each type of uncertainty, we use examples from pollinator and pollination research to illustrate how its extent can be monitored, and/or how it can be reduced. For instance, incomplete knowledge of the ecological system (a type of epistemic uncertainty) and mistakes in observations (a type of scientific uncertainty) will always lead to uncertainty in predictions, but the extent of these types of uncertainty can be accounted for and potentially reduced in different ways. Table 6.6.2 suggests policy responses and applicable tools for the different sources of uncertainty.

The sources of uncertainty in Table 6.6.1 help to explain *why* there is uncertainty, rather than how much uncertainty there is. The overall amount of uncertainty, or level of confidence in a particular finding, combines different sources together and does not distinguish among them. This report defines the amount of uncertainty with consistent, well-defined terms based on authors' evaluations of the quantity, quality and consistency of the evidence and level of agreement for each finding (see IPBES Guidance on a Common Approach to Applying Uncertainty Terms, in preparation). These terms (well established, established but incomplete, unresolved, and inconclusive) are generally selected using expert judgement, although probabilistic or statistical information would be used if it were available.

46. Table 6.6.1. Summary of sources and types of uncertainty in ecological studies and ideas to quantify and/or diminish uncertainties, with examples for pollinators and pollination
(modified from Elith et al., 2002; Regan et al., 2002; Li and Wu, 2006; Keenan et al., 2011; Kujala et al., 2013; Mosadegui et al., 2013). Uncertainty is divided into four main sources, each given a plain English (and a technical) name in bold font. Two or more types of uncertainty are identifiable within each source.

***Sources* and types of uncertainty**	**Brief explanation and examples**	**Ideas for dealing with it in pollinator and pollination research**
1. Imprecise meanings of words **(Linguistic uncertainty)**	Uncertainty about language and meaning of expression.	Can be reduced through research and communication. Cannot easily be quantified.
1.a. Vagueness	Nature does not always arrange itself into strict classes, so sharp boundaries and homogenous classes do not represent reality. For example, categories for plant compatibility systems, or degrees of dependence on biotic pollination, are defined arbitrarily. Describing crop dependence ratios according to crop type, without specifying variety, ignores the variation among varieties.	Can be reduced by exposing clearly the meaning of categories, terms, and measurements, and the scale at which they are defined (e.g., Ruiz Zapata and Kalin Arroyo, 1978; Chautá-Mellizo et al., 2012; Liss et al., 2013).
1.b. Ambiguity	Words can have more than one meaning. For example, plant reproductive success can mean fruit set, seed set, pollen removal, pollen load, pollen tube growth or number, overall male and female reproductive output, and all can be used as measurements of pollination.	Can be reduced by exposing clearly the meaning of terms (concepts), indicators and dimension of the variables (e.g., Aguilar and Galetto, 2004).
2. Inherently unpredictable systems **(Stochastic uncertainty)**		Cannot be reduced through more research. Can be quantified and its potential impacts understood.
2.a. Randomness of nature	Chaotic or unpredictable nature of natural phenomena. For example, global climate change, extreme rainy/dry years, differences in pollination rates within the season, among sites, etc.	Can be identified through large-scale (spatial and temporal) studies (e.g., Brosi et al., 2008; Winfree et al., 2008; Aizen et al., 2009; Cameron et al., 2011; Garibaldi et al., 2011; Holzschuh et al., 2012) or by meta-analyses (e.g., Aguilar et al., 2006; Ricketts et al., 2008; Winfree et al., 2009). Competing factors can be clarified through experimental design. For example, effects of wind/bee pollination within the season (Hayter and Cresswell, 2006).
2.b. Economic fluctuations	The economic costs of employing managed pollinators can fluctuate strongly depending on availability and projected benefits. The value of pollination services to crops is strongly tied to the sale price of the crop. This may be influenced by market forces such as stochastic variations within the supply chain or agricultural subsidies.	An example for econometric analysis of the price of pollination service provision is Rucker et al. (2012). Crop price fluctuations can be analysed by statistical averaging or medians of prices over a series of years (Leonhardt et al., 2013).

***Sources* and types of uncertainty**	**Brief explanation and examples**	**Ideas for dealing with it in pollinator and pollination research**
3. Limits of methods and data **(Scientific uncertainty)**		Can be reduced through better quality research. Can be quantified and impacts understood.
3a. Measurement error	Imperfect measurements or techniques, e.g., available methodology may not record data precisely. For example, uncertainty in land-cover maps can propagate into ecosystem services maps (Eigenbrod et al., 2010; Schulp and Alkemade, 2013).	Selection of the best available measurements or techniques, and acknowledgement of this source of uncertainty.
3.b. Systematic error	Methods produce biased data, e.g., sampling of pollinators in a crop is always close to main roads; pan trap samples of pollinator communities systematically underestimate social bee abundance.	Experimental designs should include a reasonable heterogeneity for the experimental unit. For example, to evaluate the effects of the forest on *Macadamia* pollination, treatments were applied in orchards that varied in distance from rainforest, to compare the effects of the contrasting pools of available pollen vectors (Blanche et al., 2006). Bias in measurement techniques to evaluate the diversity of pollinators of different communities can be tested and controlled for (e.g., Popic et al., 2013).
3.c. Model uncertainty	Models are simplifications of real processes, and several alternative models may fit the same data. For example, there are different models for pollen dispersal in *Brassica napus* (Lavigne et al., 1998; Klein et al., 2006; Hoyle et al., 2007; Ceddia et al., 2007, 2009)	Models can be improved through their structure (i.e., modelling processes and formulation by equations and algorithms) or parameters (i.e., estimation, calibration).
3.d. Data uncertainty (or input uncertainty for modelling) and low statistical power	Studies of low data quality, low sample size, low number of replications or not fully representing relevant variation. For example, native bees provide pollination but how this varies with land management practices can be unknown.	Data sets can be improved through increasing sample size or replications, controlling heterogeneity, reducing missing data, etc. For example, native bee communities providing pollination for a crop (watermelon) with heavy pollination requirements (Kremen et al., 2002).
4. Differences in understanding of the world **(Epistemic uncertainty)**	Incomplete knowledge through available theory (web of concepts) and data. Uncertainty from subjective human judgments and beliefs. This might also be called decision uncertainty.	Can be reduced through further research. Can also be quantified and its potential impacts understood.

***Sources* and types of uncertainty**	**Brief explanation and examples**	**Ideas for dealing with it in pollinator and pollination research**
4.a. Natural and anthropogenic variations	Natural and agro-ecological systems are complex and hard to characterise because processes vary across space, time, etc. For example, crop pollination studies measuring fruit set or seed set have seldom taken account of the effects of nutrients, water and other limiting resources, also important for seed set (Bos et al., 2007).	Our knowledge can be improved through large-scale (spatial and temporal) studies (e.g., Brosi et al., 2008; Ricketts et al., 2008; Winfree et al., 2008; Aizen et al., 2009; Cameron et al., 2011; Garibaldi et al., 2011; Holzschuh et al., 2012), by reviews (e.g., Potts et al., 2010), meta-analyses (Aguilar et al., 2006; Winfree et al., 2009) and larger data sets. (e.g., Steffan-Dewenter et al., 2002).
4.b. Confusing reasoning	Uncertainty due to lack of clarity or differences in argument structure, derived hypothesis and/or predictions and/or experimental design. For example, pollinators may deliver services locally, but their individual behaviour, population biology and community dynamics could also be affected by a landscape scale.	Better articulation when presenting the theoretical framework, methodology, conclusions, etc., among the different researchers, institutions, social actors. For example, the development of a new conceptual model of how pollination respond to different spatial scales and land-use change (Kremen et al., 2007).
4.c. Subjective judgement or context dependence uncertainty	The same data set or the meaning of a concept can be differentially interpreted by experts from different research fields. For example, whether pollinator diversity and crop pollination are at risk depends on how you interpret the evidence, while different methods for assessing the economic value of pollination services capture different values of different benefits (Chapter 4).	The interpretation, judgements, and point of view can be better compared if the values, preferences, objectives, aims and goals can be exposed by each of the different researchers or institutions. For example, a debate between Ghazoul (2005a, b) and Steffan-Dewenter et al. (2005) about the existence of a global pollinator crisis and the uncertainty of human dependence upon pollination. Economic valuation should emphasise the context in which it was derived and a range of methods should be considered when examining trade-offs and benefits (e.g. Bos et al., 2007; Winfree et al., 2011; Cunningham and Le Feuvre, 2013; Arbetman et al., 2013; Cutler et al., 2014).
4.d. Human decisions under economic uncertainty	For example, non-Market values are difficult to assess and subject to a number of complexities in their elicitation (see Chapters 4 and 5). Different groups of people can experience different values from the same element of an ecosystem, or at a different time – beekeepers, almond growers and citrus growers in the same	Decisions can be clarified through careful development of survey instruments (Bateman et al., 2009). Focus upon final goods resulting from pollination (Fischer et al., 2009), or on values of a defined set of beneficiaries in a particular

***Sources* and types of uncertainty**	**Brief explanation and examples**	**Ideas for dealing with it in pollinator and pollination research**
	landscape view honey bee pollinators differently, for example (Sagoff, 2011).	context (Potschin and Haines-Young, 2013; Villa et al., 2014).

Table 6.6.1 clearly shows that the study of pollinators and pollination is a multi-dimensional social construct, and includes dimensions that involve the entire process (generation and communication) of the production of scientific knowledge.

The major area of discussion about uncertainty in the scientific literature concerns modelling processes and model selection, just one of the sources of uncertainty in Table 6.6.1 (e.g., Walker et al., 2003; Wintle et al., 2003; Li and Wu, 2006; Pappenberger and Beven, 2006; Rivington et al., 2006; Refsgaard et al., 2007; Ascough II et al., 2008; Cressie et al., 2009; Reilly and Willenbockel, 2010; Hildebrandt and Knoke, 2011; Keenan et al., 2011; Beale and Lennon, 2012; Rinderknecht et al., 2012; Mosadeghi et al., 2013; Riveiro et al., 2014; Sileshi, 2014).

Other sources of uncertainty are prominent in the use of pollinator and pollination science for policy and decision-making. For example, uncertainty surrounding the impact of sublethal effects of pesticides on pollinators might be considered an example of data uncertainty (a type of scientific uncertainty), because the true levels of field exposure are poorly known and the sublethal effects are only characterised for a small selection of pollinator species (see section 2.2.1.4). Maxim and Van der Sluijs (2007) also demonstrated epistemic uncertainty in the debate surrounding the insecticide imidacloprid in France, through the use of 'contradictory expertise’ leading to different interpretations; epistemic uncertainty includes variations in the interpretation of scientists about concepts, methodologies, data sets, and ethical positions that may come from different epistemological positions or understandings of the world.

Another area of uncertainty is the extent to which crop yields depend on pollination. There is stochastic uncertainty at local scales, because both yield and pollination, and their interaction, are affected by soil and weather conditions (see Chapter 3). Liss et al. (2013) found considerable variation in how the pollination is defined (linguistic uncertainty) and measured (scientific uncertainty), and recommended that pollination measurements and metrics are explicitly clarified (reducing linguistic and scientific uncertainties).

Finally, the effects of organic farming on pollinators (see section 6.4.1.1.4) look different if you take the view that wild nature beyond farmland has a higher value than farmland biodiversity, or overall food production at a large scale is more important than local impacts, because organic farms tend to have lower yields than conventional farms. Debates around organic farming are therefore subject to uncertainty that comes from confusing reasoning, an element of differences in understanding of the world.

47. Table 6.6.2. Suggested policy responses and applicable tools to account for or reduce different sources of uncertainty.

Source of Uncertainty	Qualities	Available policy responses and applicable tools
Imprecise meanings of words	Reducible Not quantifiable	• Clear, common definition of terms (such as the IPBES conceptual framework) • Develop and communicate standardised methods (such as the COLOSS Bee Book Neumann et al., 2013; 6.4.6.3.3)
Inherently unpredictable systems	Not reducible Quantifiable	• Clear communication • Support large-scale, long-term multi-site studies to quantify the variation over space and time • Evidence synthesis (6.5.2) • Vulnerability assessment (6.5.7) • Participatory Integrated Assessment and scenario building (6.5.11) • Multi-criteria analysis (6.5.4) • Decision support tools (6.5.12) • Precautionary principle
Limits of methods and data	Reducible Quantifiable	• Improve experimental design • Expand data collection • Support detailed, methodological research • Evidence synthesis (6.5.2) • Develop and communicate standardised methods (such as the COLOSS Bee Book Neumann et al., 2013; 6.4.6.3.3) • Capacity building for scientists • Precautionary principle
Differences in understanding of the world	Sometimes reducible Sometimes quantifiable	• Support detailed, site-based and modelling studies to understand systems • Acknowledge existence of biases • Acknowledge differences in conceptual frameworks (within and between knowledge systems) • Multi-criteria analysis (6.5.4) • Decision support tools (6.5.12) • Capacity building for decision makers

6.7 Trade-offs and synergies in decisions about pollination

This section reviews what is known about trade-offs and synergies among responses or policy options related to pollinators and pollination. A trade-off is considered as the simultaneous enhancement of one aspect of pollination and the reduction in other ecosystem services or another aspect of pollination. Synergy here is when two or more services, or aspects of pollination, are concurrently enhanced by the same action. Trade-offs and synergies need to be understood and acknowledged at all steps of the decision-making process about pollination and food production.

6.7.1 Trade-offs and synergies between pollination and other ecosystem services

Ecosystem services and pollination encompass various natural processes and are surrounded by sociological systems, so trade-offs and synergies between them need to be well thought out. For instance, actions to maximize crop pollination and conservation of culturally important pollinators may be in conflict with the other. Research analyzing how a single focused response affects trade-offs and synergies among pollination and other ecosystem services, as well as the economic costs and benefits, should be considered. For example, Kleijn et al. (2015) recently demonstrated that simple actions such as planting flowers to support crop pollinators (see section 6.4.1.1.1) do not necessarily also support declining or specialised species of wild bee. They suggest that managing for pollinator diversity requires different actions, more focused on habitat protection or restoration.

It is important to understand whether multiple ecosystem services changing together are responding to the same driver or interacting with each other (Bennett et al., 2009). It is also necessary to consider trade-offs and synergies among sectors, stakeholders, or constituents because each ecosystem service is used differently by diverse groups of humans.

Several reviews and meta-analyses have examined the trade-offs and synergies among multiple ecosystem services alongside pollination. Reviews have indicated that the creation and conservation of pollinator habitats, such as biologically diverse faming systems in agricultural landscapes, can enhance biodiversity and several ecosystem services such as natural pest control, soil and water quality, and rural aesthetics (Kremen and Miles, 2012; Wratten et al., 2012). In coffee and cacao agroforestry systems, it has been shown that the presence of shade trees, which enhances the presence of pollinators, could lead to synergies such as pest control (Tscharntke et al., 2011). Natural habitats provide pollinator habitats and facilitate the movement of organisms that can be providers of other ecosystem services (Mitchell et al., 2013). In a meta-analysis, Shackelford et al. (2013) compared the abundance and richness of pollinators and natural enemies in agricultural landscapes and found that some pollinators and natural enemies seem to have synergetic responses, although the evidence is limited. An investigation of the relationship between the genetic diversity of crops and the delivery of ecosystem services implied that increasing crop genetic diversity was useful in pest and disease management, and might have the potential to enhance pollination (Hajjar et al., 2008). Breeding crops to reduce pollinator dependence (see section 6.4.1.1.11) could reduce production uncertainty or instability in the short term, but this can reduce overall crop genetic diversity, thus increasing potential vulnerability to pests and diseases (Esquinas-Alcázar, 2005).

A case study on a *Cordia alliodora* plantation in Ecuador indicated that economic trade-offs do not necessarily occur among timber provision, regulation of carbon dioxide, and pollination of adjacent coffee crops with moderate silvicultural interventions (Olschewski et al., 2010). A modeling study in the United States indicated trade-offs between income provision and other ecosystem services, including pollination, when replacing annual energy crops with perennial energy crops (Meehan et al., 2013). Several spatially explicit frameworks to investigate the trade-offs of multiple ecosystem services, with pollination estimated mainly by the proxy of natural vegetation, found both negative and positive correlations between pollination and other ecosystem services. Pollination was weakly negatively correlated with forage production, and weakly positively correlated with carbon storage and water provision in the United States (Chan et al., 2006). Positive relationships of pollination and water quality regulation with recreational and commercial fisheries were found in Australia (Butler et al., 2013).

Using a spatially extensive data set of trade-offs and synergies for Great Britain, Maskell et al. (2013) demonstrated that nectar plants for bees were positively correlated with other services or service providers, such as plant species richness and soil invertebrate diversity. Additionally, trade-offs and synergies between pollination, indexed by the sampling of actual pollinators and/or the pollination success of plants and other ecosystem services, have been reported. A study conducted in the United Kingdom that examined the effects of grazing management showed that grazing intensity did not affect potential pollinators or total carbon stock, but affected some groups of pest-regulating invertebrates (Ford et al., 2012). Another study in the United States, of perennial bioenergy crops that provide an alternative to annual grains, found that pollination, methane consumption, pest suppression and conservation of grassland birds were higher, whereas biomass production was lower in perennial grasslands (Werling et al., 2014).

6.7.2 Trade-offs between pollination and food provisioning services (crop yield and honey)

Among ecosystem services, provisioning services, especially food production, are likely to be a priority for human societies. Therefore, trade-offs between pollination and provisioning services (e.g., crop yield and honey) warrant special consideration.

There is potentially a direct trade-off between using land to grow food and using land to provide pollinator habitat. To illustrate, using farmland to provide flower strips or other pollinator habitat (see section 6.4.1.1.1) takes land out of production and so overall yields may be lower. However, because there may be existing pollination deficits (see Chapter 3, section 3.8.3), and management for pollinators has been shown to enhance crop yields (6.4.1.1.1), it is important to calculate the net yield and economic outcomes of such management at both farm and landscape scales. There is a major knowledge gap about the net yield effects of managing for pollinators in different farming systems. Elements of it have been analysed for a few farming systems or contexts.

A model-based study of a low intensity agricultural system in northern Scotland examined the trade-off between the conservation of bumble bees and agricultural income, and showed that both agricultural profits and bumble bee densities can be enhanced (Osgathorpe et al., 2011). A study of coffee production systems in India (Boreux et al., 2013) found that management to enhance pollination (use of shade trees) slightly increased coffee yields, but much greater increases in production could be achieved through liming (no influence on pollination), or irrigation timed to promote flowering when other coffee farms were not flowering. Irrigation enhances the pollination without the light and nutrient costs of shade plants, but it is a very context-dependent solution. Another way to reduce the trade-off between providing habitat for pollinators and net yield is to provide pollinator habitat on low-yielding, sometimes called ‘marginal’ land, such as field edges or steep slopes.

Organic farming and diversified farming systems contribute to maintaining pollinator habitats and effective crop pollination, but many studies indicate that these farming systems are often, not always, less productive than conventional agricultural management (Badgely et al., 2007; de Ponti et al., 2012; Seufert et al., 2012; Ponisio et al., 2015) (see Chapter 2, 2.2.3). Here again there is apparently a direct trade-off between management to enhance pollination and yield. Yields on organic farms are on average around 20-25% lower than on conventional farms (Ponisio et al., 2015: 19.2%; Seufert *et* al. 2012: 5-34%, depending on the system). We could not find any analysis to indicate how observed increases in pollinator abundance, diversity and pollination on organic or diversified farms (see section 6.4.1.1.4 and 6.4.1.1.8) contribute to reducing this trade-off. However, there is clear evidence that the trade-off can be reduced by practices that could be considered diversification, or ecological intensification (see Chapter 1 for definitions) on organic farms, such as multi-cropping and crop rotations (see section 6.4.1.1.8). These practices reduced the yield gap between organic and conventional farms to 9% and 8% respectively (Ponisio et al., 2015). It has also been suggested that the trade-off could be minimised by encouraging organic farming in landscapes with low productivity due to soil or climate conditions, where yield differences between organic and conventional agriculture are lower (see section 6.4.1.1.4).

Elmqvist et al. (2011) emphasize the importance of incentives, institutions and governance in effectively managing trade-offs between provisioning services and regulating services, including pollination, in agricultural landscapes. For example, they suggest payments for ecosystem services (see section 6.4.3.3), or compensation through incentive payments or certification schemes (see section 6.4.1.3), can allow farmers to retain equivalent income with lower yields, in return for improvements to the landscape as a whole.

Honey bees are managed for honey production as well as crop pollination, and there is a trade-off between these if the best food sources or landscapes for honey production are not the same as the landscapes where pollination are needed (Champetier, 2010). For example, honey bees are taken to almond orchards for pollination, but this reduces production of honey. This trade-off is compensated for in pollination markets by increased pollination fees (Champetier, 2010).

6.7.3 Trade-offs between pollination and ecosystem dis-services

Food-producing ecosystems also generate ecosystem dis-services that reduce yield or increase production costs, in addition to providing ecosystem services. Ecosystem dis-services, such as pest damage caused by birds or insects, can potentially be enhanced when using an ecosystem approach to enhance pollination. The trade-offs between a pollination and ecosystem dis-service could depend on the sectors and the stakeholders or humans involved. To manage the potential trade-offs, it is necessary to analyze the economic and social costs and benefits and explore their interactions.

Review publications have assessed the trade-offs between pollination and ecosystem dis-services provided by potential pollinators and their habitats. The available evidence suggests that promoting bird species diversity in agricultural landscapes would enhance both pollination and pest control services and ecosystem dis-services such as the consumption of crops by birds, although more studies are needed to quantify the costs and benefits (Triplett et al., 2012). Marshall and Moonen (2002) reviewed the ecological effects of field margins in Europe and reported that having semi-natural field margins can create habitats for pollinators, but some field margins will lead to some ecosystem dis-services in lower crop yield due to weed and pest species that spread into cropland. Another review reported that having non-crop habitat for pollinators may result in competition for pollination from flowering weeds and non-crop plants, which would reduce crop yields (Zhang et al., 2007). Additionally, competition for pollinators between crops and wild plants might result in a potential threat to the fitness of concurrently-flowering wild plants (Holzschuh et al., 2011).

6.7.4 The importance of spatial scale, location and timescale to trade-offs and synergies

Management of pollinators requires consideration not only at the local field scale, where services are delivered, but also at the larger surrounding landscape scale. This is because pollinators depend on habitats for nesting, larval development, mating or overwintering that are often spatially segregated from the flowers where they feed. There is a potential for trade-offs or synergies among spatial scales, because the effects of actions taken at one spatial scale to support pollinators can depend on what is happening at a different spatial scale. For example, a meta-analysis showed that pollinators benefit from agri-environmental management at a local scale in simple, but not in complex landscapes (Batáry et al., 2011). This means actions at landscape scale to improve landscape complexity could potentially make local scale actions such as planting flower strips less effective (a trade-off). A case study in blueberry fields in the United States showed that the scale at which land cover had the strongest effect on bee abundance varied according to bee body size (Benjamin et al., 2014). In this case, actions tailored to support larger bees would not be expected to benefit smaller bees, because they would be at an inappropriate scale.

There are cases where pollinators move between different countries. Then, conservation action in one country can either have synergy with conservation action in the other country, or trade off against habitat destruction or adverse management for pollinators in the other country. For example, long-nosed bats (genus *Leptonycteris*), which are pollinators of agave plants, move between Mexico and the United States (Lopez-Hoffman et al., 2010).

In addition to the spatial trade-offs, there must also be trade-offs between the present and future pollination, although management decisions often focus on an immediate time frame (Power, 2010). Technical developments associated with pollinators, pollination systems, and pollination may increase future food production, whereas some practices used to provide foods confer economic benefits in the present, but might be costly in the future.

6.7.5 Trade-offs and synergies among responses

Different responses can have opposing or synergistic effects on different aspects of pollinators or pollination. For instance, using managed pollinators to promote crop pollination may have negative impacts on native biodiversity, including wild pollinators (see section 6.4). This could lead to economic consequences for producers that may be passed onto consumers (Rucker et al., 2012). There can be trade-offs among responses for pollinators and responses designed to protect other elements of ecosystems (see case study: Eucalyptus trees and honey bees in South Africa).

Kitti et al. (2009) used an economic model to assess whether measures to reduce poverty (minimum wages for labourers) or protect forest (conservation payments for retaining forest) lead to conflicting outcomes in a coffee producing area of Costa Rica. Their model accounted for the positive impact of forest patches on pollination. In this context, minimum wages did not favour the production of ‘sun coffee’, and would not lead to a decrease in forest cover, so there was not a trade-off between forest protection and poverty reduction.

CASE STUDY: Eucalyptus in South Africa: bad for water, good for bees

The Working for Water programme in South Africa was founded in 1995 to clear non-native plants while providing social services and rural employment. Australian eucalyptus trees were a focus of the programme, because they are heavy water users. Beekeepers in all South African provinces depend heavily on eucalyptus trees as a forage resource for their honeybees and were very worried about large-scale removal of eucalyptus. The Department of Environmental Affairs funded the Honeybee Forage Project (http://www.sanbi.org/biodiversity-science/state-biodiversity/applied-biodiversity-research/global-pollination-honeybee-fo) to provide evidence about the importance of eucalyptus for honey bees and to search for indigenous replacements. This project has confirmed that the amount of bee forage provided by eucalyptus trees is not replaceable from indigenous plant communities. Negotiations between beekeepers and conservationists to resolve this issue are ongoing. One element of compromise is that landowners can apply for a permit to demarcate their listed eucalyptus trees as “bee-forage areas”, as long as they are not in water courses or invading into natural vegetation.

6.8 Gaps and future research

There have been four independent exercises to identify important research questions, or knowledge needs, relating to pollinators and pollination. One was a scientific exercise that defined 86 research questions in from evolution and ecology to implementing pollinator conservation (Mayer et al., 2011). Two defined key questions related to pollinators from the perspective of end-users of research, involving policy makers, businesses and non-Governmental organisations (Ratamaki et al., 2011; Dicks et al., 2012). In both of these exercises, the role of pollinator diversity and the relative importance of wild and managed pollinators in crop production were identified as prominent and high priority questions. Sutherland et al. (2011) assessed synthesized evidence to identify ten research priorities on wild bee conservation (see section 6.5.2).

There is no published analysis of the extent to which the questions or research priorities are being addressed by current research effort. It is likely that many are, especially through the pollinator-focused research efforts described in section 6.4.6.3.2.

6.8.1 Agricultural, agroforestry and horticultural practices

More research is required to establish firmly the impact on food production of planting and managing new pollinator forage resources into agricultural landscapes. Such research could focus on: What flowering species are needed to support the nutritional needs of the required pollinator communities? When to sow, when to cut? How does the quantity (total and area margin/area of crop) and configuration (location, connectedness of patches) of field margins impact their effectiveness on pollinators and services? Studies should measure the effects of enhancing floral resources at local and landscape scales, on pollination and on populations of pollinators measured at larger spatial scales than individual fields.

The net yield and economic outcomes of such management, at both farm and landscape scales are a major knowledge gap that has been analysed for very few farming systems or contexts (see section 6.7.2).

Another research gap is in identifying crop mixes that can promote pollinator species and communities. A recent study suggested that abundance of pollinator communities is as enhanced by polyculture as it is by surrounding natural habitat (Kennedy et al., 2013). Thus areas that are planted to productive crops could, in combination with margin enhancements, support pollination.

Similar attention needs to be paid to the possibilities of increasing nesting resources for pollinators, which could be a limiting factor in agricultural landscapes. These studies must be accompanied by investigations of farmers' acceptance and motivations to introduce such measures on their land.

Ecological intensification emerges as a priority strategy in countries where agricultural production is already approaching maximum exploitable yields, with the principal aim being to reduce environmental costs and erosion of ecosystem services that are now under pressure. A main priority for supporting food security should be directed at closing existing yield gaps around the world with ecological enhancement (Bommarco et al., 2013). Findings ways to reduce the apparent trade-off between yield increases and pollinator benefits (as shown in studies on organic farming, for example) is an inherent part of this research programme (see section 6.4.1.1.4 and 6.7.2).

The effects of climate change on plant-pollinator interactions are still mostly unknown, so adapting farming methods to deal with global warming requires substantial additional research, especially in the tropics.

Interdisciplinary research that combines ecological, economic, social and psychological research to elucidate the processes underlying successful agri-environmental policies is greatly needed around the world.

Finally, transdisciplinary work is essential to implement pollinator-supporting practices in real-world landscapes and support long-term yields of pollinator-dependent crops (Garibaldi et al., 2014). Developing farmer-researcher platforms or networks, helping researchers to interact with farmers and understand farmer problems, and assisting researchers to work within the complexity of on-farm research (e.g. http://aeix3dev.devcloud.acquia-sites.com), are key ways of finding practical answers in a context that involves the participation of farmers.

6.8.2 Pesticides, pollutants and genetically modified organisms

Research is needed for more accurate predictions of exposure and risks, to inform approaches to reduce the exposure of pollinators to pesticides, and to help determine the impacts of pesticides on pollinators.

Risk assessment tools will need to be further developed and implemented. Impacts assessments need to address adverse sublethal effects and risks to wild bees. For instance, a risk assessment based on a literature review identified lack of exposure and toxicological information for pollinators other than the honey bee as the primary area of uncertainty (Cutler et al., 2014b). Knowledge gaps include mitigation of negative impacts of pesticides on pollination (Nienstedt et al., 2012), on actual population trends and dynamics of pollinators, and of combined effects of multiple environmental pressures and pesticides, or mixes of pesticides and other pollutants on pollinators (Gonzaléz-Varo et al., 2013).

A development of specific risk indicators from exposure of pesticides to pollinators would be useful for evaluating possible impacts on pollinators of risk reduction programmes.

Higher-tier registration studies are costly to perform and process, and it is not necessary to repeat them in each country. Sharing information among countries can help raise and harmonise registration standards globally. Making registration studies available globally needs to be accompanied by raising the skills to interpret the studies and distinguish which studies may not be necessary to conduct locally.

There is no global overview of pesticides regulation among countries. Efforts to reduce risks need to be directed to regions and crops in which pollinators and pollination are most probably at the highest risk. Schreinemachers et al. (2012) give a nice overview of the pesticide use in the world related to economy type; it is highest in middle income economies. Most crop pollination values are generated in Asia while 58%, 8% and 10% are generated in Africa, and South and Central America, respectively (Gallai, 2009) where pesticide use is also high. If this information were matched with where regulation is weak, where and in which crops impact studies have been performed (probably mainly in field crops in Europe, North America and Brazil), there is a high probability to find clear mismatches and knowledge gaps.

Continual investments into agricultural research and development of technology are needed that reduce risk to pollinators. Research funding to develop IPM strategies and crop production systems with no or reduced use of pesticides, would provide options to decrease exposure and risks to pollinators. Cost-benefit comparisons of IPM or no-pesticide options against conventional pesticide use are also needed. Assessing pollination dependence in flowering crops that are now considered self-pollinated remains to be performed for major crops. For instance, pollinators contribute to crop yield in soy beans, but pest management is not considering pollination in soy beans (Chiari et al., 2005; Milfont et al., 2013).

It is clear that adverse effects for beneficial organisms such as pollinators from exposure to pesticides can be reduced. There are, however, few examples where the actual effectiveness of these efforts has been estimated specifically for pollinators.

Many pesticides are used in urban green spaces. Risk management and risk mitigation for pollinators is poorly developed for urban settings and amenity areas. Education and awareness-raising targeted at gardeners and professional managers of urban amenity areas (e.g., playing fields and golf courses) need more attention.

There is also a lack of standardized monitoring and research of GM-crop impacts on pollinators. Risk assessment of GM-crops on non-target organisms needs to be developed for bee species other than the honey bee, for GM organisms in combination with environmental stressors, and on populations and communities of pollinators (Arpaia et al., 2014).

6.8.3 Nature conservation

Research is needed to understand better how the composition and configuration of the landscape affects plant-pollinator interactions. More studies are needed that address the diversity of pollinators and population attributes (e.g., density fluctuations and survival) and to evaluate changes in diversity and behavioural attributes (e.g., species mobility and foraging patterns) that could affect the efficiency of different pollinators. These knowledge gaps apply equally to crop pollinators and wild plant pollinators.

That type of research is particularly needed for tropical ecosystems, where the recent increase in the number of studies has been lower than in temperate regions and where the higher diversity of plants and pollinators impedes a more thorough knowledge of these systems. Due to the high worldwide importance of those regions for the production of food and primary agricultural goods, more attention should be given to the development of knowledge of pollinators and pollination processes in complex tropical landscapes (Viana et al., 2012).

Lennartson (2002) states that habitat loss and fragmentation can lead to abrupt qualitative changes in landscape structure, limiting the survival and movement of pollinators. To conserve pollinator diversity properly, habitat loss should never reach threshold levels that lead to local extinctions of pollinator species (Radford et al., 2005). However, the critical threshold levels of habitat loss that could lead to drastic increases in pollinator extinction rates and the collapse of plant-pollinator interaction networks (Viana et al., 2012) are not known.

Understanding how pollen is dispersed and investigating the factors that affect pollinator mobility are essential, in order to design land management strategies that can secure crop and wild plant pollination. However, to complete this task, methodological and technical obstacles must be overcome. The development of better individual tracking technologies will inevitably lead to more detailed studies on pollinator movement through the landscape, which together with the knowledge already available in the literature will lead to the development of better tools and guidelines for the management and design of landscapes with highly-efficient ecosystem services, also ensuring the long-term conservation of pollination in agro-natural systems (Viana et al., 2012).

Studies to evaluate the effectiveness of ecosystem service payments or stewardship mechanisms to protect pollinators and pollination are also needed for both developed and developing countries.

As taxonomic capacity is essential for pollinator monitoring, conservation and management, a targeted effort is needed to surmount the taxonomic impediment: the adequacy and accessibility of identification services, the status of taxonomic knowledge, and the provision of tools to assist non-experts in identification.

Policy makers need to have concrete, practical information on pollinator declines which can only be provided by a broad, collaborative global effort to monitor pollinator trends and status effectively. Then strategies are required for monitoring in the face of large expected natural pollinator population variation (FAO, 2008).

6.8.4 Pollinator management and beekeeping

There is a clear need for research on how to improve or optimise the pollinating abilities of managed pollinators, and to develop management techniques for new pollinator species suitable for different crops.

More research is needed on the effects of combined interventions in managing pollinators, to determine when and how different interventions interact. Such research could focus more generally on best practices for pollinator management; these practices in many cases should be developed to be regionally specific.

However, the most prominent knowledge gaps on managed pollinators are related to the control of parasites and pathogens. Major gaps are:

6.8.4.1. Detection / Diagnosis

1. Improvements are needed in terms of speed, reliability, cost, and accessibility of diagnostic tests.
2. From a policy perspective, a key knowledge gap is how best to link inspections of managed bees and detection of parasite / pathogen problems to legal responses.

6.8.4.2. Prevention

1. How to manage pollinator movement across multiple spatial scales to reduce the spread of infection, especially without greatly interfering with the delivery of pollination and farmer and beekeeper profitability, is a key policy challenge and knowledge gap.
2. Another key policy challenge and knowledge gap is how best to reduce infection spread and support best management practices in rearing facilities while maintaining profitability, especially for bumble bees, but potentially for other bee species in the future

6.8.4.3. Treatment

1. Overall, treatment of parasites and pathogens of managed pollinators is a major knowledge gap and there are few parasite / pathogen problems with effective treatment strategies.
2. Little is known about treatment options for managed pollinators other than honey bees, comprising another general knowledge gap.

3. Treatment of viral diseases is a key knowledge gap, as there are no known effective treatments for any viral diseases of managed pollinators.
4. Control of *Varroa* mites, the single largest cause of honey bee colony losses worldwide, is another major knowledge gap. This is particularly true given that *Varroa* has evolved resistance to miticide treatments that were previously very effective.
5. Interference RNA (RNAi) technology has been shown in laboratory, and limited field trials, to reduce viral diseases and *Varroa* mites, and to improve beekeeping outcomes in honey bees, but the optimization and commercialization of this technology represent a specific knowledge gap. An additional knowledge gap is the use of RNAi against parasites and pathogens other than viruses and *Varroa*.
6. Fungal diseases of managed bees, represented primarily by *Nosema*, stonebrood, and chalkbrood, have few treatment options. *Nosema* in honey bees (but not bumble bees) is controlled in some countries by the antifungal agent fumagillin, but it is expensive and toxic to mammals, and likely has toxicity impacts on honey bees as well. Alternatives to fumagillin and development of antifungal agents effective against chalkbrood and stonebrood present another knowledge gap.

6.8.4.4. Social Immunity

1. Social managed pollinators (including honey bees, bumble bees, and social stingless bees) have evolved elaborate defense mechanisms at a group (rather than individual) level. A knowledge gap is understanding these “social immunity” defense mechanisms, and how to protect and support them in managed taxa, especially given that there is some evidence of common management practices disrupting social immunity.

6.8.4.5. Management of pathogen and parasite evolution

1. Little is known about best management practices for reducing the evolution of resistance by parasites and pathogens of managed bees to treatments.
2. We know little about managing pollinators, and their parasites and pathogens, to select for less-virulent parasites or more-resistant pollinators.

6.8.5 Urban and transport infrastructure

Currently around half the world's population lives in urban areas and this is set to increase dramatically during the next 50 years (Grimm et al., 2008), yet pollination and pollinator conservation are not a major focus of urban design or policy.

Many initiatives are underway to restore or create urban green space, but the success of these efforts often fails to evaluate the effect on pollinators (Lomov et al., 2010).

Early successional habitats such as urban brownfields and vacant land provide valuable foraging habitat for pollinators, yet these areas are not considered important in conservation planning (Gardiner et al., 2013). Determining how to manage these habitats to support pollinators is critical to sustaining needed pollination.

Studies conducted in developing countries, where urban food production is much more extensive, suggest that urban agriculture can provide extra nutrition and food security for households (Maxwell et al., 1998; Drescher, 2004). However, there is a great lack of knowledge from some of the most rapidly developing cities within China and India, addressing the importance of garden and allotment food production in both the developing and developed world. The vast majority of studies have been performed in Brazil, USA and Europe (primarily Northern Europe) (Hernandez et al., 2009).

Organizations and governments have identified right-of-way infrastructure as a key way to support pollinators and connect habitat patches, however, there are few policy strategies underway to institute these efforts for large-scale landscape management.

Finally, studies are essential to evaluate the impact of urban management on pollination, the value of pollination for food production in cities, and the efficient and economic options for managing right-of-way infrastructure to support pollinators.

6.8.6 Tools and methods

The most prominent knowledge gap when it comes to comparing responses is the lack of information on relative costs of different responses. There has been a great deal of research to assess the value of pollinators and pollination (see Chapter 4), and to measure the effectiveness of different measures. Researchers and policymakers must now work together to quantify the costs, and find viable measures of relative effectiveness, for the different responses discussed in this report.

We urge ongoing investment in method development for identifying best practice, risk assessment, vulnerability assessment, mapping pollination, and decision support tools. There are a number of specific gaps, or methodological uncertainties.

For example, it is necessary to analyse the strengths and weaknesses of methods for mapping pollination and validating pollination maps. Mapping techniques should be standardised to improve the use of pollination information in decision making. The pollination must be incorporated into global Integrated Assessment Models to accomplish new perspectives for stakeholders when deciding on complex environmental problems.

Risk assessment methods for wild pollinators and sub-lethal effects of current practices in agro-environments have still to be considered when quantifying and mapping the supply or demand of pollination.

The diversity of pollinators and pollination should be incorporated into a range of standard model sets for analysing trade-offs between ecosystem services, especially pollination with treatment of non-monetary values such as, for example, the value loss associated with a decrease of native pollinators.

6.9 Conclusion

The available strategic responses to the risks and opportunities associated with pollinators range in ambition and timescale, from immediate, relatively easy responses to reduce or avoid risks, to larger scale, long-term transformative responses. Table 6.9.1 describes seven strategies, linked to actions responding to risks and opportunities, including a range of solutions that draw on Indigenous and Local Knowledge (ILK). These strategies can be adopted in parallel, and would be expected to reduce risks associated with pollinator decline in any region of the world, regardless of the extent of available knowledge about the status of pollinators or the effectiveness of interventions. The first two strategies (‘Manage immediate risks’ and ‘Exploit immediate opportunities’) are relatively short-term and low in ambition. Some, not all, of the specific responses involved would also be part of the longer-term, more ambitious strategies.

We envisage three possible strategies for moving towards more resilient, sustainable agriculture in the longer term, with an associated reduction in risks generated by pollinator decline: i) ecological intensification, ii) investing in ecological infrastructure and iii) strengthening existing diverse farming systems. These are not mutually exclusive, but each has a different focus. Definitions of ecological intensification, diversified farming, and other farming systems are provided in Chapter 1.

Ecological intensification (Bommarco et al., 2013; Tittonell, 2014) emphasizes management that increases the intensity of ecological processes that support production, such as biotic pest regulation, nutrient cycling, and pollination. It involves making smart use of nature’s functions and services, at field and landscape scales, to enhance agricultural productivity and reduce reliance on agro-chemicals. The end point of ecological intensification is a farming system that is likely to meet the definition of a diversified farming system.

Some specific actions that farmers or land managers may take to achieve ecological intensification are the same as those that would improve current conditions for pollinators, listed in the first two rows of Table 6.9.1, such as creating flower-rich field margins or road verges. In ecological intensification, these actions would be actively designed to support pollination of specific crops in the locality.

Strengthening existing diversified farming systems is an important strategic response because there is clear evidence that such systems support a higher diversity and abundance of pollinators. Diversified farms integrate the use of a mix of crops and/or animals in the production system. Many such systems are practised by indigenous peoples and local communities across the globe, and contribute to maintenance of pollinators and pollination resources (see Chapter 5, section 5.2.8).

The **ecological infrastructure** needed to benefit pollination comprises small to medium-sized patches of semi-natural habitat, providing nesting and floral resources, distributed throughout productive agricultural landscapes (see section 6.4.3.1.1). The same approach can also be expected to benefit the diversity of pollinators and pollination of food crops in urban areas (see sections 6.4.5.1.1 and 6.4.5.1.2). Such distributed ecological infrastructure may not be the same as the infrastructure needed for other ecosystem services or elements of biodiversity. For example, wild species associated with natural habitats such as wetland or forest may benefit more from protection of larger areas of habitat (tens or hundreds of hectares), separated from agriculture (Phalan et al., 2011), while other species, including some pollinators, rely on entire landscapes with diversified farming systems (Loos et al., 2014; Sutcliffe et al., 2014).

Finally, pollinators and pollination offer a real opportunity to begin to transform the relationship between humans and nature, because of their tangible values (Chapter 4), and the demonstrable benefits of sharing knowledge systems and working collaboratively across sectors (see Table 6.9.1).

48. Table 6.9.1 Overview of strategic responses to risks and opportunities associated with pollinators and pollination.

Examples of specific responses are provided, selected from chapters 5 and 6 of the assessment report to illustrate the scope of each proposed strategy. This is not a comprehensive list of available responses and represents around half of the available options covered in the assessment report. Not all the responses shown for “improving current conditions” will benefit pollinators in the long term, and those with potential adverse, as well as positive, effects are marked with an asterisk. All the responses from chapter 6 that are already being implemented somewhere in the world and have well established evidence of direct (rather than assumed or indirect) benefits to pollinators are included in the table and are highlighted in bold.

Ambition	Strategy	Examples of responses	Chapter references
Improving current conditions for pollinators and/or maintaining pollination	**Manage immediate risks**	• **Create uncultivated patches of vegetation such as field margins with extended flowering periods**	2.2.1.1, 2.2.1.2, 2.2.2.1.1, 2.2.2.1.4, 6.4.1.1.1, 5.2.7.5, 5.2.7.7, 5.3.4
		• Manage blooming of mass-flowering crops*	2.2.2.1.8, 2.2.3, 6.4.1.1.3,
		• Change management of grasslands	2.2.2.2, 2.2.3, 6.4.1.1.7
		• **Reward farmers for pollinator-friendly practices**	6.4.1.3, 5.3.4
		• Inform farmers about pollination requirements	5.4.2.7, 2.3.1.1, 6.4.1.5
		• Raise standards of pesticide and genetically-modified organism (GMO) risk assessment	2.3.1.2, 2.3.1.3, 6.4.2.1.1, 6.4.2.2.5
		• Develop and promote the use of technologies that reduce pesticide drift and agricultural practices that reduce exposure to pesticides	2.3.1.2, 2.3.1.3, 6.4.2.1.3, 6.4.2.1.2
		• Prevent infections and treat diseases of managed pollinators; regulate trade in managed pollinators	2.4, 6.4.4.1.1.2.2, 6.4.4.1.1.2.3, 6.4.4.2
		• Reduce pesticide use (includes Integrated Pest Management, IPM)	6.4.2.1.4
	Utilize immediate opportunities	• Support product certification and livelihood approaches	5.4.6.1, 6.4.1.3
		• **Improve managed bee husbandry**	2.4.2, 4.4.1.1, 5.3.5, 6.4.4.1.3
		• Develop alternative managed pollinators*	2.4.2
		• Quantify the benefits of managed pollinators	6.4.1.3, 6.4.4.3
		• Manage road verges*	2.2.2.2.1, 6.4.5.1.4, 6.4.5.1.6
		• Manage rights of way and vacant land in cities to support pollinators	2.2.2.3, 6.4.5.1.4, 6.4.5.1.6, 6.4.5.4
Transforming agricultural landscapes	**Ecologically intensify agriculture through active management of ecosystem services**	• **Support diversified farming systems**	2.2.1.1, 2.2.1.2, 2.2.2.1.1, 2.2.2.1.6, 5.2.8, 5.4.4.1, 6.4.1.1.8
		• Promote no-till agriculture	2.2.2.1.3, 6.4.1.1.5
		• Adapt farming to climate change	2.7.1, 6.4.1.1.12
		• Encourage farmers to work together to plan landscapes; engage communities (participatory management)	5.2.7, 5.4.5.2, 6.4.1.4
		• Promote Integrated Pest Management (IPM)	2.2.2.1.1, 2.3.1.1, 6.4.2.1.4, 6.4.2.2.8, 6.4.2.4.2

Ambition	Strategy	Examples of responses	Chapter references
		- Monitor and evaluate pollination on farms	5.2.7, 6.4.1.1.10
		- Establish payment for pollination services schemes	6.4.3.3
		- Develop and build markets for alternative managed pollinators	6.4.4.1.3, 6.4.4.3
		- Support traditional practices for managing habitat patchiness, crop rotation and co-production of knowledge between indigenous and local knowledge holders, scientists and stakeholders	2.2.2.1.1, 2.2.3, 5.2.7, 5.4.7.3, 6.4.6.3.3
	Strengthen existing diversified farming systems	- **Support organic farming systems;** diversified **farming systems; and food security**, including the ability to determine one's own agricultural and food policies, resilience and ecological intensification	2.2.2.1.1, 2.2.2.1.6, 5.2.8, 5.4.4.1, 6.4.1.1.4, 6.4.1.1.8
		- Support biocultural diversity conservation approaches through recognition of rights, tenure and strengthening of indigenous and local knowledge and traditional governance that supports pollinators	5.4.5.3, 5.4.5.4, 5.4.7.2, 5.4.7.3
	Invest in ecological infrastructure	- **Restore natural habitats** (also in urban areas)	6.4.3.1.1, 6.4.5.1.1, 6.4.5.1.2
		- Protect heritage sites and practices	5.2.6, 5.2.7, 5.3.2, 5.4.5.1, 5.4.5.3
		- Increase connectivity between habitat patches	2.2.1.2, 6.4.3.1.2
		- Support large-scale land-use planning and traditional practices that manage habitat patchiness and biocultural diversity	5.1.3, 5.2.6, 5.2.7, 5.2.9, 6.4.6.2.1
Transforming society's relationship with nature	**Integrate peoples' diverse knowledge and values into management**	- Translate pollinator research into agricultural practices	2.2.1, 2.2.2, 2.2.3, 2.2.1.2, 6.4.1.5, 6.4.4.5
		- Support knowledge co-production and exchange among indigenous and local knowledge holders, scientists and stakeholders	5.4.7.3, 6.4.1.5, 6.4.6.3.3
		- Strengthen indigenous and local knowledge that fosters pollinators and pollination, and knowledge exchange among researchers and stakeholders	5.2.7, 5.4.7.1, 5.4.7.3, 6.4.4.5, 6.4.6.3.3
		- Support innovative pollinator activities that engage stakeholders with attachments to the multiple socio-cultural values of pollinators	5.2.3, 5.3.2, 5.3.3, 5.3.4, 5.4.7.1, 6.4.4.5
	Link people and pollinators through collaborative, cross sectoral approaches	- Monitor pollinators (collaboration between farmers, the broader community and pollinator experts)	5.2.4, 5.4.7.3, 6.4.1.1.10, 6.4.4.5, 6.4.6.3.4
		- Increase taxonomic expertise through education, training and technology	6.4.3.5
		- Education and outreach programmes	5.2.4, 6.4.6.3.1

Ambition	Strategy	Examples of responses	Chapter references
		- Manage urban spaces for pollinators and collaborative pathways	6.4.5.1.3
		- Support high-level pollination initiatives and strategies	5.4.7.4, 6.4.1.1.10, 6.4.6.2.2

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USA - California: Food and Agricultural Code Section 29120-29128: http://www.leginfo.ca.gov/cgi-bin/displaycode?section=fac&group=29001-30000&file=29120-29128

USA – California: Food and Agricultural Code Section 29120-29128: http://www.leginfo.ca.gov/cgi-bin/displaycode?section=fac&group=29001-30000&file=29120-29128

USA/Canada: United States Environmental Protection Agency, Health Canada Pest Management Regulatory Agency & California Department of Pesticide Regulation (2014): Guidance for Assessing Pesticide Risks to Bees: http://www.epa.gov/pesticides/science/efed/policy_guidance/team_authors/terrestrial_biology_tech_team/GuidanceAssessingPesticideRisk2Bees.pdf

USA/Canada/Mexico: NAPPO (2008): Guidelines for the Petition for Import and Release of Non-*Apis* Pollinating Insects into NAPPO Countries. RSPM No. 29: http://www.nappo.org/en/data/files/download/PDF/RSPM29-20-10-08-e.pdf

APPENDIX A. Methods and approaches used in this Chapter 6

A1. Defining responses in each sector

Our list of responses was compiled from:

i) suggested responses from published lists related to bee conservation or pollination services (Dicks *et al*. 2010, Sutherland *et al*. 2014);
ii) items listed during a workshop session at the first author's meeting, July 2014; and
iii) a consultation with all authors, the pollination Technical Support Unit and the ILK Task Force.

Responses were then grouped according to policy sectors. The sectors are: a) Agricultural/horticultural/forestry practices; b) Pesticides and other pollutants; c) Nature conservation; d)

Pollinator management and beekeeping; and e) Urban and transport infrastructure. These sectors were selected based on a combination of the Millennium Ecosystem Assessment and the important policy areas selected by an FAO policy workshop on pollinators.

We developed a section on **integrated response** types that could be applied across sectors, such as participatory processes, regional co-ordination of policies or trans-disciplinary research. The application and effects of integrated responses within each sector are still considered within the relevant sectors (for example, regional co-ordination of bumblebee importation policies would be in the managed pollinator section). The integrated response section looks across sectors and describes evidence gathered across sectors that cannot easily fit in the individual sectoral sections.

A2. Review methods

Our search methods followed the protocol outlined in the IPBES guidance document. The following databases were searched: Environmental Evidence Systematic Review Library; ISI Web of Science; Conservation Evidence synopses. Search terms for each sector are shown as in Table A.1. Search terms used for other sections of the chapter (also combined with All row from Table A.1) are in Table A.2.

A3. Examining the chosen responses

In each section we reviewed responses that have been **proposed** in response to evidence of drivers, status and trends in pollinators (see also Chapters 2 and 3). Then we asked which, if any, have been **tested** or are already **established**. Within each sector, responses were grouped according to the type of response (see List of Responses document).

For each chosen response or category of response, we reviewed what is known about its effectiveness at reducing the risks or enhancing the opportunities associated with pollinators and pollinators (see section 6.2).

For the main sectors (section 6.4), information about the effectiveness of each type of response is summarized in a table at the end of each subsection. In these tables, and to accompany summary statements in other parts of our chapter, we have used the confidence terms adopted by this IPBES assessment. The choice of terms has been made by consensus among the Lead and Coordinating Lead Authors of Chapter 6.

Knowledge gaps important for understanding the responses and issues discussed in Chapter 6 were identified by individual lead authors, in response to reviewing the literature. These are brought together in section 6.8. Separately, in section 6.6 we provide an overview of the research and activities that have focused on identifying knowledge needs across the whole of pollinator and pollination science. This is related to a discussion about how research and monitoring needs are being met overall.

49. Table A1. Search terms used for responses in each sector in section 6.5.
In the initial search, terms from all the cells in the ‘All’ row and the appropriate sector row were combined in a single string of search terms, using AND. If no suitable review or synthesis studies were found, subsequent searches were conducted without the ‘Review OR meta-analysis…’ term.

Sectors	Search terms used for responses in each sector		
All	Review OR meta-analysis OR "systematic review"[1]	OUTCOME TERMS: (Pollinat* OR bee OR bees OR Apoid* OR syrphid OR ((butterfl* OR Lepidoptera OR moth OR moths OR beetle* OR Coleoptera OR bird* OR bat OR bats) AND pollinat*))	Option OR policy OR policies OR action OR intervention[2] OR trade-off OR sustainab* OR conserv* OR "ecosystem service" OR benefi* OR "pollinat* serv*"
Agricultural/ horticultural/ forestry	SECTORAL TERMS: agricultur* OR farm* OR farmland OR horticultur* OR crops OR arable OR livestock OR forestry OR Agroforestry OR organic	SECTOR SPECIFIC RESPONSE TERMS: "flower strip" OR "habitat" OR non-ag* OR non-crop OR non-timber OR off-field OR non-tillage OR "no till" OR "reduced tillage" OR "conservation agriculture" OR field margin OR heterogen* OR hedgerow OR crop rotation OR connect* OR meadows OR species-rich OR pasture OR "forest fragment" OR remnant OR Agri-environment* OR Agrienvironment* OR integrated pest management OR IPM OR fertilizer* OR "mass-flowering crop*" OR "variety" OR automatic OR mechanical OR robotic OR certificat* OR extension OR training OR "land abandonment"OR "not-dependent pollinat* crop*" OR "manual pollinat*" OR "manual-pollinat*" OR "mechanical pollinat*" OR "automatic pollinat*" OR "hand pollinat*" OR "hand-pollinat*"	SECTOR SPECIFIC OUTCOME TERMS:
Pesticides and other pollutants	SECTORAL TERMS: pesticid* OR insecticid* OR herbicid* OR algicid* OR molluscicid* OR	SECTOR SPECIFIC RESPONSE TERMS:	

Sectors	Search terms used for responses in each sector		
	miticid* OR rodenticid* OR biocid* OR agrochemical* OR agro-chemical* OR toxic* OR pollut*		
Nature Conservation	SECTORAL TERMS: habitat* OR native veg* OR remnant OR grassland* OR woodland OR wildflower* OR veg*	SECTOR SPECIFIC RESPONSE TERMS: restor* OR manage* OR conserv* OR plant*OR reforest* OR afforest*	
Pollinator management and beekeeping	SECTORAL TERMS: beekeeping OR apicultur* OR “managed bees”		SECTOR SPECIFIC OUTCOME TERMS: Disease* OR varroa OR honey
Urban and transport infrastructure	SECTORAL TERMS: right-of-way or rights-of-way or urban* or road* or electrical* or power* or “transmission line*” or infrastructur* or infra-structur* or transport or garden*		

[1] This term removed and search repeated if no reviews found

[2] This term not used for searching Conservation Evidence synopses, which at present only include evidence relating to policies and actions.

50. Table A2. Search terms for other issues covered in Chapter 6.

All cells from the appropriate row were combined with cells from the All row from Table A1. If no suitable review or synthesis studies were found, subsequent searches were conducted without the ‘Review OR meta-analysis…’ term.

Other issues covered in Chapter 6	Search terms
Risks	(risk OR risks OR opportunit*) AND (“pollination deficit” OR yield* OR quality OR food OR biodiversity OR “farm income” OR “species richness” OR “seed production” OR honey OR “bee product*” OR “cultural value” OR “cultural service*” OR health) NOT (venom OR insecticide)[1]

Tools and methodologies	(“case study” OR model* OR evidence OR InVEST OR “cost benefit analysis” OR CBA OR “cost-benefit” OR “risk assessment” OR “multicriteria analysis” OR “multi-criteria analysis” OR “multicriteria decision analysis” OR “multi-criteria decision analysis” OR “multicriteria evaluation” OR “multi-criteria evaluation” OR MCDA OR MCA OR MCE OR “Vulnerability analysis” OR scenario* OR mitigation OR pathway* OR priorit* OR “natural capital account*” OR map* OR ”decision tree” OR “DSS” OR “Decision support” OR “Participatory Integrated Assessment” OR PIA OR “Ecosystem approach” OR “Environmental Impact Assessment” OR EIA)[2]
Uncertainty	“ecolog* uncert*” OR “ecolog* vagueness” OR “ecolog* ambiguity” OR “uncert* analysis”
Analyzing trade-offs	**Web of Science (Review OR meta-analysis):** (review* OR metaanalysis OR "meta-analysis") AND (pollinat* OR bee OR bees OR Apoid* OR syrphid*) AND (policy OR policies OR action* OR response* OR intervention* OR service* OR conserv* OR sustainb*) AND (trade-off* OR "trade-off*" OR synerg* OR conflict* OR cost* OR benefit*) **Web of Science (Non review OR meta-analysis):** (pollinat* OR bee OR bees OR Apoid* OR syrphid*) AND (policy OR policies OR action* OR response* OR intervention* OR service* OR conserv* OR sustainab*) AND (trade-off* OR "trade-off*") **Google Scholar:** pollination AND policy AND trade-off
Integrated responses	**Web of Science** (“citizen science” AND [TERMS FROM TABLE A1 ROW 1]) **Google** “pollinat* AND research AND (centre OR initiative OR funding)” Search conducted 20 August 2014. First 100 hits examined.

[1] This search was carried out without the general search terms in the top right cell of Table A1.

[2] Underlined terms used in a search with the Review term from Table A1.

Where appropriate, we consulted databases, websites, people and organisations for each section. These sources are listed in Table A3.

51. Table A3. List of organisations, websites and people consulted by each section.

ORGANISATIONS					
Name	**Country**	**Website/URL**	**Contacted person**	**Data/Information obtained**	**Section**
ARIES (Artificial Intelligence for Ecosystem Service) Development Team	USA	http://www.ariesonline.org		No response	6.5
EcoMetrix Solutions Group	USA	www.ecometrixsolutions.com	Michelle Kenna	Details of underlying pollination model	6.5

AfroMaison Technical Team	South Africa	http://www.afromaison.net/	Fonda Lewis	Details of underlying treatment of pollination in model	6.5
WEBSITES					
Title		**Website/URL**		**Data/Information obtained**	**Section**
Mapping and Assessment of Ecosystem Services in Europe		http://biodiversity.europa.eu/maes Accessed 2-Sep-14		One document found	6.5
Ecosystem Services Partnership		http://www.es-partnership.org/esp Accessed 2-Sep-14		No new material found	6.5
PEOPLE					
Name	**Country**	**Affiliation**		**Data/Information obtained**	**Section**
Joachim Maes	Belgium	Leader of European Commission MAES (Mapping and Assessment of Ecosystems and their Services) project		Pollination maps have not been used for policy decisions in Europe yet.	6.5
Anne Teller	Belgium	European Commission		None	6.5
Paul Cross	UK	University of Bangor		Clarified interpretation of Pesticide Toxicity papers	6.4
John Bolte	USA	Lead developer of Envision model		Check that a pollination module from InVEST included in the model.	6.5
Tereza Giannini	Brazil	University of Sao Paulo		Findings on climate change and passion fruit pollinators have not been used by industry.	6.5
Mike Harfoot	UK	United Nations Environment Programme-World Conservation Monitoring Centre (UNEP-WCMC)		Asking if progress incorporating pollination into IAMs.	6.5
Per Rydahl	Denmark			Asking about use of Plant Protection Tool	6.5

Bob Bulmer/Jeremy Macklin	UK	InVivo Agricultural Solutions	Details and use of InVivo farm pollinator resource model	6.5
Virginie Boreux	Germany	Universitat Freiburg	Asked about Sacred grove research	6.4.3
Hisatomo Taki	Japan	Forestry and Forest Products Research Institute	Asked for help on regional (Asian) perspectives - got some new references on Japanese bumble bees	6.4.3
Connal Eardley	South Africa	Agricultural Research council, Plant Protection Institute	Asked about Kenyan taxonomy initiative - got a useful reply	6.4.3
Anton Pauw	South Africa	Stellenbosch University	Asked for help on regional (African) perspectives no reply yet	6.4.3
Ariadna Lopes	Brazil	Universidade Federal de Pernambuco	Asked for help on regional (South American) perspectives - got some new references	6.4.3
Blandina Viana	Brazil	Universidade Federal da Bahia	Asked for help on regional (South American) perspectives - got some new references	6.4.3
Gretchen LeBuhn	USA	San Francisco State University	Asked for examples of citizen science, and outcomes of the Great Sunflower Project.	6.4.3 6.4.6
Sam Droege	USA	United States Geological Survey	Told Chapter 6 about a bee monitoring program in northeast US	6.4.3

Laurie Adams	USA	North American Pollinator Protection Campaign www.pollinator.org	Reports of success or other outcomes, and case study	6.4.6
Celine Geneau	Switzerland	Syngenta	Reports of success or other outcomes, and case study from Operation Pollinator	6.4.6
Gemma Light	UK	Welsh Government	Reports of success or other outcomes, and requested case study from Welsh Pollinator Action Plan	6.4.6
Una Fitzpatrick	Ireland		Reports of success or other outcomes, and requested case study from Irish Pollinators Initiative	6.4.6
Debbie Harding	UK	Biotechnology and Biological Sciences Research Council	Reports on the amount of investment, success or other outcomes of UK Insect Pollinators Initiative	6.4.6
Margaret Heath	Australia	Rural Industries Research and Development Corporation	Reports on the amount of investment, success or other outcomes of Pollination Programme	6.4.6
Christina Grozinger	USA	Pennsylvania State University	Reports on the amount of investment, success or other outcomes of the Center for Pollinator Research	6.4.6

Amina Harris	USA	University of California, Davis		Reports on the amount of investment, success or other outcomes of the Honey and Pollination Centre	6.4.6
Parthib Basu	India	University of Calcutta		Reports on the amount of investment, success or other outcomes of the Centre for Pollination Studies	6.4.6
Norman Carreck	UK	University of Sussex		Reports on the amount of investment, success or other outcomes of the International Bee Research Association	6.4.6
Nicolas Deguines	France			Outcomes of SPIPOLL citizen science project	6.4.6
Gretchen LeBuhn	USA	San Francisco State University		Outcomes of Great Sunflower citizen science project	6.4.6
Emma Krafft	USA	Xerces Society		Evidence of outcomes from pollinator training events	6.4.6
Lynn Dicks	UK	University of Cambridge		Evidence of trade-offs and synergies (Bennett et al. 2009; Dicks et al 2013)	6.7
Tom Breeze	UK	University of Reading		Evidence of trade-offs and synergies (Carvalheiro et al. 2011; Holzschuh et al. 2011; Rucker et al. 2012)	6.7
Carol Poole	South Africa	South African National Biodiversity Institute		Case study on eucalyptus and honeybees in South Africa	6.7

Mike Allsopp	South Africa	Agricultural Research Council		Case study on eucalyptus and honeybees in South Africa	6.7
Brin Hughes	UK	Conservation Grade/Fair to nature		Asking for evidence of effects of Conservation Grade on pollinators. Two MSc thesis and an PhD thesis under development were provided.	6.4.1
ILK (Indigenous and Local Knowledge) Task Force Global Dialogue Workshop	Panama			Workshop attended by Maria del Coro Arizmendi to gather ILK stories for Chapter 6.	6.4
Phil Lyver	New Zealand	The Intergovernmental Platform on Biodiversity and Ecosystem Services - Indigenous and Local Knowledge (ILK) Task Force		Validating text on the experience of using video to pass on biocultural tradition.	6.4.6
Harold van der Valk		Independent		For information on relevant policies and actions to avoid or reduce impacts of pesticides and pollutants on pollination and pollinators	6.4.2
Barbara Ekbom	Sweden	Swedish University of Agricultural Sciences		For information on relevant policies and actions to avoid or reduce impacts of pesticides and pollutants on pollination and pollinators	6.4.2
Daniel Ward	New Zealand	Nature Watch		To check verification process for Nature Watch	6.4.6
Karen Oberhauser	USA	Monarch Larva project		To check details of scheme for Table 6.4.6.3	6.4.6
PP Dhyani	India	Govind Ballabh Pant Institute of Himalayan Environment and Development-EarthWatch Project		To check details of scheme for Table 6.4.6.3	6.4.6

Richard Fox	UK	National Moths Recording Scheme		To check details of scheme for Table 6.4.6.3	6.4.6
Stuart Roberts	UK	Bees, Wasps and Ants Recording Society		To check details of scheme for Table 6.4.6.3	6.4.6
Geoffroy Williams	Switzerland	Institute of Bee Health, University of Bern		To check text on COLOSS and ask for additional information on outputs.	6.4.6