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REVIEW OF POLLINATORS AND POLLINATION RELEVANT TO THE CONSERVATION AND SUSTAINABLE USE OF BIODIVERSITY IN ALL ECOSYSTEMS, BEYOND THEIR ROLE IN AGRICULTURE AND FOOD PRODUCTION

*Note by the Executive Secretary***

I. INTRODUCTION

1. The present report is presented in response to decision XIII/15, paragraph 11, in which the Conference of the Parties to the Convention on Biological Diversity requested the Executive Secretary, subject to the availability of resources, in partnership with relevant organizations and indigenous peoples and local communities, to compile and summarize information on pollinators and pollination relevant to the conservation and sustainable use of biodiversity in all ecosystems, beyond their role in agriculture and food production for consideration by the Subsidiary Body on Scientific, Technical and Technological Advice at a meeting held prior to the fourteenth meeting of the Conference of the Parties.

2. The Assessment on Pollinators, Pollination and Food Production[1] of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) used literature available until July 2015. Since this point in time, a considerable amount of new literature has been made available. A workshop convened from 27-29 November 2017 in collaboration with IPBES, the University of Reading, and the Convention on Biological Diversity brought together regional experts on pollinators to discuss and assess the role of pollinators and pollination services in supporting ecosystems beyond agricultural systems and in supporting ecosystem services beyond food production. This report draws on the contributions of researchers and partners around the world listed in appendix 1. A summary of this report is provided in annex II of document CBD/SBSTTA/22/10.

3. Section II of this document describes roles and values of pollinators and pollinator dependent plants beyond agriculture. Section III presents the status and trends of pollinators and pollinator dependent plants in all ecosystems. Section IV provides the status of the main drivers of shifts in pollinators, pollination and pollinator dependent wild plants in all ecosystems. Finally, the Section V brings some response options for conservation and management of pollinators, their habitats and their food and water resources in all ecosystems.

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* CBD/SBSTTA/22/1.

** Issued without editing.

[1] IPBES (2016). *Assessment Report on Pollinators, Pollination and Food Production.*

II. ROLES AND VALUES OF POLLINATORS AND POLLINATOR DEPENDENT PLANTS BEYOND AGRICULTURE

There is a wide diversity of values linked to pollinators and pollination beyond agriculture and food production. Pollinators and their habitats provide ecological, cultural, financial, health, human, and social values. Pollinators enhance the reproduction and genetic diversity of around 80% of the plant species. More than half of plant species are self-incompatible or dioecious and completely dependent on biotic pollination. These plants are critical for the continued functioning of ecosystems as they provide food, form habitats and provide other resources for a wide range of species. Some examples include mangroves, dominated by obligate outbreeder plants, which provide important services such as preventing coastal erosion, supporting fisheries, protecting from flood and salt intrusion, providing wood fuel and timber, as well as habitat and food provision for bees and many other species (e.g. birds, mudskippers) among others (Mukherjee et al. 2014). Another example are tropical forests, as they contain a high number of dioecious species, contributing to climate regulation, wild meat, malaria and other diseases regulation, fruits and seeds that support many other species in the forest, among other services.

Furthermore, products produced by stingless bees, such as honey, wax, cerume (a mix of wax and propolis), propolis, pollen and the bees themselves, are used by many indigenous people for different purposes, which include nourishment, traditional medicine, activities related to their spiritual and contemplative life and their hand-crafting (Rodrigues 2006).

2.1 Ecological and intrinsic values

Most of the estimated 350,000 angiosperm species, as well as many gymnosperm species, depend on animal pollinators for production of seeds and the maintenance of their populations. Animal-mediated pollination usually leads to some degree of outcrossing and thus promotes and maintains genetic variation in populations, which in turn allows species to adapt to new and changing environments. For these two reasons - ensuring a supply of seed propagules and promoting genetic variation - pollinators are considered to be of fundamental importance for the maintenance of plant diversity and ecosystem functioning (Wilcock and Neiland 2002). However, plant species vary enormously in their degree of dependence on pollinators for seed production, and also in their degree of dependence on seeds for maintenance of populations (Bond 1994).

Pollinators and other flower visitors utilize flowers for food in the form of nectar and pollen, and, in some cases, oils and resins, as well as for shelter and mating rendezvous sites (Simpson and Neff 1983). Some pollinators also use flowers as brood sites (Sakai 2002, Hembry and Althoff 2016). Thus mutualisms between plants and their floral visitors sustain not only plant diversity, but also the diversity of an estimated 350 000 animal species, mainly various insects, birds and mammals (Ollerton 2017). The degree of ecological dependence of these animals on the flowers ranges from completely obligate, as in species that use particular flowers as brood sites or sources of food, to facultative, as in species that have generalist diets that include some food from flowers.

Most generalized species are usually network keystone species because they interact with most plant species and play an important role to maintain the whole network (Gonzalez et al 2010).

Plant dependence on animal pollinators for seed production

The great majority (c. 87.5%) of flowering plants are adapted for pollination by animals (Ollerton et al. 2011), with the remainder of species being either wind-pollinated (Linder 1998, Friedman and Barrett 2009) or completely reliant on autonomous seed production. The degree of ecological dependence of plants on pollen-vectors for seed production depends on their breeding systems. Plants that are dioecious (including some gymnosperms such as cycads) or that are genetically self-incompatible are wholly dependent on cross-pollination for seed production (Richards 1997). It has been estimated that about 50% of angiosperms fall into this category of obligate dependence on pollen vectors (Fryxell 1957, de Nettancourt 1997). The remaining plant species are genetically self-compatible, but these often have an obligate dependence on pollen vectors for seed production on account of spatial or temporal separation of their reproductive parts (Richards 1997). Self-compatible plants that are capable of autonomous self-fertilization often undergo some outcrossing due to pollinator visits. Data from molecular markers show that most self-compatible plants experience mixed-mating and fall along a continuum from selfing to outcrossing (Schemske and Lande 1985, Vogler and Kalisz 2001). Only a very small proportion of plant species produce seeds entirely through autonomous self-fertilization or through non-sexual processes such as apomixis (Moldenke 1979).

Cross-pollination is not only essential for seed production for around half of all plant species, but also results in higher seed production and performance of progeny in many self-compatible species (Aizen and Harder 2007). This is because cross-fertilization reduces the likelihood of inbreeding depression which is commonly observed in the self-fertilized progeny of plants (Darwin 1895, Husband and Schemske 1996, Keller and Waller 2002). Cross-fertilization also promotes the build-up of genetic variation and thus the ability of plant species to adapt to new and changing environments (Jump and Penuelas 2005, Morran et al. 2009).

Studies involving supplemental hand-pollination of flowers have shown that seed production of plants is often limited by the quantity and quality of pollen received naturally (Knight et al. 2005). This phenomenon of pollen limitation of fecundity occurs naturally in relatively undisturbed ecosystems, but is often exacerbated when plants populations become small and fragmented (Ågren 1996, Wilcock and Neiland 2002). It can arise because pollinators are rare or because plants have too few mating partners and pollinators carry inadequate amounts or quality of pollen. Because self-incompatible plants cannot use their own pollen to produce seeds, they are more likely to experience pollen-limitation than self-compatible plants (Larson and Barrett 2000).

The availability of effective pollinators in plant communities is an obvious requirement for successful seed production in most plant species. It is often shown experimentally that selective exclusion of a single group of effective pollinators to which plants are adapted can result in failure of plants to produce fruits or seeds. In a global meta-analysis, Ratto et al. (2018) showed an average 63% loss of fruit or seed production when vertebrate pollinators are excluded from the flowering plants they visit. For plants with generalist pollination systems, a link between pollinator diversity and seed production can also be established (Albrecht et al. 2012)

Demographic dependence of plants on seeds

Plants vary in their dependence on seeds for demographic viability of their populations. Many plants are obligate re-seeders and replace their populations entirely from seeds. Such plants include annuals and plants in fire-prone habitats that are killed by fire. Pollen limitation of seed production in small populations of the annual plant species *Clarkia concinna* (Onagraceae) was associated with reduced population persistence, suggesting links between pollination, seed population and demographic viability in this species (Groom 1998). However, even plants that are not short-lived or killed by fire usually depend on seeds for establishing new populations and for demographic replacement due to age-dependent mortality of adults. The effects of pollination failure on the demographics of long-lived woody species may not be obvious for decades, even though the demographic consequences may be just as serious in the long-term (Janzen 1974). While it is obvious that a complete cessation of seed production would have a negative impact on most plant populations, it is also important to ask whether quantitative variation in seed production due to differing levels of pollination service will affect plant demography (Groom 1998, Lundgren et al. 2016). An older view was that plant recruitment was limited by microsites or by self-thinning processes, such as that variation in propagule supply would have little effect on plant recruitment (Harper 1977). However, recent meta-analyses suggest that the maintenance and establishment of plant populations is often limited by the supply of propagules (Clark et al. 2007).

Experimental augmentation of the number of seeds at sites or introduction of seeds into sites where populations are absent have clearly shown that seed supply is important for plant demography (Turnbull et al. 2000, Clark et al. 2007). Seed limitation of plant demography is most evident for plants in disturbed habitats, for those with large seeds, and for those with short-lived seed banks (Turnbull et al. 2000, Clark et al. 2007). Variation in seed supply is less demographically important where there is a high background density of seedlings or adults, as is often the case for woody species with self-thinning among saplings

(Turnbull et al. 2000). For example, seed predation which reduces the supply of seeds in a manner similar to lack of pollination has been shown to have limited demographic consequences in some shrubs that occur in dense populations with self-thinning of seedlings (Hoffmann and Moran 1991).

Scaling the ecological importance of pollination to the landscape level

If pollinators are ecologically important for plant demography at local scales, we can predict that plant distribution should correlate with the availability of pollinators and that plants that do not depend on pollinators should be better colonizers.

Indirect evidence for the ecological importance of pollinators comes from evidence that pollinator availability can restrain the geographical distribution of plant species. In one study, niche models that include availability of key pollinator mutualists do a better job of explaining actual plant distributions than do niche models that only abiotic factors (Duffy and Johnson 2017). This suggests that pollinators are part of the fundamental ecological niche of certain plant species. This is also supported by translocation studies which show that pollination success of some plants with specialized pollination systems drops close to zero when plants are moved beyond the range of their pollinator species (Waterman et al. 2011). A corollary of this finding is that the geographical ranges of plants which are capable of facultative self-fertilization should not be constrained by pollinator distributions. Indeed, the success of self-fertilizing species as weeds and as colonizers of islands is testimony to the strong role of pollinators as an ecological filter of the distribution of plant species that lack the capacity for uniparental reproduction, including autonomous self-fertilization (Rambuda and Johnson 2004, Van Kleunen et al. 2008, Pannell et al. 2015, Grossenbacher et al. 2017).

Breakdown in mutualisms and extinction

Failure of mutualisms leading to lower seed production and sometimes also extirpation of plant populations have been recorded in several studies (Groom 1998, Wilcock and Neiland 2002). Bond (1994) argued that the overall risk of extinction through mutualism failure could be predicted by the probability of mutualism failure (such as the level of specialization), reproductive dependence on the mutualism (such as the breeding system) and the degree of dependence on seeds (for example, whether a species is an obligate reseeder). Few studies have examined all of these factors together and the failure of mutualisms has generally been assessed in terms of the deleterious effects of habitat fragmentation on plant fecundity.

Mutualism failure does not seem more likely overall to occur in plants with specialized pollination systems than in those with generalized pollination system (Ashworth et al. 2004). It has been suggested that this is due to the asymmetry of pollination systems whereby plants with specialized pollination systems often rely on generalist animals (Ashworth et al. 2004). Another possible explanation is that plants with specialized pollination systems often have compensatory mechanisms that assure seed production, such as facultative self-fertilization (Bond 1994, Fenster and Marten-Rodriguez 2007). Indeed, genetic self-incompatibility has emerged as a predictor of the magnitude of mutualism failure as measured by declines in fecundity (Aguilar et al. 2006, Merrett et al. 2007).

Changes in plant community composition arising from mutualism failure have been documented in a guild of orchids specialized for pollination by an oil-collecting bee in South Africa (Pauw and Bond 2011). Plant species richness in the guild decreases with declining pollination success and the species that persist when pollination fails completely are all clonal, suggesting that non-clonal species are most prone to local extinction through failure of the pollination system (Pauw and Bond 2011).

Ecological reliance of pollinators on plant food resources

A very large number of insect and vertebrate species depend on flowers as a source of food (Ollerton 2017). Their dependence on food from flowers varies from obligate, as applies to almost all bees, hawkmoths, long-proboscid flies, and specialist avian nectarivores such as hummingbirds and sunbirds, to facultative, as applies to many short-tongued flies, beetles, and opportunistic avian nectarivores. Among the animals that have an obligate dependence on flowers for food, only some depend on a specific plant taxon for food. Estimates of the percentage of bees in various communities that are oligolectic (ie. depend on a particular plant taxon, usually a genus, for food) range from 15-60% (Minckley and Roulston 2006). This form of specialization usually involves collection of pollen and univoltine reproduction that is synchronized with flowering of the host plants (Minckley and Roulston 2006). The flowers utilized by oligolectic bees are often morphologically unspecialized and abundant (and often also used and pollinated by many polylectic bees). This may reflect that specialization by bees on rare and ephemeral plant species is unlikely to persist through evolutionary time.

It is obvious that a landscape devoid of flowers, such as a monoculture of cereal crops, will lose many of the animal species that visit flowers for food. A positive relationship between floral abundance and diversity of flower-visiting animals at the habitat scale is now very well established (Potts et al. 2003, Hines and Hendrix 2005, Scheper et al. 2015). However, such relations may in some cases reflect local aggregation of animals and not necessarily a change in population sizes (Roulston and Goodell 2011). Declines of bee species at the countrywide scale have been shown to be correlated with declines in specific host plants (Biesmeijer et al. 2006, Scheper et al. 2014), but it is not easy to establish causality in such large scale correlative studies. In Europe, specialist long-tongued bumblebees have narrower host plant ranges than generalist short-tongued bumblebees and have also declined more rapidly, suggesting that lack of floral resources, particularly of pollen-rich legumes, has been a causal factor in their decline (Goulson and Darvill 2004, Goulson et al. 2005). The most convincing evidence for an effect of floral resources on pollinator populations comes from demographic studies which show that colony growth and reproduction in bees can reflect the availability of floral resources (Crone and Williams 2016, Spiesman et al. 2017). There is some evidence that oligolectic bees are more vulnerable to habitat fragmentation and changes in particular floral resources than are polylectic bees. Detailed demographic studies have shown that population persistence of oligolectic bees can be strongly correlated with availability of particular floral host plants, implying that survival and reproduction of the bees is pollen-limited (Williams and Kremen 2007, Franzen and Nilsson 2010, Palladini and Maron 2014) (See Figure 1).

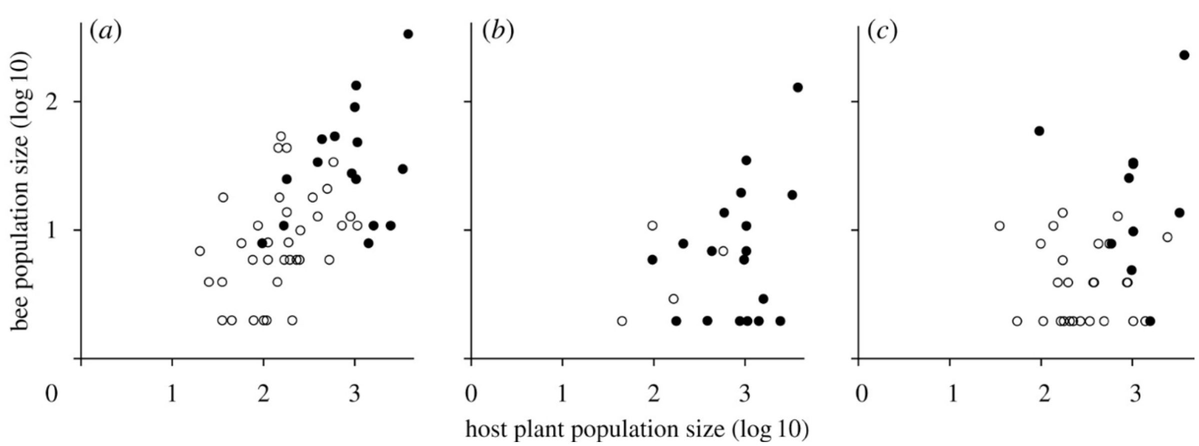

Figure 1. The relationship between local extinctions and the size of local female bee A. hattorfiana populations the year before the extinction events occurred, and the local pollen plant population size (number of K. arvensis stalks), over 3 consecutive years ((a) 2003–2004, (b) 2004–2005 and (c) 2005–2006). Extinct local bee populations are denoted by an open dot and local bee population present both years are denoted by a filled dot. Logistic regression 2003–2004, all variables included in the model: patch area: B = −0.24, p = 0.82; bee population size: B = −2.99, p = 0.035, plant population size: B = −4.58, p = 0.003; bees in surrounding 3 km2 in 2003: B = −0.009, p = 0.24. Logistic regression 2005–2006, all variables included in the model: patch area: B = −2.87, p = 0.15; bee population size: B = −5.39, p = 0.009; plant population size: B = −5.83, p = 0.044; bees in surrounding 3 km2 in 2005: B = −0.06, p = 0.34. Extinctions in 2005 were not analysed statistically owing to few observations
From Franzen and Nilsson 2010

In his discussion of the putative relationship between a Malagasy orchid and giant hawkmoth, Darwin (1862) originally proposed the idea that the extinction of a particular plant species could lead to extinction of a pollinator species that depended on food from its flowers. While there is strong evidence for local extirpation of pollinator populations due to a lack of floral resources (Franzen and Nilsson 2010), we are not yet aware of any cases where an animal species has become globally extinct because of a lack of floral resources. However, given the extent of habitat fragmentation, the large number of plant species that have become extinct or nearly-so in the past 100 years and the paucity of our knowledge about host plant usage by flower-visiting animals, the possibility that this is occurring without being documented is very real. Data on population changes in wild flower-visiting animals are notoriously hard to obtain and the causes of these changes even harder to establish.

2.2 Economic and non-economic values

Some examples of the vast contribution of pollinators to different dimensions of human wellbeing, including economic and non-economic values, will be presented below.

Cultural

Pollinators, pollinator habitats, and pollinator products are sources of inspiration for art, education, literature, music, religion, traditions, and technology. Honey-hunting and beekeeping practices based on indigenous and local knowledge have been documented in more than 50 countries (Crane 1999; Gupta et al. 2014). Bees inspire imagery and texts in religions all over the world, including the three-bee motif of Pope Urban VIII, the Surat An-Naĥl in the Qur'an, and sacred passages within Buddhism, Hinduism, and Chinese traditions such as the Chuang Tzu (Potts et al. 2016). Many of the fruits, berries, and other non-cultivated plants that we enjoy in gardens, parks, and semi-natural habitats depend on wild pollinators for their propagation. Indeed, home gardens provide a large set of ecosystem services, being cultural services the category most valued, including aesthetic information, recreation and tourism, inspiration for culture, art and design, and spiritual experience (Calvet-Mir, Gómez-Baggethun & Reyes-García 2012). Pollinators such as hummingbirds (the national symbol of Jamaica) and sunbirds (the national symbol of Singapore) contribute directly to identity (IPBES 2016) and also indirectly, for example, by supporting aesthetically important flowers in landscapes (Junge et al. 2015). Furthermore, many sites listed under the Convention Concerning the Protection of the World Cultural and Natural Heritage depend on pollination to maintain their values, while the Convention for the Safeguarding of the Intangible Cultural Heritage recognizes several practices that rely on pollinator-dependent plants as globally important (IPBES 2016; Potts et al. 2016).

Pollinators play an important role in the life of indigenous peoples and in their environment. The key principles for Māori[2] people in relation to the pollinators and the pollination process are described in the Box 1.

Financial

Bee products, such as honey, propolis, and beeswax contribute to increased income to beekeepers around the globe. Beekeeping is critical for local development as it typically requires minimal investment, generates diverse products, can occur without land ownership or rent, and provide flexibility in timing and locations of activities (Hilmi, Bradbear & Mejia 2011). Moreover, pollinator-dependent plants contribute to many material and non-material services with important financial value such as ornamentals, medicines, biofuels, fibres, construction materials, musical instruments, arts, crafts, and recreation activities (Potts et al. 2016). For example, in the Nilgiri Biosphere Reserve in India, it was found that 40% of plants that provide non-timber forest products, including construction materials and medicine, benefit to some extent from biotic pollination (Rehel et al. 2009).

Health

Several medicinal plant species depend on pollinators for their reproduction. Pollinator-dependent plants also recycle CO_2, regulate climate, and improve air and water quality. Furthermore, several micronutrients, including vitamins A and C, calcium, fluoride and folic acid are obtained primarily from pollinator-dependent plants (Smith et al. 2015). Also, pollinator products (e.g. honey, propolis, bee sting) are employed for improving health, such as anti-bacterial, anti-fungal, and anti-diabetic agents derived from honey (Jull et al. 2015). Pollinator insects, including the larvae of bees, beetles, moths, and palm

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[2] Māori are the indigenous Polynesian people of New Zealand

weevils constitute an important proportion of the approximately 2,000 insect species consumed globally, being high in protein, vitamins, and minerals (IPBES 2016).

Human

Beekeeping provides jobs for hundreds of thousands of families all over the world (Hilmi et al. 2011; Gupta et al. 2014; Johannsmeier 2001). Beekeeping can also form the basis for gaining and transmitting knowledge about ecological processes (IPBES 2016). Furthermore, understanding of flowers and pollinators is part of the knowledge base for indigenous people and local communities, for example, in some regions, flowering phenology provides indications for decisions regarding weather predictions. Pollinators and pollinator-dependent plants support technological and knowledge advances through inspiration and application of their biology to human innovations, such as the visually guided flight of robots (IPBES 2016).

Social

Beekeeping associations can enhance social bonds among beekeepers that facilitate cooperative actions and social connections, having potential to increase or exchange knowledge, improve livelihoods, and long-term resilience (Garibaldi et al. 2016). Furthermore, beekeeping can be a potentially effective intervention tool for reducing relapses in youth criminal behaviour, and for empowering youth to link biodiversity, culture and society and take action on issues of environmental impacts on pollinators and pollination (IPBES 2016). Pollinators are also critical for the subsistence of many home gardens, which contribute to the creation and maintenance of social relations (Calvet-Mir et al. 2012).

As it has been shown, pollinators provide a full range of benefits, including cultural (e.g. identity), financial (e.g., honey sales), health (e.g. pharmaceutical properties of bee products), human (e.g., employments in beekeeping), and social (e.g., beekeepers associations) dimensions.

BOX 1. New Zealand – Māori Pollination values, impacts and policy (Brad G. Howlett and Phillip O'B. Lyver)

The connection and relevance of Māori to pollinators and the pollination process can be understood through the interacting principles of *whenua* (through its dual definition of land and placenta), *tātai whakapapa* (connectedness of genealogy), *mauri* (life force or essence) and *mana* (authority and prestige). The first principle of 'whenua' in its dual context connects a person to their land, and in doing so defines their identity and place in the world (Doherty and Tumarae-Teka 2015; Timoti et al., 2017). Whenua is considered as the placenta for life and is born from the womb of *Papatūānuku* (Mother Earth; Royal 2012a). For this, when the placenta and *pito* (umbilical cord) of a Māori newborn is buried on their lands it signifies their attachment and relationship to Papatūānuku. The principle of 'tātai whakapapa' is recognition of connectedness in the world and the genealogical links between humans and the elements of the spiritual and natural world, including the mountains, oceans, rivers and the residing flora and fauna (Harmsworth and Awatere 2013). These elements all have 'mauri' or a life essence that binds and animates them in the physical world (Royal 2012b). Within an ecosystem the presence of mauri defines its health and potential to sustain life (Marsden 2003; Timoti et al. 2017). For Māori, to be the *kaitiaki* (guardians) of a healthy natural environment from which food and resources can be easily procured is an important source of 'mana' for the people. For Tūhoe Tuawhenua Māori within the Te Urewera region of New Zealand, the fundamental role of pollinators in underpinning the health and function of their forests and providing food has been widely understood through the generations.

Bird pollinators and bird pollinated plants: Native birds are well recognized both by Maori and other researchers as key pollinators of many native plant species. These included the kōparapara (bellbird, *Anthornis melanura*), tūī *(Prosthemadera novaezelandiae*), kererū (*Hemiphaga novaeseelandiae*), hihi (stitchbird, *Notiomystis cincta*), kākā (*Nestor meridionalis*) and tieke (saddleback, *Philestrunus carunculatus*) (Craig et al. 1981; Clout and Hay 1989). Bird pollinated plants include kōwhai (*Sophora* spp.), rātā (*Metrosideros* spp.) which was used by Māori to treat skin diseases and wounds, while the pūriri (*Vitex lucens)* was used for ulcers and were sometimes boiled and used to remedy backache (Brooker et al., 1987). Harakeke (*Phormium tenax*) was an important source of *muka* (fibre) to make cloth and rope (Carr et al., 2005). Wild growing and cultivated flax grown in a *pā harakeke* (a site for the

planting of selected varieties of harakeke[3] based on their muka or *raranga* (weaving/plaiting) qualities) had a number of uses including clothing, cordage, fishing nets, bird snares and baskets (Murray 1836; Best 1942; McAllum 2005). Beyond the value that bird-pollination brought to the forests plants many of the bird pollinators were hunted by Māori for food and feathers (Best 1942; Lyver et al. 2008) and were fundamental to cultural identity and heritage values expressed through language, knowledge systems, and customary management regimes (Timoti et al., 2017) . The skin and feathers of bird pollinators were often used for making garments such as *korowai* (traditional cloaks; Harwood, 2011). Feathers from the kererū, kākā, and kākāriki (parakeet, *Cyanoramphus* spp.) frequently adorned korowai in the early 19th Century (as listed by Harwood, 2011).
Biological and cultural impact of bird declines: Since European settlement (1800)[4] in NZ, predation by introduced mammalian predators such as rats (e.g. Ship rat, *Rattus rattus*), feral cats (*Felis catus*), and mustelids (e.g. stoats, *Mustela erminea*) and habitat degradation have significantly reduced the populations of native bird species, such as the kererū and kōkō (Kelly et al., 2010; García et al., 2014). Biocultural indicators such as flock size and harvest tallies of kererū and kōkō in Te Urewera have been used by Tuawhenua over the last century to monitor the declines of these two species (Lyver et al., 2017). The large-scale declines of cultural (and ecological) keystone native birds around NZ has had a significant impact on the cultural identity and heritage of many Maori tribes. The losses affect cultural diversity as much as biological diversity of New Zealand.

Policies for the inclusion of Maori cultural knowledge and values and knowledge on pollinators and pollinated plants: The role of Maori values and *mātauranga* (indigenous knowledge) in the management and conservation native ecosystems has becoming increasingly recognised over the past three decades in NZ (Taiepa et al. 1997; Awatere and Harmsworth 2014) and governmental agencies are increasingly mandating for its inclusion in decision-making (Ministry for the Environment, and Statistics New Zealand. 2015). For example, the Te Kawa o Te Urewera and Whirinaki Te Pua-a-Tāne Conservation Management Plans incorporate the principles of kaitiakitanga (customary guardianship regimes), whanaungatanga (inter-relationships where all things live and interact as whanau (family)), tuakana–teina (positive and supportive relationships), he tangata (people as an essential part of the ecosystem) and mauri reflecting the life essence of ecosystems and tribes (Te Uru Taumatua 2017; Ngati Whare and Department of Conservation 2017). The arrangement gives effect to the treaty of Waitangi (ensuring consultation with iwi regarding management including access to sites and resources, establishing protocols for tree removal and pest control, regeneration, collection and propagation of seeds and seedlings, concessions to use cultural information identify business opportunities that are consistent with natural, historical and cultural heritage values (Ngati Whare and Department of Conservation 2017).

III. STATUS AND TRENDS OF POLLINATORS AND POLLINATOR DEPENDENT PLANTS IN ALL ECOSYSTEMS

3.1 Global status and trends of pollinators, their habitats and biomes

A lack of global Red List assessments specifically for insect pollinators and, in most parts of the world, the lack of long term population data or benchmark data to compare the present status of wild pollinator populations makes it difficult to discern any temporal trend.

Evidence of changes in species richness or abundance along a habitat degradation gradient can be a proxy for temporal changes if the more pristine areas in the gradient today are seen as what the degraded areas used to be in the past. In a global meta-analysis, Aguilar et al. 2006 found that local declines in pollinator abundance and diversity were linked to decreasing trends in wild plant pollination and seed production in habitat fragments. Morales and Traveset (2009) evaluated data from 40 global studies on 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. Montero-Castaño and Vila (2012), using 143 studies, showed that habitat alteration and

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[3] http://www.landcareresearch.co.nz/resources/collections/harakeke/establishing-a-pa-harakeke

[4] http://www.teara.govt.nz/en/maori/page-3

invasions affected pollinators by decreasing visitation rates. Visitation rates by vertebrates in altered landscapes and by insects (excluding bees) in invaded areas were the most affected.

While it has not yet been possible to carry out a comprehensive global monitoring of pollinator or pollination trends, two global databases lend themselves to visualizing some key global patterns. The first of this is a map depicting the Red List status of vertebrate pollinators globally (Figure 2). Vertebrate pollinators of course are much more amenable to tracking their population numbers, and thus their status does not represent pollinators in general, but are nonetheless indicators of regions where some pollinator groups and their floral resources are under different levels of risk.

The second is a world map showing the annual growth rate in the number of honey bee colonies and honey production for countries reporting those data to FAO between 1961 and 2012 (Figure 3). Managed bee colonies can pollinate both crop plants and also wild plants, and thus their trends are relevant for this report, although causes in each case are complex.

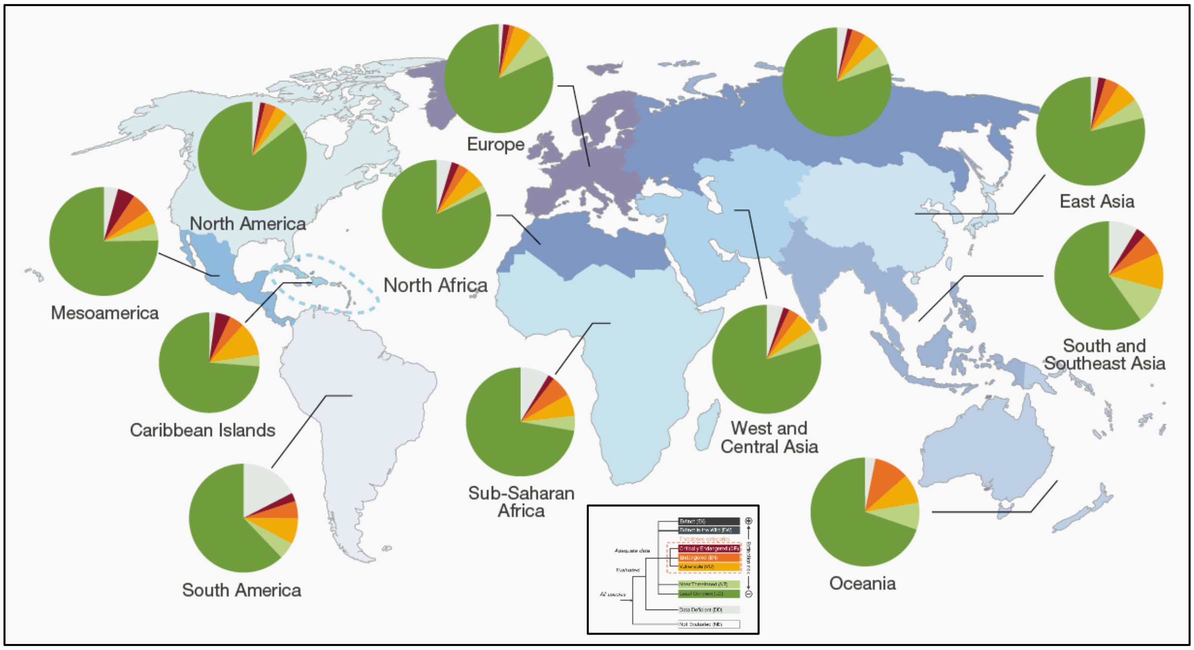

Figure 2. The International Union for Conservation of Nature (IUCN) Red List status of vertebrate pollinators (including mammals and birds) across IUCN regions. IUCN relative risk categories: EW = Extinct in the wild; CR = Critically Endangered; EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern; DD = Data Deficient; NE = Not Evaluated.

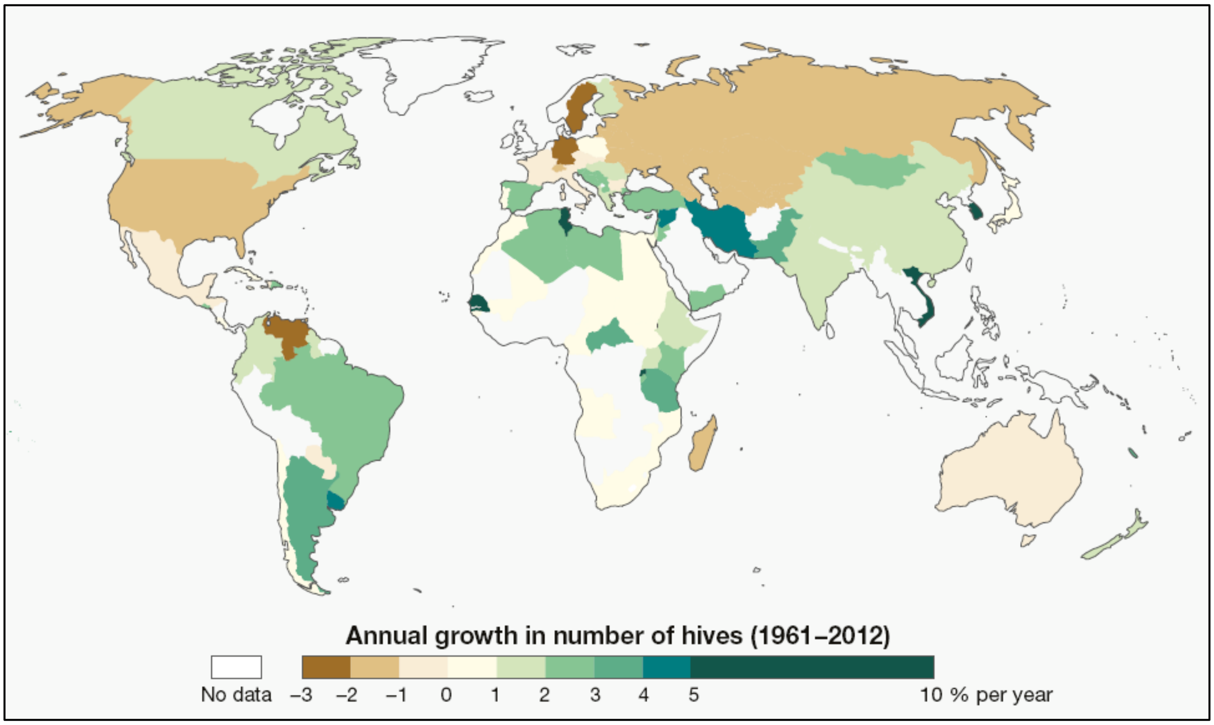

Figure 3. World map showing the annual growth rate (% change per year) in the number of honey bee colonies and honey production for countries reporting those data to FAO between 1961 and 2012 (IPBES 2016b).

3.2 Regional status and trends of pollinators, their habitats and biomes

Current knowledge on regional status and trends of pollinators is detailed in each of the four sections below. In each section, what is known of wild pollinators and managed pollinators are both featured, recognizing that managed pollinators also have important roles beyond agriculture, foraging in natural and semi-natural habitats.

For this report, the following regions are considered: Africa, Asia-Pacific, Latin America, and West Europe and Others Group and their respective biomes[5].

3.2.1 Africa Region

Habitats and biomes vulnerable to pollinator declines

Tropical forest: Pollination dependence is poorly documented in African tropical forests. High vulnerability might be expected, however, given that for lowland forests globally, >98% of flowering species are animal pollinated, with effective visitors limited to one or two classes of visitor, usually from the same order (Bawa 1990).

Dry Deciduous Forest: A number of Madagascan forest species are pollinated by lemurs (Sussman and Raven 1978), many of which are threatened, suggesting vulnerability of the system.

Subtropical forest: Most trees are insect pollinated (Griffiths et al. 2006), so declines in insects would be detrimental.

Medditeranean: Cape Floristic Region, South Africa: High vulnerability – many asymmetrical interactions – i.e. many specialized species reliant on a single species or functional group of pollinator (Johnson 2010; Pauw and Stanway 2015).

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[5] The biomes used in this report follows the IPBES category

Mountain grasslands: Pollination systems in mountain (C4) grasslands are quite highly specialized (Johnson et al. 2009).

Tropical and subtropical savannas and grasslands: Some groups have quite marked reliance on one or a couple of species. For example, Convolvulus hawkmoth, Agrius convolvuli is the most important pollinator of African plants with very long-tubed (>8 cm long) flowers. More than 70 grassland and savanna plant species in Africa belong to the Agrius pollination guild (Johnson and Raguso 2016).

Drylands and deserts: In the semi-arid areas of South Africa (Namaqualand), bees are prone to reproductive failure during drought (Mayer and Kuhlmann 2004). Many of the dominant daisies that appear to be generalists actually require specialist pollinators (Bombyllid flies) (Ellis and Johnson 2009; De Jager and Ellis 2014). Monkey beetles (Scarabaeidae: Hopliini) carry high pollen loads, so are likely important pollinators in some systems, e.g. Succulent Karoo, monkey beetle community composition has been found to change in response to heavy grazing (Mayer et al. 2006).

Wetlands and dambos: Often habitats for orchids with specialized pollinators (Burgoyne et al. 2000; Johnson 2006), breeding sites for some important pollinators (this is a knowledge gap).

Urban and peri-urban: Vulnerable – decline in rare butterflies associated with urbanization in South Africa (Mecenero et al. 2015), decline in visitation to bird-pollinated Erica perspicua near roads (Geerts and Pauw 2011). There is a loss of bird pollinator functional diversity along a gradient of increasing urbanization (Pauw and Louw 2012).

Coastal areas: Many mangrove species are bird or insect pollinated (Noske 1993; Okoth 2010).

The status and trends of pollinators and pollination

Wild pollinators: A lack of spatial and temporal changes in wild pollinators in Africa, combined with poorly-known taxonomy hampers assessment of the status and trends of Africa's pollinators, although some ongoing initiatives may help address this issue[6]. Certain trends can be predicted from studies that substitute space for time, however.

Bees *(Apidae)*: Data sets on the status and trends of bees in the continent are not yet available. Some work has been conducted on how bees might be affected by climate change: in a region with high bee endemism in South Africa, range contractions are expected for half of the 12 species considered (Kuhlmann et al. 2012).

Butterflies and moths *(Lepidoptera):* Since the late 1980s, South Africa has produced three red data assessments of the approximately 800 species of butterfly found in the country. A comparison of the two most recent assessments found an increase in severity of threat status of Lepidoptera. In the 2009 assessment, 60 (7.5%) of the total 801 species were considered threatened (i.e., vulnerable, endangered or critically endangered), and in the 2013 assessment (Mecenero et al. 2013), although this number was unchanged (Edge and Mecenero 2015), the severity of threat has increased. Six species new to science were considered to be under immediate threat from ongoing habitat destruction, suggesting that species may go extinct before being described (Henning et al. 2009). Furthermore, the 2013 assessment considered three species to be extinct (Mecenero et al. 2013). When rare or very rare species are also considered, 151 species (18.8% - one in every six Lepidoptera) are of high conservation concern in the region (Edge and Mecenero 2015). The IUCN red list[7] for Lepidoptera for the whole of Africa includes red list assessments for only 355 species. Many of the species listed in the IUCN red list are also included in the South African assessment, including two of the three species listed as extinct for Africa. The third species listed as extinct for Africa is the nymphalid *Libythea cinyras*, which was endemic to Mauritius. Excluding the extinct species, 96 (27.3%) are threatened or near threatened, and almost 12% (42 species) are data deficient. Some of the data are also dated: one quarter of the species assessed in the IUCN redlist were last assessed over two decades ago, the remainders were assessed during or after 2011. Population

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[6] http://jrsbiodiversity.org/jrs-awards-two-grants-study-africas-wild-pollinators/ and https://www.sanbi.org/biogaps

[7] http://www.iucnredlist.org/

trends for the African Lepidoptera indicated that 10% were stable, 9% were decreasing and trends for the remaining 81% were unknown.

Pollinating birds: A number of bird families in Africa may be instrumental in pollination, including Zosteropidae and Icteridae, but their effectiveness in this role may be fairly minor. Therefore, we consider here the sunbirds (Nectariniidae) and sugarbirds (Promeropidae). The IUCN lists 11 (12%) of the 93 sunbird species included in their data base as threatened or near-threatened; there are only two species of sugarbird, and one of these (*Promerops gurneyi*) is considered near threatened. For sunbirds, 22% are considered to be decreasing, the remainders are considered stable, and none are increasing. Of the sugarbirds, one species is decreasing, the other is stable. The two Southern African Bird Atlas Projects (SABAP1, 1987-1991; and SABAP2, 2007-2017) allows comparison of reporting rates over the two different time periods. This comparison shows that 15 out of the 23 species (i.e., 65%), show declines over time, while the remainder showed increases. As with many other species, habitat transformation is largely responsible for these losses.

***Pteropodid* bats**: The role of bats in pollination in Africa is poorly studied. The IUCN red list provides assessments for 42 species of the Pteropodidae in Africa, the bats most likely to be pollinators. Thirty of these are found on the mainland and 12 on islands. The threat status of the island species is more severe than that of the continental species: 11 of the 12 island species are threatened compared to three of the 30 mainland species, although five of the mainland species are considered data deficient. Almost half (45%) of island species and 27% of mainland species are declining, but we lack data for 40% of the continental species, so proportion of species in decline could be much greater. None of the mainland species seem to be increasing, although two (18%) island species appear to be increasing.

Managed pollinators - honey bees (*Apis mellifera*): In a study assessing managed honey bees colonies in South Africa, losses were almost 30% in 2009-2010 and 46.2% in 2010-2011 (Pirk et al. 2014). None of the beekeepers surveyed indicated that these losses were threatening their livelihoods (Pirk et al. 2014), possibly because colonies are replaceable by capturing wild colonies. As honey bees are native to Africa, their health and ability to reproduce in swarms is key both to their survival and their use in beekeeping and pollination. A survey of the health of Africa’s pollinators, carried out in 2016, suggested that aside from an outbreak of American foulbrood in the Western Cape, South Africa, large scale colony losses had not been observed on the continent (Pirk et al. 2016).

A recent study suggests that although deformed wing virus has a long evolutionary history with *A. mellifera,* the parasitic mite, *Varroa destructor*, spread by human transport of colonies from Europe and North America to other regions, makes the disease far more virulent (Wilfert et al. 2016). Africa has been invaded by *Varroa destructor* since turn of century (i.e., 1990s – 2000s, Allsopp 2004; Wilfert et al. 2016), so its full effect on disease virulence may not yet be fully manifest, making it a future concern for Africa, and necessitating close monitoring. In Kenya, where only remote apiaries were free of *Varroa destructor,* remote colonies were also free of Deformed Wing Virus (DWV), Acute Bee Paralysis Virus (ABPV) and Black Queen Cell Virus (BQCV) (Muli et al. 2014). There was also a significant correlation between viral diversity and *Varroa* loads (Muli et al. 2014).

Apart from the relatively recent introduction of *Varroa* to Africa, part of the African honeybees’ resistance to disease may lie in their large wild populations (Dietemann et al. 2009), and their high genetic diversity (Wallberg et al. 2014), which along with their hygienic behaviour (i.e., finding and removing parasites from hives; Frazier et al. 2009) may give them some immunity to diseases.

Figure 4 depicts the current status of documentation of the presence of the three main pathogen groups (bacteria, fungi and viruses) associated with honeybees, as reported by national governments in Africa.

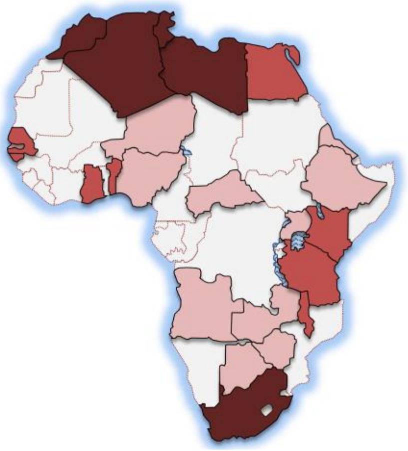

Figure 4. Distribution of pathogen groups (i.e., bacteria, fungi and viruses) associated with honey bees (from Pirk et al. 2016). White = none or no data, Pink = one of these pathogens present; Red = two of these pathogens present; Maroon = three of these pathogens present.

Managed pollinators - Stingless bees: A survey of stingless bee species in Tanzania found them to be in relatively low densities, attributed to threats from fire, honey hunting and logging activities (Hamisi 2016).

Managed pollinators - Bumble bees *(Bombus spp.)*: As yet, bumble bees have not been introduced to Africa and it is advisable to avoid such introductions, as they can introduce diseases, and have been associated with declines of native *Bombus* species in South America (Schmid-Hempel et al. 2014) as well as reduced native plant seed and fruit set (Kenta et al. 2007).

Wild plants dependent on pollinators in Africa: In many African ecosystems, there is a high degree of pollinator-plant specialisation, particularly in the global biodiversity hotspots. For example, the majority of species in the Orchidaceae and Iridaceae of southern Africa are pollinated by a single species, and plants adapted to pollination by oil-collecting bees, long-proboscid flies and the *Aeropetes* butterfly are usually pollinated by a single species (Johnson and Steiner 2003), or some plants are pollinated by a small group of species (Johnson and Steiner 2003). In the semi-arid Succulent Karoo, even daisies, which phenotypically may appear very generalised, often have specialist pollinators (e.g., Ellis and Johnson 2009; De Jager and Ellis 2014). Such specialisation suggests a marked susceptibility to pollinator loss (e.g., Steiner and Whitehead 1996). This susceptibility has been demonstrated in studies where habitat fragmentation adversely impacts pollinator species, and has a striking effect on plant seed set, particularly those species dependent on particular pollinator species (Donaldson et al. 2002; Pauw 2007).

Very few datasets exist to enable comparison of current with historical pollination rates. An elegant study comparing century-old herbarium specimens with more current specimens in highly fragmented habitat in an urban matrix found the rate of flower visitation to the orchid *Pterygodium catholicum* had declined significantly post 1950 and was associated with a decline in abundance of this and other non-clonal species (Pauw and Hawkins 2011).

Plant-pollinator networks: Plant pollinator networks confirm high specialisation in some systems, e.g., the Cape Floristic Region (Pauw and Stanway 2015). At various sites in Africa, flower visitor network studies have confirmed the value of natural vegetation to agriculture in supporting agricultural pollinators (Hagen and Kraemer 2010; Simba et al. 2018), and that alien invasion in grassland fragments can reduce modularity of networks (Hansen et al. 2018). Within grasslands and savannas in Kenya and South Africa, hundreds of long-tubed flowered species have become adapted for pollination by a single abundant species, the convolvulus hawkmoth, *Agrius convolvuli* (Johnson and Raguso 2016; Johnson et al. 2017). Reliance on a single species of pollinator is potentially risky in the face of global change.

Around the globe, anthropogenic disturbance has been found to simplify pollination networks, and Africa is no exception. Pollination networks for heath habitats in Madagascar in which hand-weeding of invasive species had helped restore native flora over 20 years had almost twice as many flowers as found at a non-restored sites, and a plant pollinator web with a visitation rate by flower visitors about 1.8 times that of the restored site, suggesting far greater network complexity at the restored site (Kaiser-Bunbury et al. 2009). Environmentally-friendly agricultural practices have been found to reduce or even circumvent network homogenisation (Kehinde and Samways 2014).

3.2.2 Asia and Pacific Region

Habitats and biomes vulnerable to pollinator declines

Tropical dry evergreen forests (TDEF) are a unique forest type found along the east coast of India. They mostly occur as small isolated fragments of varying sizes (0.5 to ≈10 ha) and are considered as endangered forests types in peninsular India. Nayak and Davidar (2010) studied the reproductive biology of the woody species of this forest. Of the 14 woody species found in these forests four species were self-incompatible and six species were self-compatible. The majority of plants species (85%) had a generalized pollination system, receiving visits from diverse insects, such as social bees, solitary bees wasps, moths and flies of whom *Apis cerana*, *Apis dorsata*, *Braunsapis picitarius* and *Trigona* sp. Seemed to be the important pollinators. Two species namely: *Capparis brevispina* and *C. zeylanica* had butterflies and birds, respectively, as their main pollinators. Selwyn and Parthasarathy (2006) reported bees to be the most important pollinators in this type of forest.

In a long term study spanning 10 years Devy and Davidar (2003) recorded the pollination system of 86 tree species (80% of total arborescent trees) in a mid-elevation wet evergreen forest in the Indian Western Ghats, a region rich in endemic flora and is a biodiversity hotspot. Approximately 75% of these species were specialized to a single pollinator group such as bee, beetle, or moth revealing the vulnerability of the reproductive system.

Status and trends of pollinators and pollination

Present Status and Temporal changes in pollinators: Only one species of Leaf-cutter (Megachilid) bees from the entire Asia Pacific region (*Megachile pluto*) from Indonesia has a mention in the IUCN Red data list (marked as vulnerable). In comparison, butterflies of this region are relatively better assessed by IUCN. Of 270 species of butterflies that have been reported in IUCN Red Data list, 3 species have been reported to be Critically Endangered, 21 species to be Endangered, 30 species to be Near Threatened and 42 species to be vulnerable. However, nothing is known about the status of thousands of other butterfly species that are reported from the region.

In an attempt at identifying population trend of the pollinating insects Smith et al. (2017) reported a decline of a number of wild pollinators in Eastern India based on farmers' perception. Partap and Verma (2000) observed decline of native populations of *A. cerana* in the Himalayan region of Nepal. Verma (1993) and ICIMOD (1994), citing Crane (1992), reported the onset of local extinctions of *A. cerana* in the hilly areas of Bhutan, Afganistan, Bangladesh, China, Myanmar, Pakistan and to some extent in India. A study in Japan reports disappearance of endemic small bees from an archipelago due to introduced lizard predation (Abe 2006). In another study Xiong et al. (2013) reported pollinator scarcity to be the reason for delayed selfing of Himalayan mayapple (*Podophyllum hexandrum*) in China. But apart from these few studies no other assessment reporting any temporal change in wild pollinator species populations is available from this region save some general observations regarding threats to the bee fauna (Krishnan et al 2012; Teichroew et al. 2017). The sole available study on the trend of genetic diversity change of a single species is by Nagamitsu et al. (2016) that show loss of genetic diversity of the Japanese sub species of *A. cerana – A. cerana japonica*.

South-West Pacific islands have been reported to have a depauperate bee fauna and an even smaller endemic bee fauna Groom et al. (2015). However, a recent study by da Silva et al. (2016) reported current status of *Braunsapis puangensis*, an introduced species in Papua New Guinea and found the species to be effectively contributing to the existing plant species in the island and therefore seeming to have a positive impact. Lee et al. (2016) reported on the diversity of the stingless bees (Meliponinae) for the first time from Cambodia and Laos.

Wild invertebrate pollinators: Basu et al. (2016) found areas with more natural vegetation or forested areas to have greater bee diversity in a region adjacent to the Eastern Himalayan biodiversity hotspot. In another study Thomas et al. (2009) reported wet forests of the Western Ghats to have more diversity of social bees than relatively degraded and drier forest sites. Bee abundance, particularly belonging to Apidae, had higher abundance in large primary forests compared to the degraded forests in Malaysia and Singapore (Liow et al 2001). Wild bee diversity was reported to be significantly higher in primary forests in a changing tropical landscape in southern Yunnan, south-west China (Meng et al. 2012), although the same study reported hover fly diversity to be higher in early successional forests than the primary forests. Plantations of higher tree diversity have been reported to host higher nest densities of honey bees in Sumatra (Salmah et al 2014). Agricultural areas close to forested areas have been found to be more species diverse for bees (Krishnan et al. 2012; Klein et al. 2006; Klein et al. 2009) and wasps (Klein et al. 2006). On the other hand the reverse trend has also been observed where opening up of landscape from forested areas to open land or agroforestry in Sulawesi, Indonesia increased the diversity of bees due to greater availability of bee-associated herbs in open areas (Hoehn et al. 2010).

Managed Bees - *Apis mellifera:* the European Honey bee and *A. cerana* are the two major managed bees in Asia–Pacific region apart from *Trigona* spp. that is managed for honey production in some parts of the region. Aizen et al. (2009) reported an overall rise in the global stock of managed honeybees but apart from China - that followed the trend, there was no specific mention of other countries in the Asia–Pacific region in the report. Kohsaka et al. (2017) reports falls in honey production from managed bees in Japan and South Korea although the same study reports gradual increase in number of bee keepers and bee colonies in Japan. According to the same study, managed colonies of *A. cerana* have declined significantly both in S. Korea and Japan. Xie (2011) reported nearly 3 fold increase in managed honey colonies and honey production in China during the past 50 years. However, Chen et al. (2017) reported *A. cerana* colony losses in China. Similar colony losses was also reported from Japan for *A. mellifera* (Gutierrez 2009, Taniguchi 2012). An overall increase in beekeeping has been reported in Malaysia (Ismail 2016, Harun et al. 2017) and the practice of stingless bee (*Trigona itama* and *T. thorasica*) keeping has also been reported to be on the rise (Ismail 2016). Melipone bee culture has been reported to be on the rise in Thailand too (Chuttong et al. 2015).

Vertebrate pollinators*: Pteropus hypomelanus*, a nectarivorous bat known as pollinator of Durian (*Durio zibethinus*) in Malaysia has been reported to be locally endangered (Aziz et al. 2017). Among other reports apart from population trends include specific associations of bat pollinators with *Parkia* spp. in Thailand (Bumrungsri et al. 2009) and *Mucuna* sp. in China (Kobayashi et al. 2016). The same study reports squirrel pollination of the same plant in Taiwan. Japanese macaque (*Macaca fuscata*), and the Japanese marten, (*Martes melampus*) have been reported to be the effective pollinators of *Mucuna macrocarpa* found in Japan (Kobayashi et al. 2015). As reported in Devy and Davidar (2003) birds seemed to play an important role in pollinating the epiphytic plants belonging to Loranthaceae and many ground herbs in mid elevation wet evergreen forests in the Indian Western Ghats. A fruit bat, *Eonycteris spelaea*, has been identified as the legitimate pollinator of late-successional tree, *Oroxylum indicum* (Bignoniaceae) in Thailand (Srithongchuay et al. 2008). Yoshikawa and Isagi (2014) reported recording 24 terrestrial bird species including three exotic bird species to act as pollinators in Japan. The same study reported 4 flower feeding bird species. Liu et al. (2002) reported long-tongued fruit bats (*Macroglossus sobrinus*) and sunbirds (*Arachnothera longirostris*) to be the major pollinators of *Musa itinerans* (Musaceae), a pioneer species in tropical rain forests of Yunnan, China.

Wild plants: A rich body of information exists on pollination biology of specific wild plants from across Asia–Pacific Region. A number of studies have highlighted pollination biology of plants of conservation concerns that attract remarkably **high diversity of pollinators**. A number of species of beetles and bees – *Trigona* sp. in particular, were reported to be potential pollinators of eight species of *Amorphophalus* found in the Northern Western Ghats that are endemic to this region (Punekar and Kumaran, 2009). Murali (1993) reported several species of Carpenter bees to be the pollinators of *Casia fistula*, a common understory tree found across India. *Pittosporum dasycaulon*, a self-incompatible rare medicinal plant found in the Indian Western Ghats is pollinated chiefly by *Apis cerana* and *Apis dorsata* (Gopalakrishnan and Thomas, 2014). *Myristica dactyloides* is a vulnerable swamp species found in the Western Ghats and has been reported to be pollinated by a range of insects - thysanopterans (thrips), coleopterans (beetles), halictid bees and dipterans (syrphid and phorid flies) (Sharma and Shivanna, 2011). *Terminalia pallida* Brandis (Combretaceae), an endemic and medicinal tree species of India is pollinated by *A. dorsata* and wasps (Raju et al. 2012). A recent review (Ollerton et al., 2017) has listed various Dipteran species that are responsible for pollination of *Cercopagia* spp. found across various countries of the Asia – Pacific region. *A. dorsata* and *Nomia* spp. seem to be the effective pollinators of *Decaspermum parviflorum* (Lam.) Am Ja Scott (Myrtaceae) a cryptically dioecious plant reported from Sulawesi, Indonesia (Kevan and Lack 1985). *Eomecon chionantha* Hance (Papaveraceae), an endemic species in China has been reported to be most effectively pollinated by *A. cerana, Bombus atripes and Episyrphus* sp. (Syrphidae: Diptera) (Xiao et al. 2016). *Trillium tschonoskii* Maxim. (Trilliaceae), a wide ranging plant species reported from Japan to the Korean archipelago and Taiwan reportedly has a diverse set of pollinating insects including Scatophagiae (Diptera), *Oedemera lucidicollis* (Coleoptera) and *Andrena* sp. (Hymenoptera). *Gastrodia elata*, a mycoheterotrophic Japanese orchid has been reported to be pollinated by a number of insects including 20 insect species belonging to Dermaptera, Thysanoptera, Diptera, Lepidoptera and 10 species of bees (Suguira et al. 2017). Sugiura (2012) reported a number of insect taxa including 9 species of bees, 3 species of Diptera and 1 species each of Lepidoptera and Coleoptera to be the potential pollinators of 2 species of understory *Rhodendron* in secondary forests in Central Japan.

Report on **pollination of ecologically important plant**s include Raju and Rao (2016a) who described *Pavetta tomentosa*, a keystone shrub species found in the Southern Eastern Ghat forests to be pollinated most effectively by Papillinoid and Pierid butterflies and the Sphingid moths. Fiala et al. (2011) studied the pollination system of pioneer species of *Macaranga* in Malaysian rain forests and found thrips and heteropterans to be the most effective pollinators. *Schima superba* Gardn. et Champ. is a perennial, evergreen tree valued for its eco-protection and commercial values in China. *Apis cerana* and two species of beetles (*Protaetia brevitarsis* Lewis, and *Popillia mutans* Newman) has been identified as the most effective pollinators of this tree (Yang et al. 2017).

Pollination biology of some economically important plants outside of crops have been reported. *Callistemon citrinus*, an ornamental Australian plant with high nectar content was reported to be visited by a wide array of animals including insects, birds and squirrels. While *Apis dorsata* was found to be the dominant potential pollinators, sunbirds, parrots, oriental white-eye sparrow and squirrels were reported to be occasional pollinators (Chauhan et al. 2017). Deka et al. (2011) reports bumble bee (*Bombus haemorrhoidalis* Smith) to be the only effective pollinator of large cardamom (*Amomum subulatum* Roxb.) in the Western Ghats, India. Rao and Raju (2002) reported *A. dorsata* to be the only effective pollinator of Red Sanders (*Pterocarpus santalinus*), an endangered tree species endemic to Indian Eastern Ghats. *Aquilaria* spp. (Thymelaeaceae), an important commercial tree in Indonesia and the principal source of Gaharu, a valuable resin, was reported to be essentially dependent upon cross pollination and is visited by 20 different insect species (Sohertono and Newton, 2001). Ghazoul and McLeash (2001) reported *Trigona* spp. as the sole effective pollinator of self-incompatible *Shorea siamensis*, an important commercial timber found in Thailand.

A number of studies also report **noteworthy pollination syndromes or system**. Ruchisansakun et al. (2017) described a new and endangered species in Myanmar *Impatiens tanintharyiensis* (Balsaminaceae) to have a bee pollination syndrome. Koshy et al. (2001) reports bee visits in bamboo flowers in the Western Ghats in India and infers possible bee pollination. *Pedicularis chamissonis*, a nectarless plant from Japan has been reported to be self-incompatible and is pollinated effectively by buzz pollinating *Bombus hypocrita sapporoensis (*Kawai and Kudo, 2009). Plants of the genus *Mucuna* (Fabaceae) are vine plants that are distributed primarily in tropical and subtropical Regions. *Mucuna* macrocarpa, a species found in the northern limit of its distribution – in Japanese archipelago, has been reported to be pollinated by the Japanese macaque (*Macaca fuscata*), and the Japanese marten, (*Martes melampus*) who force open the flowers (Kobayashi et al. 2015). Kawai and Kudo (2009) report buzz pollination by bumble bees of the Japanese plant *Pedicularis chamissonis*.

Plant pollinator network: A few studies are available from this region that report plant pollinator interaction networks. In a study conducted in a bio-diverse meadow in South-West China, Fang and Huang (2016) looked at temporal network stability and found the network to be stable across years. In another study Zhang and He (2017) documented the plant pollinator network of a sub-tropical forest in China. In this network comprising 84 pollinator species and 28 plant species (9 dioecious and 19 monomorphic) plant reproductive systems seemed to influence the network structure, in that dioecious plants were reported to have more generalized pollinators. Chakrabarty et al. (2016) investigated a nocturnal pollen transport network focused on hawkmoths and found the hawkmoths to predominantly associate with non-crop plants in the agrarian landscape. Yoshikawa and Isagi (2014) is the only available study reporting bird pollinator – plant network in Japan. Abe (2006) reported domination and expansion of introduced honeybees have the potential for disruption of the native pollination network in the two main, and several satellite, islands of the Ogasawara Islands in Japan.

3.2.3 Latin America Region

Habitats and biomes vulnerable to pollinator declines

Pollinator-dependence floras prevail across ecoregions in Latin America (Ollerton 2011). It has also been identified that intensive agriculture is the most important cause of local bee decline in this region (De Palma 2016), probably followed by alien bee invasions (Aizen 2018). Therefore, the biomes likely to be experiencing the highest rates of disruptions of plant-pollinator interactions are those undergoing the highest destruction rates due to, a large extent, soybean expansion (Altieri & Pengue 2006). These include the subtropical Chaco forest, the Cerrado savannah, the Pantanal wetland, and the Amazonian forest (e.g. Fearnside 2001, Grau et al 2005, Pacheco 2006).

Increasing soybean cultivation has also impacted long-time highly modified biomes like the Pampas grassland, where much agricultural diversity has been lost due to the raising dominance of this crop (Aizen et al 2009). Another highly-menaced biome rich in plant-pollinator mutualisms is the Atlantic forest of which only ~10% remains (Ribeiro et al. 2009). The extreme fragmentation of this biome has implied a differential loss of plant species with relatively-specialized pollination and sexual systems that only survive in the interior of large remnants (Lopes et al. 2009). Impacts of extensive deforestation of cloud forests along the Andes and Mesoamerican mountains (e.g. Dirzo and Garcia 1992) as well as destruction and degradation of high-altitude Paramo vegetation (e.g. Podwojewski et al. 2002) on plant-pollinator interactions can be predicted but need to be studied. Finally, bumblebee invasion has been seen to disrupt plant-pollinator interactions in the temperate forests of South America (Aizen et al. 2008), increasing flower damage and causing the decline of the Patagonian giant bumble-bee Bombus dahlbomii (reviewed in Aizen et al. 2018, see Box 2).

Bee surveys have been conducted in Brazil since the end of the 1960s and these observations became a standardized methodology to perform bee surveys and allowing comparisons between the different localities and biomes (Sagakami et al 1967). In Brazil, the Africanized honey bees (*Apis mellifera scutellata)* are present and well adapted to urban areas, forests borders and open vegetation. However, in the Amazon rainforest the Africanized honey bees are rare (M.L. Oliveira; Silva, 2005) and the reasons are unclear. Native bees has resisted to the competition with Africanized honey bees as they are used to compete against other native aggressive species (Zanella, 1999; Kerr *et al.* 2001), but still there are concerns about the installation and management of Apis colonies in natural areas and reserves. On one hand, the introduction of Africanized honey bees helped to increase the honey production in Brazil from 5000 ton/year to 40000 ton/year (Gonçalves, 1994). On another hand, it is still unclear the impacts of their competition with native bees, their relation with plants in natural environment and the impact on the reproduction of native plants (Silveira et al. 2002). Some negative impacts were observed on perturbed forests or forests with open vegetation. In campos rupestres[8] the Africanized bees cause the reduction in the production of seeds of Clusia arrudae (pollinized by Eufriesea nigrohirta) because they rarely visit the

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[8] Campos rupestres is an ecoregion of the montane subtropical savanna biome, located in eastern Brazil. It is situated within the South American Atlantic Forest, and borders the Cerrado subtropical savanna ecoregion. https://www.worldwildlife.org/ecoregions/nt0703

male flowers of this plant (Carmo & Franceschinele 2000). For Vellozia leptopetala e V. Epidendroides, the Africanized honey bees also promoted negative effects by reducing the number of seeds due to the competition with other native pollinators (Jacobi 2002).

Status and trends of pollinators and pollination

Wild insect pollinators: Most of the Neotropics occurs within the boundaries of Latin America. This biogeographical region hosts one of the highest species diversity in the entire world, including plant and insect pollinator diversity, due to its extensive tropical biomes, abrupt climatic gradients, high landscape heterogeneity, and complex geological history (Morrone 2006, Kreft & Jetz 2007, Freitas et al. 2009). For instance, just a study from one site in Colombia reports close to 300 different bee species (Smith-Pardo & Gonzalez 2007), a figure that has not been so far surpassed by any other study in any other continent (de Palma et al. 2016). At a more continental scale, some typical Boreal bee groups like the bumblebees (*Bombus*) are poorly represented in the Latin America (Williams 1998). However, this region is extremely species-rich in highly diversified groups of tropical bees like the stingless bees (Meliponini) (Rasmussen & Cameron 2009) and the exclusively Neotropical and charismatic orchid bees (Euglossini) (Cameron 2004). Although long-term studies are needed to establish some clear trends, this biodiversity, and that of plant and pollinators in particular, is threatened by a combination of large-scale drivers, including land clearing and conversion to cattle pasture and extensive monocultures, pesticide and herbicide use, climate change, and invasive bees (Bradshaw 2009, IPBES 2016). Regarding the latter, two major bee invasions have been recorded in this region. In the 1950's , the African honey bee, *Apis mellifera scutellata*, was introduced into Brazil to produce, colonies more resistant to tropical conditions through hybridization with European subspecies. However, the Africanized honey bee rapidly escaped confinement and in <50 years invaded most of tropical and subtropical America, from central Argentina to southern USA (Moritz et al. 2005). Nowadays the Africanized honey bee dominates most flower insect-visiting assemblages in a high variety of biomes, (reviewed in Morales et al. 2017). Recorded in more detail is the invasion of southern South America by two European bumble bees, long-tongued *Bombus ruderatus* and short-tongued *B. terrestris*, introduced into Chile for crop pollination in 1982 and 1997, respectively, and the related decline of the giant Patagonian bumble bee *B. dahlbomii* (see Box 2). The present distribution of *B. terrestris* now extends from Tierra del Fuego to northern Chile and probably southern Perú (reviewed in Aizen et al. 2018).

Box 2. Uncoordinated trade policies facilitate bumblebee invasion in South America

Chile has allowed continuous importation of alien bumble bees to pollinate agricultural crops. Since 1997, this policy has authorized the importation of more than a million colonies of *Bombus terrestris* fom Europe. During 2015 alone, more than 200,000 colonies and queens were imported. Unfortunately, bumble bees are mobile and do not respect international boundaries, even those established along major geographic barriers. As a consequence, this alien species has spread widely in Chile and Argentina, and is now on the verge of entering Bolivia and Peru. The invasion of Argentina across the Andes and its unintended consequences has occurred despite Argentina having banned importation of non-native bumble bees (Aizen et al. 2018).

The most serious biological impact of this invasion is the decline of *Bombus dahlbomii*, the only native bumble bee in southern South America and one of the world's largest bumble bees (See Figure 5). The alien invaders, *Bombus ruderatus*, another European bumble bee that was brought from New Zealand to Chile during 1982 and 1983, and especially *Bombus terrestris*, are potent competitors and carry foreign bee diseases (Arbetman et al. 2013, Schmid-Hempel et al. 2014). As they spread, *Bombus dahlbomii* disappeared from much of Chile and Argentina (Morales et al. 2013, Aizen et al. 2018). The demise of *Bombus dahlbomii* is so severe that it is now recognized in Chile and internationally as an endangered species (http://gefespeciesinvasoras.cl/abejorro-europeo-o-abejorro-comun/, Morales et al. 2016).

The invasion of *Bombus terrestris* has also caused detrimental effects on native and crop plants. To access nectar, this bee damages flowers of many plant species (nectar robbing), reducing nectar for other flower visitors, but often not pollinating flowers effectively (Combs 2011). Nectar robbing and other flower damage caused by *Bombus terrestris* in commercial raspberry fields reduces fruit quality and might compromise honey production by honey bees (Sáez et al. 2014, 2017). Invasion by *Bombus terrestris* also promotes the spread of alien plants, which compete with native species (Morales et al. 2014). For example, in Argentina pollination by *Bombus terrestris* increases seed production and establishment of scotch broom, a pernicious plant invader. The environmental costs of this invasion should alert governments about the convenience of importing alien bumble bees or any other pollinator. Despite these costs, Chile still allows unabated importation of this alien bumble bee.

A retrospective lesson of the *Bombus terrestris* case is that coordinated risk assessment and cautious implementation are essential components of regional and global policy development to avoid transnational species invasions. In particular, policies concerning the importation of potentially invasive species must be established regionally among neighboring countries with suitable habitat. Unilateral investment and effort will be futile if the countries involved adopt conflicting policies. The South American example also calls for more strict international regulations of the bumble bee trade, regulations that should involve not only the importing but also the exporting countries.

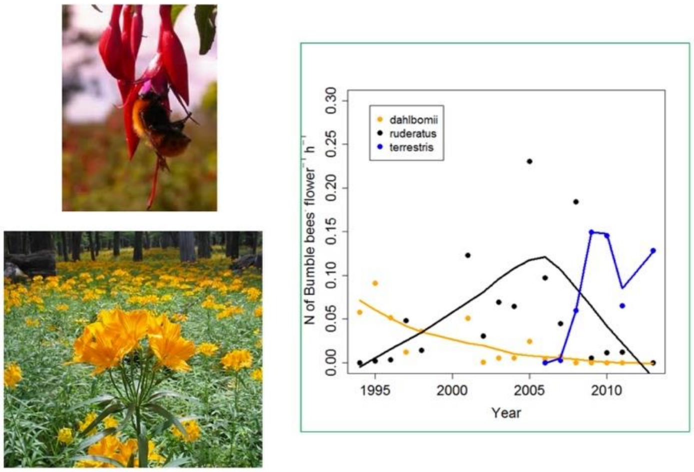


Figure 5. Visitation to flowers of Alstroemeria aurea (low, left panel) changes over time amongst one native bumble bee (B. dahlbomiii) (upper, left panel) and two introduced bumble bees (B. ruderatus and B. terrestis). The right panel is reproduced from Morales et al. 2013.

Vertebrate pollinators: The most diversified and emblematic group of vertebrate pollinators in Latin America are the hummingbirds (family Trochilidae) that includes about 240 species of relatively specialized pollinators (Bleiweiss 1998), followed by the New world glossophagine bats (family Phyllostomidae) that includes close to 40 nectarivorous species (Fleming 2009). Although a comprehensive database to evaluate invertebrate pollinator decline is still missing for Latin America, it is estimated that 12 and 8% of the vertebrate pollinator fauna or South and Meso America are threatened with extinction, respectively (Aslan et al. 2013). This list includes several species of hummingbirds, whose decline could cause the demographic collapse of many plant species that coevolved with them (Martin-González et al. 2015).

Managed pollinators: Despite the introduction of alien bees, there has being an incipient trend in Latin America to rear native bee species for commercial purposes. This is case of several species of stingless bees for honey production and/or pollination services in Brazil, Paraguay, Bolivia, Perú, Venezuela, Costa Rica, and México (Cortopassi-Laurino et al. 2006). Also, several species of large bees have started to being reared to provide pollination services for specific crops. This list includes *Bombus epiphiatus* in México (Vergara & Fonseca-Buendía 2012) and *Bombus atratus* in Argentina and Colombia mostly for greenhouse crops (Velozo 2013, Cruz et al. 2017), and solitary bees in the genera *Xylocopa* and *Centris* for pollination of passion fruit and acerola in Brazil (Oliveira & Schlindwein, 2009, Junqueira et al. 2013).

Managed pollinators: The introduction of Apis species in mangroves

The introduction of Apis species in mangroves to produce honey and beebread has been explored in many countries such as India, the United States, China, and Cuba (Piñeiro 1989, Krishnamurthy 1990, Yao et al. 2006, Singh & Kar 2011). Although mangrove vegetation is present in a considerable area of Brazil, covering 13,000 km^2 of the Brazilian coast from North to South (Spalding et al. 2010), mangrove apicultural activities have not been significantly explored yet. Assessing the potential contribution of the mangrove vegetation to apiculture as well as the impacts on the ecosystems is fundamental as apiculture in forests has the potential to be developed as a conservationist activity with many associated benefits (Luz and Barth 2012).

Managed pollinators: Meliponiculture

The main social pollinators of tropical areas are the stingless bees, the Meliponini. They live in tropical and subtropical areas of the world. In Latin America, the Moure catalog of Neotropical bees mentions 417 species, with many others yet to be described; more than 500 species have been described worldwide. They have perennial colonies and are generalists for food collection (Kleinert et al. 2013). They also do not have functional sting, but can defend themselves using resin, biting and some species produce caustic substances that burns the enemies (Michener, 2013). They generally have large distribution areas and common traits that allow local management, despite having diversity in size, behaviour and preferences for resources utilization.

Beekeeping with stingless bees is called meliponiculture (Nogueira-Neto 1953; 1997; Cortopassi-Laurino et al. 2006; Jaffé et al. 2016, Vit et al, 2015; Heard, 2016). Meliponiculture is a global opportunity for tropical countries, and for small holders farming. They are useful pollinators for crops and also for wild fruits (Heard 1999; Slaa et al. 2006; Nunes Silva et al. 2013; Giannini et al. 2015; Venturieri et al. 2013; Freitas et al. 2017; Campbell et al. 2018). Most of them produce a tasteful honey that is more liquid than that of honey bees; it was first honey known and used by traditional population in Americas. The stingless bee honey is used much more as medicine than as a sweetener (Cortopassi-Laurino et al. 2006)

Beekeeping with stingless bees was originally seen as an indigenous activity and curiosity for most people. Mayan Indians and others in Central America have been known for keeping *Melipona beecheii* in natural trunks. The beekeeping of *Scaptotrigona* over history has also been part of the traditions coming from ancestors of indigenous communities. Stingless bees practices have been extensively documented: the Levin-Strauss anthropological studies on Brazilian indigenous peoples; the Kayapó people in Amazon (Posey & Camargo 1985, Camargo and Posey 1990); the Guarani (Rodrigues 2006), the Enawene-Nawe people (Santos and Antonini 2008), the Pankararé people (Costa Neto 1998, Costa Neto 1999, Castro et al. 2017).

For those peoples, their holistic view considers bees as part of the ecosystem and there are no forests without bees or vice versa. These studies showed the diverse and rich knowledge that different indigenous peoples have about bees and wasps, with more than 25 ethno species divided into these two groups of insects. Their traditional knowledge go beyond the taxonomic system: they consider a range of aspects which include morphologic, ethological, ecological and social characteristics (description, distribution, nest building, seasonality, dispersion, practical aspects of handling and manipulation for the extraction of products, preservation and semi-domestication of species, and the use of their products) (Rodrigues 2006). Such indigenous knowledge has supported science to clarify some species (Nogueira Neto 1997) as some species of stingless bees have their popular and scientific names based on indigenous origins. For instance, *Melipona mandaçaia* (*manda = guard, sai = pretty*) describes a behavior found in that species which have always a guard at the hive entrance.

Another example is the observations of the Enawene-Nawe people in the western region of Brazil. They suggest a mutualism between the stingless bees *(Scaptotrigona sp.*) and the harpy eagle (harpya harpija). This bee is attracted to excrement and preys that remain on this rapine bird and the bees visit the beak, the nostril and the cloaca of it to nourish themselves (Santos and Antonini, 2008). This observations go along with the publication of Levi-Strauss, which includes the observation made by the naturalist Henry Bates suggesting that stingless bees from the Amazon obtain less of their nutrition from flowers and more of it from tree sap and from bird excrement.

Meliponiculture is generally undertaken by indigenous and local communities, and particularities can be observed according to regional and traditional knowledge. For instance, when the Kayapós open natural nests to collect food, some brood, pollen and honey is left for the spirit and this allows recovery of the colony. The Kayapós use smoke for managing aggressive bees, such as Oxytrigona, and also use a common toxic liana (Tanaecium nocturnum, Bignoniaceae) to manage the bees, which causes bees to sleep after 1 or 2 minutes. All knowledge on bees is passed orally through generations. Large-scale rearing of stingless bees is a current challenge as it must be aligned with sustainable development.

Modern meliponiculture in Brazil began with Nogueira Neto (1953; 1970; 1997), and now is spread to all continents (Vit et al, 2013). A very modern and technically developed meliponiculture arose recently (Menezes et al. 2013; Venturieri et al. 2013 for a review). Nests are obtained with trap nests instead of through cutting down trees (Oliveira et al. 2013), colonies are multiplied with established beekeeping practices. Techniques to allow laboratory *in vitro* queen production are available (Menezes et al. 2013). Research on meliponiculture has included many experiments concerning pesticides effects on bees such as that of Rosa et al. 2016, and has supported programs of large scale production of local bees. New developments in landscape genomics are showing how important is to recognize the diverse populations adapted to different environmental conditions (as shown in studies with *Melipona subnitida* in Brazil, in prep.)

Stingless bees are very diverse in size and habits, which allows them to pollinate a large diversity of plant species. Very small bees as *Plebeia minima* and *Aparatrigona*, for instance, pollinate cupuassu, (*Theobroma grandiflorum*), a fruit from Amazon forest of great local importance (Venturieri, 2014). A range of insect species visit and pollinate Assaí palm (*Euterpe oleracea*), currently one of the most important fruits in the Amazon (Campbell et al. 2018), with a growing commercial importance. Large bees from the *Melipona* genus "buzz" pollinate (a specialized form of pollination for certain crops) and are as effective as *Bombus* in the pollination of eggplants (Nunes-Silva et al. 2013). Their habitat tends to be forested land, and as they are generalists, their role in pollination is both within and beyond agriculture. A big gap remains, however, in South Hemisphere knowledge on the breeding of these bees and their ecological importance, although their honey is appreciated and with high values in Latin America countries (see Vit, Pedro and Roubik 2013).

Wild plant pollination

Pollinator dependence is high in most floras of this region, including not only floras from tropical biomes (Ollerton et al. 2011), but also floras from higher latitudes (Aizen & Ezcurra 1998) and altitudes (Arroyo et al. 1983). Characteristic of many floras of the region is the relatively high proportion of hummingbird-pollinated plant species (Stiles 1978, del Coro Arizmendi & Ornelas 1990, Aizen & Ezcurra 1998, Buzato et al. 2000). Bat-pollination can be also relatively common in some neotropical floras, encompassing species in about 160 genera across the Neotropics, twice as many genera as in the Paleotropics (Fleming et al. 2009). Even one case of rodent pollination has been documented in the Andes (Cocucci & Sersic 1998). Some relatively common plant-pollinator interactions in the Neotropics involve fragance-seeking bees, such as those existing between many orchids and male euglossine bees (Ramírez et al. 2011), and oil collecting-bees, including species in the genera *Centris* and *Chalepogenus* and plants in the genus *Calceolaria* and family Malpighiaceae (Ramussen & Olesen 2000). Differential loss of plant species with relatively-specialized pollination and sexual systems has been documented in highly fragmented biomes like the Atlantic forest (Lopes et al. 2009) and it has been suggested that an increase in selfing could be associated with the invasion of Africanized bees (Aizen & Feinsinger 1994a,b)

Wild relatives of crops: Latin America hosts the wild germplasm of many food crops that directly or indirectly depend on pollinators for high yield (Klein et al. 2007). These crops include potato, tomato, pepper, cacao, strawberry, quinoa, amaranto, avocado, sweet potato, acai, palmito, Brazil nut, guarana, passion fruit, and yuca, among others. Germplasm of these, and perhaps of hundreds of wild species with agricultural potential, persists in remnants of natural and seminatural habitats (Smith et al. 1992), and at the hands of local indigenous communities in this region (e.g. Zimmerer 1991). Therefore, diverse pollinator assemblages are important to ensure not only the reproduction of wild plants in general, but also the persistence of this germplasm. Yet, perhaps with a few exceptions, the occurrence and diversity of this germplasm and its current conservation status is unknown.

Plant pollinator networks

A diversity of studies looking at changes in the structure and dynamics of plant-pollinator networks along different environmental gradients have been conducted in Latin America. Relevant findings include a loss of network connectivity along an elevation gradient in the Andes of central Chile (Ramos-Jiliberto et al. 2010) and with decreasing habitat size in a system of isolated hills in the Argentine Pampas (Sabatino et al. 2010). In this latter system, plant-pollinator interactions are non-randomly lost, the most vulnerable being those interactions characterized as specialized and rare (Aizen et al. 2012). A similar result has been recently reported in a long-term study of a plant-pollinator network from the Monte desert (Chacoff et al. 2018). At larger geographical scales, higher biotic specialization with decreasing latitude has been reported in plant-hummingbird pollination webs (Dalsgaard et al. 2011), a trend that replicates a global pattern in plant-pollinator interactions (Olesen & Jordano 2002). Relatively specialized neotropical mutualisms, like those existing between Malpighiaceae flowers and their oil-collecting bees might be highly persistent because of exhibiting more cohesive, and presumably resilient, webs than the whole plant-pollinator network in which these sub-webs are immersed (Bezerra et al. 2009). A network approach has also been followed to infer patterns of coevolution between orchids and orchid bees (Euglossini) across de Neotropics, finding that orchid diversification has apparently tracked the diversification of orchid bees leading to asymmetric specialization (Rámirez et al. 2011). Analysis of the structure of plant-pollinator network from the temperate forests of the southern Andes showed that mutualism disruption can be associated with the introduction of ungulates (Vázquez and Simberloff 2003) and invasive plants and bees (Aizen et al. 2008).

3.2.4 Western European and Others Group (WEOG) Region

Habitats and biomes vulnerable to pollinator declines

The European Red List of Habitats (Janssen et al. 2016) reports the highest percentage of threatened types (categories Critically Endangered, Endangered, Vulnerable) in the European Union (EU28) was found amongst mires and bogs (85%), followed by grasslands (53%), heathland (29%), and scrub (14%), all of which are considered important habitats for pollinators and contain flowering plants dependent upon biotic pollination. Vogiatzakis et al. (2014) used an expert elucidation process to assign quality scores to European Corrine land use classes for wild bees; the habitats with the highest ranked quality were mixed forest, sclerophylous vegetation, natural grassland, moors and heathland. Senapathi et al. (2015) analysed land cover changes in 14 sites across in England over more than 80 years, and found significant effects on richness and composition of bee and wasp species, with changes in edge habitats between major land classes also having a key influence. Sites surrounded primarily by arable expansion showed a greater decline in species richness than sites that did not, and sites surrounded by landscapes with urban expansion have proportionally lost fewer species.

Koh et al. (2013) estimated the shifts in the relative abundance of wild bees between 2008 and 2013 across the United States using a spatial habitat model, national land-cover data, and expert knowledge. High likelihoods of declines were associated with areas where corn and grain cropland mostly replaced grasslands and pasture; high likelihoods of increases were associated with areas where grasslands, pastures, and corn/soy fields were converted to higher-quality habitat, such as shrublands or fallow crop fields.

Data for New Zealand and Australia are largely limited to studies assessing impacts of pollinator decline on the fitness of specific plant species (e.g. reduced pollination rates as mentioned earlier) or risks to pollinators and associated habitats due to environmental change (e.g. land clearance or climate change). In Australia, ongoing land clearance across multiple habitat types threaten a wide range of animal species (Reside et al. 2017). The resulting habitat fragmentation also impacts the pollination of plants depending on the size and shape of fragments (Cunningham 2000) while pollinator abundance and distribution can also be impacted (Eby 2016).

Status and trends of pollinators and pollination

Wild invertebrate pollinators: Wild invertebrate pollinators in the Western European and Other Groups (WEOG) region include bees, butterflies, moths, wasps, beetles and other insects. Bees are considered the most important wild pollinators through much of WEOG, with an estimated 1,965 species in Europe, 3,622 in North America (USA and Canada), and 1,687 in Australia and New Zealand (Nieto et al. 2014; Discover Life's bee species guide and world checklist). Regional and national assessments indicate high levels of threat for some bees and butterflies. In Europe, 9% of bee (Nieto et al. 2014) and 9% of butterfly (van Swaay et al. 2010) species are threatened and populations are declining for 37% of bees and 31% of butterflies (excluding data deficient species, which includes 57% of bees). 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.

Many wild bees and butterflies have been declining in abundance, occurrence and diversity at local and regional scales in North-West Europe (Biesmeijer et al. 2006; Carvalheiro et al. 2013; Ollerton et al. 2014) and North America (Cameron et al. 2011; Bartomeus et al. 2013; Burkle et al. 2013); data for New Zealand and Australia are currently insufficient to draw general conclusions.

Bees (Apidae): Kerr et al. (2015) analysed long-term shifts in bumble bee species ranges in Europe and North America and found consistent trends of 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. Some species have declined severely, such as Franklin's bumble bee (*Bombus franklini*) in the western United States of America (Thorp 2005) and the great yellow bumble bee (*Bombus distinguendus*) in Europe (Bommarco et al. 2012).

Butterflies and moths (Lepidoptera): Diversity and abundance of Lepidoptera species have declined in the United Kingdom (Fox 2013, Thomas et al. 2004). In North America, almost 60 species are Red Listed by the Xerces Society as Vulnerable, Imperiled or Critically Imperiled, and there is particular concern about the iconic monarch butterfly.

Other invertebrates: Trends for other wild pollinator species are variable (e.g. European hoverflies, Keil et al. 2011), unknown or are only known for a small part of the species' distribution.

Managed pollinators: The main managed pollinators in Europe, North America and Australia and New Zealand are western honeybees (*Apis mellifera*) and bumblebees (*Bombus terrestris* in Europe and New Zealand, and *B. impatiens* in N. America), although other bee species are or have been managed such as *Megachile*, *Osmia*, *Nomia* and stingless bees (Australia only), but to a much lesser extent.

FAO data show that the number of managed western honey bee hives is increasing at the global scale over the last 50 years (IPBES 2016b), however contrasting trends (increases, decreases, no change) are found among countries within Europe, North America and Australia/New Zealand (**Figure 2**). Between 2005 and 2015 there have been severe declines in some European countries (Potts et al. 2010) and seasonal colony loss is high in some European countries and in North America (Neumann and Carreck 2010).

Information on current numbers of other managed pollinators traded annually is problematic because such information is often withheld by rearing companies and there is no obligation to report commercially sensitive information. However, an estimated two million *Bombus* colonies were traded annually across the world for pollination in 2006 (Velthuis and van Doorn, 2006) and an estimated 800 million alfalfa leafcutter bees are traded commercially per year in North America (Reisen et al. 2009). While trends in western honeybees are captured in national statistics, trends at the sub-national level, and for other managed pollinators, are poorly understood.

Vertebrate pollinators: There are also vertebrate pollinators in North America (e.g. hummingbirds), Australia (e.g. some birds, marsupial, rodents and bats) and New Zealand (birds, one bat). There are no vertebrate pollinators reported in Europe. An IUCN Red List assessment shows that 16.5 per cent of vertebrate pollinators are threatened with global extinction (**Figure 1**), with similar proportions documented for North America and Europe, but higher proportions for Oceania (Aslan et al. 2013).

Wild plants and plant-pollinator networks: Biesmeijer et al. (2006) found a decline since 1980 of outcrossing plant species that rely on pollinators, when compared to self-compatible or wind-pollinated plants, in Britain and the Netherlands. This decline was strongly correlated with (but not necessarily caused by) loss of bee diversity. Burkle et al. (2013) analysed changes in plant-pollinator interactions over 120 years in Illinois, USA, and found only 50% of the bee species originally recorded in the late 1800s by Charles Robertson remained; although novel interactions had arisen, the total number of interactions in the remaining network was only half that of the original. In addition, the overall network structure had become less nested, indicating that pollination is more vulnerable to future perturbations. In New Zealand, decline in local pollinator numbers has been associated with decreased cross-pollination in a number of plant species including *Dactylanthus taylorii* (Cummings et al. 2014) and *Sophora microphylla* (Etten et al. 2015). Local 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).

A number of global meta-analyses as cited in section 3.1 above (based on considerable data from the WEOG region) indicate linkages between local declines in pollinator abundance and diversity, and decreasing wild plant pollination and seed production (Aguilar et al. 2006), noted effects of invasive plant species on visitation and reproduction of native plants (Morales and Traveset 2009), and showed the impacts on visitation rates of polliantors from habitat alteration and invasions (Montero-Castaño and Vila 2012). In most cases, however, historical base-line data are lacking and researchers use space-for-time substitution. Detecting historical trends in plant reproduction is difficult given the paucity of long-term studies and many gaps in our understanding remain.

4. Status of the main drivers of shifts in pollinators, pollination and pollinator dependent wild plants in all ecosystems

The IPBES Assessment highlighted some direct and indirect drivers of change that are affecting abundance, diversity and health of pollinators. As consequence, the negative impact in the provision of pollination services generates risks to societies and ecosystems. Indirect drivers of change include institutions and governance systems, and direct drivers includes land-use change, intensive agricultural management and pesticide use, environmental pollution, invasive alien species, pathogens and climate change. As the IPBES Assessment concluded, explicitly linking pollinator declines to individual or combinations of direct drivers is limited by data availability or complexity, yet case studies worldwide suggests that these direct drivers often affect pollinators negatively.

This report considers the following drivers:

1. **Land-use change:** habitat destruction, fragmentation and degradation, pollution, along with conventional intensive land management and agricultural practices.
2. **Pesticides:** 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.
3. **Genetically modified organisms (GMOs)**: Most GMOs carry traits for herbicide tolerance or insect resistance.
4. **Invasive species, pests and diseases**: Impacts on native pollinators and pollination may result from alien flowering plants, introduced alien pollinators and alien predators. Wild pollinators and their pollination services might be impacted by managed pollinators in different ways, mainly through competition and spill-over of diseases. 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.
5. **Climate change:** Some pollinator species have moved their ranges, altered their abundance and shifted their seasonal activities in response to observed climate change over recent decades, with impacts on their populations and overall distribution.

Land-use change

Pollinator richness generally declined with decreasing landscape heterogeneity (Andersson et al. 2013) and habitat destruction and fragmentation are likely to negatively affect pollinators (Donaldson et al. 2002; Harris and Johnson 2004). Areas where habitats are not completely transformed, such as heavy livestock grazing, can also negatively impact pollinators (Mayer 2004), changing the dominant guilds (Colville *et al.* 2002), and thus ecological processes. Although livestock grazing is not often seen as a driver of fragmentation, grazing can reduce flower availability of palatable plants, leading to changes in seed set and demography (Milton 1994; Mayer 2004).

However, fallow farmland can provide resources for pollinators and be associated with greater bee diversity (Chiawo *et al.* 2017) and bee species richness in wildflower strips on arable land is likely to increase with the amount of semi-natural habitats (Scheper *et al.* 2015). Global meta-analyses have shown how proximity to natural habitat can be important for wild pollinators, with pollinator species richness, visitation, and overall stabilisation of pollination services found to decrease with isolation from natural areas (Ricketts *et al.* 2008; Garibaldi *et al.* 2011; Kennedy *et al.* 2013).

Landscape configuration can play an important role in the maintenance of diverse pollinator communities. Decreased patch size, loss of habitat area and reduced connectivity have all been identified as important drivers of species richness declines (Marini *et al.* 2014). Furthermore, they negatively affect the ecological network link richness, leading to network contraction (IPBES 2016a).

While existing linear landscape elements in intensively used farmland (such as in Flanders, Belgium) may act as functional biological corridors facilitating pollen dispersal through pollinator movements (Van Geert *et al.* 2010), increasing wildflower patch size can lead to increases in wild bee density and result in greater seed set in wild flowers within agricultural landscapes, as found in Michigan, USA (Bennett & Isaacs 2014). There is also evidence from Great Britain that long-term declines in richness at local site levels may be buffered by the heterogeneity of the surrounding habitat (Senapathi *et al.* 2015). Thus a combination of large high-quality patches and heterogeneous landscapes may help to maintain high bee species richness and communities with diverse trait composition, which might stabilize pollination services provided to both crops and wild plants on local and landscape scales (Senapathi *et al.* 017).

The data on pollinator decline in Africa are scarce and only a few studies reporting on declines over a local scale are available (Pauw and Hawkins 2011). In Africa, many species of pollinators are in Sub-Saharan Africa are found in forest habitats. Deforestation continues to occur on the continent (Keenan et al. 2015). Reasons for deforestation are conversion of land for agriculture (Haines-Young 2009), and use of timber for construction and fuel (IEA 2016), Regions with less forest cover also show patterns of significant land use change. In South Africa, grassland is often converted to commercial plantations, resulting in a loss of foraging habitat leading to pollen limitation for plant species ultimately affecting the demographic structure of plant species as shown for *Brunsvigia radulosa* (Ward and Johnson 2005). In Durban, South Africa, the tree *Oxyanthus pyriformis* subsp. *pyriformis* is exclusively pollinated by a hawkmoth. The tree occurs in a few suburban habitat fragments, but is rarely visited by the hawkmoths and it shows almost complete absence of seed and fruit set with a >1000 fold increase by means of hand-pollination (Johnson et al. 2004).

In Asia, significant changes in landscape structure are created by oil palm plantations that cause large natural areas to be deforested. One policy response to such concerns is the pressure being applied to the buyers of these products to modify their practices to attain a sustainability certification (Carlson et al, 2018).

In Latin America, deforestation is a major driver in this respect. Three of the major biomes from South America have suffered intense deforestation mainly by conventional agriculture: the Amazonia biome (lost more than 20% of the forests during the last 30 years); the Cerrado biome (nearly half of this biome has been converted to pasture or cropland and only aproxymately 8% is formally protected by parks or indigenous reserves); and the Chaco biome (with deforestation rates among the highest in the world extending to an estimated 8 million ha of deforestation in the period 2000–2012). The Atlantic Forest is recognized for its high degree of species richness and rates of endemism, but only ca. 15% of its original forest cover is maintained. Moreover, this biome is highly fragmented with more than 80% of the forest patches less than 50 ha in size, revealing a serious situation. Nature reserves protect only 9% of the remaining forest and 1% of the original forest. In some specific locations of Amazonia, Atlantic Forest and Chaco are exhibiting a slow recovery over the last 10 years, as secondary forests establish (Teixeira et al. 2009; Yanai et al. 2017; Khanna et al. 2017; Noojipady et al. 2017; Fehlenbert et al. 2017).

Habitat loss and reduction of connectivity for pollinators in Brazil, Argentina and Costa Rica, has had documented negative effects on pollinators (e.g., Aguilar et al. 2006; Gonçalves et al. 2014; Ferreira et al. 2015; Nemésio et al. 2016; Boscolo et al. 2017; Rocha et al. 2017; Hadley et al. 2018). Social bees from the Atlantic Forest of NE Brazil (Bahia) showed negative effect of changes in forest cover at a local scale depending on regional forest cover (<35%; Ferreira et al. 2015). In addition, the reduction of habitat quality and landscape heterogeneity can cause species loss and can reduce robustness and resilience of plant-pollinator networks (data from Brazil and Argentina; Moreira et al. 2015). Hummingbirds have also declined ca. 40-50% across the gradient in deforestation in Costa Rica (Hadley et al. 2018). Fragmentation in Amazonia showed that smaller fragments had fewer species of bats and higher levels of dominance than continuous forest, and also a well conserved vegetation structure is a good predictor of bat abundance regardless landscape configuration (Rocha et al. 2017). Management strategies and conservation practices must integrate proper actions that consider both local and regional scales. For existing fragmented landscapes, it is important to increase forest availability at the regional scale, while also maintaining high environmental heterogeneity at the local scale. (Yanai et al. 2017; Khanna et al. 2017; Noojipady et al. 2017; Fehlenberget al. 2017)

Stingless bee nests in nature are in danger of depletion by logging practices. In Malaysia (Eltz et al., 2003) and in the Brazilian Amazon (Venturieri, unpubl. data) it has been shown that logging reduces stingless bees nests and, as a consequence, native pollinators, which has implications for forest recovery or restoration. Villanueva et al. (2005a) remarked that logging also reduces forest habitat that contains suitable, unoccupied nesting sites. Eltz et al. (2003) considered that the loss of pollinators occurs even if the rules for certified wood management are taken into account.
Attention to the need for general pollinator conservation among the rules of forest management need to be established.

Climate Change

With respect to global patterns in relation to climate change, regional studies show variation in species responses. A global study by Urban (2015) suggests that extinction risks will accelerate with future global temperatures, threatening up to one in six species under current policies with extinction risks highest in South America, Australia, and New Zealand.

Climate change has been documented as impacting various taxa at multiple levels, including causing range shifts in species and populations across the globe including Europe and North America (Parmesan & Yohe 2003; Chen *et al.* 2011a; Kerr *et al.* 2015), altering phenology – i.e. the timing of key life events (Fitter & Fitter 2002; Parmesan & Yohe 2003; Both & te Marvelde 2007), causing mismatches in phenological events (Visser *et al.* 2012; Chevillot *et al.* 2017) and increasing extinction risks (Thomas *et al.* 2004; Butchart *et al.* 2010; Rasmont & Iserbyt 2012). Alterations in physiology and phenology can de-synchronise ecological interactions and thereby threaten ecosystem function (Thackeray *et al.* 2016). These changes may also affect pollinators via changes in their spatial distribution, physiology and/or seasonal phenology through spatial and temporal mismatches between plants and their pollinators (Hegland *et al.* 2009; Kerr *et al.* 2015).

Increasing temperatures may either increase or decrease synchrony between species, depending on their respective starting positions (Singer & Parmesan. 2010). Bartomeus et al. (2011) reported that the phenology of 10 bee species from North-eastern North America has advanced by about 10 days over about 130 years with most of this advance occurring since 1970. 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. Observations that show the phenological de-coupling of plants and their pollinators are also presented by Gordo and Sanz (2006) and McKinney et al. (2012). While such long-term studies maybe rare due to lack of historic data, experimental studies such as the one by (Schenk *et al.* 2018), on *Osmia bicornis* show that even short temporal mismatches can cause clear fitness losses in solitary bees. Moreover, evidence suggests that climate constitutes a significant driver of plant-pollinator interactions, even within relatively narrow latitudinal gradients (Petanidou *et al.* 2018).

At ecosystem level, historical phenology data generate conflicting results, with some studies predicting plant-pollinator interactions will be vulnerable to future climate changes (Burkle *et al.* 2013), while others argue that these interactions will remain robust (Hegland *et al.* 2009). A modelling study (Memmott et al. (2007) using a highly resolved empirical network of interactions in England 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. 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 in USA (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 (Bartomeus *et al.* 2011; Burkle & Alarcón 2011).

Published work from the UK, clearly shows that autumn/winter temperature shifts can drastically alter the diapause or overwintering behaviour in insects (Coleman *et al.* 2014), and can even prevent emergence altogether (Bale & Hayward 2010). Quantitative data on key pollinators is severely lacking, however experimental studies on *Bombus terrestris* show that diapause is already being disrupted with queens attempting to establish winter colonies (Stelzer *et al.* 2010) and weight loss, survival, and emergence times being impacted in the mason bee *Osmia cornuta* (Bosch & Kemp 2004)

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 Hegland *et al.* 2009; Settele *et al.* 2014). Research conducted at Cornell University showed 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). 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 (Chen *et al.* 2011b) indicates that terrestrial species (e.g. butterflies) have moved poleward about 11-17 km per decade across various sites including Europe and North America. Kerr et al.'s (2015) study on bumblebees across Europe and North America (using over 110 years of observed data) found cross-continentally 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. These effects are were found to be independent of changing land uses or pesticide applications and underscore the need to test for climate impacts at both leading and trailing latitudinal and thermal limits for species.

There is 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 behaviour (Devictor *et al.* 2012). While the rate of climate change in Europe was equal to a 249 km northward shift between 1990-2008 (IPBES 2016a), 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.* 2012).

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 (Oliver *et al.* 2009; Willis & Bhagwat 2009).

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 (Gonzalez-Varo *et al.* 2013a; Vanbergen *et al.* 2013). 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).

A recent modelling study on hoverfly species in Europe has shown how species adapted to high mountains are projected to almost vanish from the Balkans and only regions of the Alps would remain suitable for them (Radenkovic *et al.* 2017). Another modelling study on butterflies in Britain has showed that while widespread drought-sensitive butterfly population extinctions could occur as early as 2050, measures to manage landscapes for pollinators and in particular to reduce habitat fragmentation could significantly improve the probability of persistence (Oliver *et al.* 2015). While regional studies show variation in species responses, a global study by Urban (2015) suggests that extinction risks will accelerate with future global temperatures, threatening up to one in six species under current policies with extinction risks highest in South America, Australia, and New Zealand.

The African continent is strongly exposed to the effects of climate change, especially through unpredictability and disrupted equilibria. For bees in the Cape Floristic Region diversity hotspot in South Africa it has been shown, applying modelling approaches, that most bees will show range shifts and/or range contractions under realistic models for increasing temperatures, while only a minority of species might show range expansions (Kuhlmann et al. 2012). Tropical insects might be affected more strongly by even a small increase in temperature than are insects at higher latitudes; tropical insects are often already living at the higher ranges of optimal temperatures. By contrast, species in higher latitudes have broader temperature tolerance and live in cooler climates well within broader physiological optima (Deutsch et al. 2008).

Unexpected disturbances of natural habitats might affect pollinators directly or indirectly via direct effects on their foraging plants or nesting habitats. In particular Mediterranean-type environments (which occur in different regions on the continent, in both southern and northern Africa) are sensitive to the frequency of fires (LeHouerou 1974), which in turn affect the reseeding and resprouting of plants. The replenishment of the vegetation depends on the juvenile periods of certain plants and might be restored after ten years in Mediterreanean-type ecosystems (Burrows et al. 2008). By contrast, grassland savannahs might be less vulnerable (Staver et al. 2011), as there might be plants may be adapted to quick resprouting (Higgins et al. 2000).

In all ecosystems however, frequencies of fires affect the ability for some plant species to mature and produce flowers (pyrophytes) (Jackson et al. 1999; Cavender-Bares et al. 2004), indirectly affecting pollinators that might specialize on such species. In regions where pollinators are scarce, plants competing heavily for pollinators resulting in specialized relationships between plant and pollinator species, making them more vulnerable to environmental disturbances as fires. Despite the importance of the frequency of fires, the spatial scale of the fire is equally important. Over the past two decades, the area burnt per year has slightly decreased in Northern Hemisphere Africa, while it has increased in Southern Hemisphere Africa (Giglio et al. 2013). Climate change models sugest that fires might increase in frequency, as the length of the fire wheather season will increase (Jolly et al. 2015). The recruiting and re-establishment of plant and pollinator populations is easier after small-scale than after large-scale fires, which will take longer time to be re-invaded by pollinator species (Brown et al. 2017).

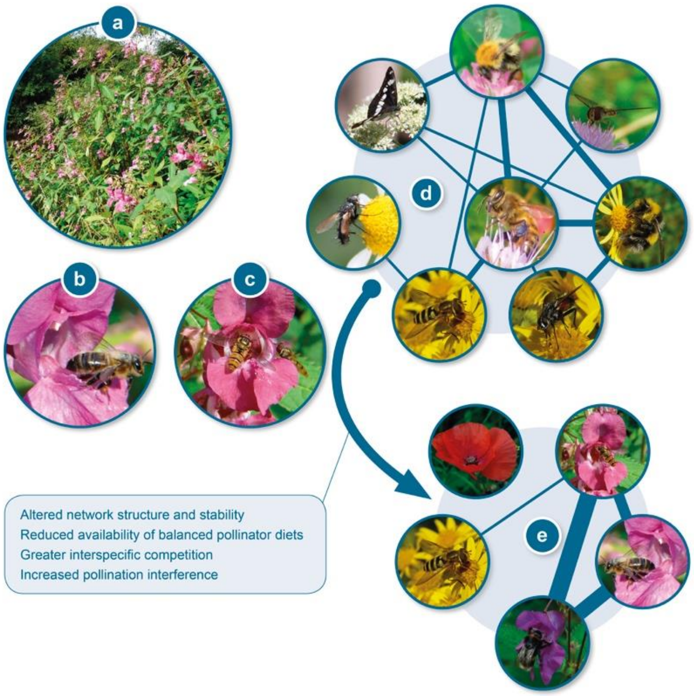
Grazing in grasslands might represent another kind of disturbance, which can affect pollinator species (Mayer 2005) and pollination services (Mayer 2004). Grasslands are generally adapted to grazing. Grazing might affect the availability of high-value nutritious flowering plants as forage resources for pollinators. However, grazing might contribute to removal of vegetation cover, which might create nesting habitats for ground-nesting pollinator species. Positive as well as negative effects might occur as a resonse to grazing depending on the pollinator species' biology resulting in shifts in community structure of invertebrates (Seymour and Dean 1999) and specifically pollinator insects (Mayer et al. 2006).

In Latin America, models shows that climate change will impact native stingless bees (e.g. *Melipona subnitida*, *Melipona bicolor* and *Melipona scutellaris,* which are locally important pollinator of wild and crop plants in NE Brazil, also exploited for honey production by regional beekeepers) disconnecting suitable habitats in the future that can threaten species dispersal and gene flow (Giannini et al. 2012, 2017). *Bombus bellicosus* is a South American species whose populations have disappeared from their northern distribution. Changes in land uses and climate are stressors believed to cause species declines. Climate change models showed that the suitable climatic conditions for *B. bellicosus* will cause their populations to retreat southwards (Martins et al. 2015).

Invasive Alien Species

With respect to the impacts of invasive alien species on pollinators, recent research on a global level has documented that these are spread as a consequence of the global growth in economic wealth, trade, commerce, and transport efficiency (Mack *et al.* 2000; Hulme 2009; Dawson *et al.* 2017). Invasive alien species create risks and opportunities for pollinator nutrition, re-organise species interactions to affect native pollination and community stability, and spread and select for virulent diseases (Vanbergen *et al.* 2018). For example, invasive alien species possessing generalised ecological traits or that are evolutionarily close to natives are readily incorporated into species networks and ecosystems. When attaining great abundance, they substantially modify structure and function of pollination systems, often negatively for native species (Vanbergen *et al.* 2018). Invasive alien species thus tend to represent a significant biological risk to pollinators and pollination, but effects are complex and contingent on the abundance of the invader and the ecological function and evolutionary history of both the invader and the recipient ecosystem.

Abundant invasive alien plant species can provide a substantial food resource that dominates pollinator diet (Fig. X, Invasive alien plant impact on pollinator visitation and network structure) (Praz *et al.* 2008; Bartomeus *et al.* 2010; Pysek *et al.* 2011; Chrobock *et al.* 2013; Masters & Emery 2015; Russo *et al.* 2016; Montero-Castaño & Vilà 2017; Stiers & Triest 2017). This may produce risks for pollinator health such as by altering the nutritional quality of the landscape. Pollinators require a specific balance of essential nutrients (Sedivy *et al.* 2011; Stabler *et al.* 2015; Vaudo *et al.* 2016) and if a plant invasion reduces the availability or diversity of particular nutrients provided by diverse floral resources, then nutritional deficits for pollinators might arise (Vanbergen *et al.* 2018). Other potential risks to pollinators from alien plants are from novel plant chemicals in alien pollen and nectar that can be toxic to native pollinator species (Praz *et al.* 2008; Arnold *et al.* 2014; Tiedeken *et al.* 2016); or alteration of flowering phenology at the community level affecting availability of floral resources' to foraging insects (Vanbergen *et al.* 2018).



Altered network structure and stability
Reduced availability of balanced pollinator diets
Greater interspecific competition
Increased pollination interference

Figure 6. Invasive alien plant impact on pollinator visitation and network structure. An example of an alien plant species (A) Himalayan balsam, Impatiens glandulifera native to Asia and invasive in Europe. This plant attains high densities, produces copious nectar and pollen and possesses a large, enduring floral display, all of which enables it to readily penetrate and dominate plant-pollinator networks by co-opting pollinators, such as (B) the honeybee and (C) syrphid hoverflies. In turn, alien plant invasions can alter the composition and structure of native plant-pollinator networks from (D) to (E). This raises the risk of (E) pollinator nutritional deficits due to reductions in availability of essential nutrients from diverse floral resources, poorly matched mutualisms and impaired native plant pollination. Source of images: D. Chapman, Cl. Carvell and A. Vanbergen.

However, there are comparatively few recorded examples of alien plant invasions, consistently lowering overall pollinator diversity or abundance (Lopezaraiza-Mikel *et al.* 2007; Moron *et al.* 2009; Nienhuis *et al.* 2009). This suggests that either pollinators can compensate for IAS-associated changes in floral quality or availability (Harmon-Threatt & Kremen 2015), or that effects are subtle, chronic (possibly undetected), or only adversely affect pollinators in combination with other stressors (Schweiger *et al.* 2010; Vanbergen & the Insect Pollinators Initiative 2013).

While direct links between invasive alien plants and pollinator decline are unproven, there is much evidence that invasive alien plant species are incorporated readily into, and often assume a key role, in native community networks (Bartomeus *et al.* 2008; Ramos-Jiliberto *et al.* 2012; Albrecht *et al.* 2014; CaraDonna *et al.* 2017; Goodell & Parker 2017; Kaiser-Bunbury *et al.* 2017). Once the invasive alien is established and abundant in the network, then it may affect pollination of coflowering native plants generally by outcompeting them for pollination services (Brown *et al.* 2002; Morales & Traveset 2009; Dietzsch *et al.* 2011; Thijs *et al.* 2012), although there are situations where they facilitate native pollination (McKinney & Goodell 2011; Masters & Emery 2015; Albrecht *et al.* 2016; Russo *et al.* 2016) Whether facilitation or local competition pre-dominates depends on the relative abundance and distribution of the alien plant and the species composition of the recipient ecosystem or habitat (Montero-Castano & Vila 2012; Carvalheiro *et al.* 2014; Albrecht *et al.* 2016; Russo *et al.* 2016).

Invasive alien plants can disrupt native pollination by improper native pollen transfer or reproductive interference (stigma clogging or chemical inhibition of pollen germination) ultimatley reducing native plant reproduction (Chittka & Schurkens 2001; Brown *et al.* 2002; Morales & Traveset 2008; Thijs *et al.* 2012; Bruckman & Campbell 2016b). Yet this may be mitigated by plant compensatory mechanisms, e.g.

self-reproduction, that can assure reproduction (Morales & Traveset 2009; Dietzsch *et al.* 2011; Emer *et al.* 2015). Overall, pollinator sharing and impacts on native plant pollination and reproduction tend to be greater where, relative to the native flora, the alien plant produces higher densities of flowers, they are related, or they have similar flower phenology or morphology (Pysek *et al.* 2011; Thijs *et al.* 2012; Carvalheiro *et al.* 2014; Bruckman & Campbell 2016a; Herron-Sweet *et al.* 2016).

Humans have globally translocated many different bee species for apiculture and crop pollination services, mostly from the European region (IPBES 2016a) and in some cases (e.g. *Apis mellifera*) these species are considered naturalized in other WEOG regions (N. America, Australasia). The principal managed pollinators (the western honeybee *Apis mellifera* and the bumblebee *Bombus terrestris*) possess traits (sociality, generalist diet and nesting flexibility) that coupled to recurrent introduction of managed colonies and frequent establishment of feral populations risk impacts on native pollinators and pollination. Compared to the situation in other world regions, there is less evidence that translocation and establishment of European bees across much of the WEOG has competitively excluded native pollinators or disrupted native pollination (Thomson 2004; Kenis et al. 2009; IPBES 2016a). One explanation might be that any impacts occurred long ago (e.g. ~200yrs in N. America) and so went unrecorded. The introduction exotic bees into Australia and New Zealand occurred in the 19th Century (Doull 1973; Donovan 2007) but, evidence of their impact on native flora and their pollinators is also limited (Howlett & Donovan 2010). In Australia, reduced seed set in the endemic shrub Melastoma affine was associated with high abundances of Apis mellifera, that in turn disturbed the foraging activity of native bees (Gross 1998).

In the WEOG region, the greatest risk to native pollinators from alien pollinators probably comes from trans-continental movement of pollinating bees leading to pathogen and parasite spill over into native pollinators, with potential implications for community epidemiology (IPBES 2016a; Vanbergen *et al.* 2018). Sustained movement by humans of managed honeybee (*A. mellifera*) colonies into Asia led to the host shift of the parasitic *Varroa* mite from Asian honeybee (*A. cerana*) populations and its subsequent worldwide spread, as part of trade in managed *A. mellifera*, along with a complex of viral pathogens it transmits among bee hosts (Martin *et al.* 2012; IPBES 2016a; Wilfert *et al.* 2016). This *Varroa* host shift appears to allowed eco-evolutionary changes that increased virulence of strains of Deformed Wing Virus (DWV) infecting honeybees, which contribute to colony losses (Martin *et al.* 2012; Ryabov *et al.* 2014; McMahon *et al.* 2016; Wilfert *et al.* 2016; Brosi *et al.* 2017)**.**This sort of process may lead to future risks for native pollinators where bees (managed or feral) are introduced because there are indications of pathogen sharing among a broad spectrum of flower-visiting insect hosts (Singh *et al.* 2010; Furst *et al.* 2014; McMahon *et al.* 2015; Wilfert *et al.* 2016).

Where predators of pollinators are introduced beyond their native range they typically exert strong top-down pressure on plant pollination and fitness by consumption of pollinators (IPBES 2016a). In the WEOG region, the accidental introduction (2004) into Europe from Asia and subsequent spread of the predatory yellow-legged hornet (*Vespa velutina*) represents a new threat to already stressed European honey bee populations (Monceau *et al.* 2014; Keeling *et al.* 2017b). In California invasive ant species that are more aggressive or competitive than native ants deter pollinators and seed dispersers thereby reducing plant fitness (Hanna *et al.* 2015). Effects of invasive alien predators on the web of interspecific interactions affecting native pollination can be highly complex and is well illustrated by the case of the invasive predatory wasp (*Vespula pensylvanica*) in Hawaii (Hanna *et al.* 2013, 2014). This wasp is a generalist predator of arthropods, but also behaves as a nectar thief to compete with native *Hylaeus* bees and the alien honeybee *A. mellifera*. This predator thereby lowers pollination and fruit production of native tree *Metrosideros polymorpha*, (Hanna *et al.* 2013, 2014). Experimental removal of the wasp revealed the alien *A. mellifera* was the most effective pollinator in this system, in all likelihood fulfilling the niche of extinct or declining bird pollinators, themselves reduced by introduced vertebrate predators (Hanna *et al.* 2013).. In a similar instance, predatory alien wasps from Europe in New Zealand beech forests outcompete native bird pollinators and alien honeybees (*A. mellifera*) for energy rich honeydew secretions produced by native scale insects and thus threaten the native bird pollinators (Moller *et al.* 1991; Markwell *et al.* 1993).

Several invasive alien species have been recognised in Africa, although the impacts on pollinators have been found for a few species only. As invasive plant species are often capable of selfing, interactions with pollinators are not expected and hence direct effects might be negligible. The prime examples of invasive plant species are *Parthenium hysterophorus* and plants of the genus *Prosopis*, also known as Mesquite. The latter one was introduced as beneficial plant to provide fodder, fuel and shade in arid regions. Especially in water scarce areas *Prosopis* turned into an undesirable invasive alien weed due to its huge water usage and occurrence in dense stands (Le Maitre et al. 2000). *P. hysterophorus* has a high potential to disrupt natural ecosystems resulting in grass- and woodlands dominated by *P. hysterophorus* reducing the number of indigenous plants. The genus *Prosopis* is highy attractive to pollinators, provided both a concentrated nectar and abundant pollen. However, direct effects on pollinators have not been described.

In contrast, the invasive *Rubus cuneifolius* in South Africa sets remnant grassland patches (RGPs) under threat. *R. cuneifolius* reduces the ecological complexity of RGPs, which otherwise are valuable contributors to regional biodiversity by promoting intact flower visitor networks (Hansen et al. 2018).

Sunbirds in Namaqualand have adapted to the invasive tree tobacco *Nicotiana glauca* by pollinating through hovering, as do New World hummingbirds; sunbirds usually pollinate by perching. *N. glauca* flowers pollinated by hovering sunbirds show higher seed set (Geerts & Pauw 2009).

Invasive alien animal species acting as competitors or predators could set pollinators at risk, but so far there are only very few invasive alien insect species being reported in Africa. The invasive paper wasp *Polistes dominula*, introduced in 2008 to South Africa (Eardley et al. 2009), hunts bee species, but major impacts on native wild pollinators have not been reported, especially as several parasitoids of native species also use *P. dominula* as their host (Benadé et al. 2014).

In Latin America, one example is the two species of bumblebees (*Bombus ruderatus* and *Bombus terrestris*) were introduced into continental Chile and later arrived into Argentina and also crossed to Navarino Island in southern Chile with disastrous consequences for the native bumblebee *Bombus dahlbomii*. *Bombus terrestris* has spread by some 200 km/year from Chile and had reached the Atlantic coast in Argentina in a short period of around 15 years (Torretta et al. 2006; Morales et al. 2013; Schmid-Hempel et al. 2014; Morales et al. 2017; Malinger et al. 2017; Rendoll-Carcamo et al. 2017). These invasive species are infected by protozoan parasites that seem to spread along with the imported hosts and spread out to native species (Schmid-Hempel et al. 2014). A large-scale investigation in Patagonia showed that after only 5 years post-arrival, *B. terrestris* has increased its populations with the concurrent geographic retraction of the native *B. dahlbomii* (Morales et al. 2013, 2017) For some plant species (e.g. *Alstroemeria aurea*) invasive bumblebees have replaced *B. dahlbomii*, which was the most abundant pollinator of this endemic herb few years ago (Morales et al. 2013). Stakeholders should consider the continuous evaluation of adequate densities of managed colonies for different regions, the relative resource availability for pollinators (i.e. native species and flowering crops), the health status of the managed bee colonies monitoring for pathogens and parasites, and performing baselines to evaluate declines in wild bee species.

Managed Pollinators

Management of stingless bees in Africa is still in its infancy (Cortopassi-Laurino et al. 2006). Management of colonies, including artificial reproduction and queen rearing, needs to be advanced in order to use natural resources in a sustainable way.

A mismanagement error helped to spread the Cape honeybee, *A. mellifera capensis*, from its native range around the Western Cape region into the region of the adjacent subspecies *A. mellifera scutellata* (Greeff 1997). When moved into the range of *A. mellifera scutellata*, the Cape honeybee workers invade colonies, kill the resident queen and establish themselves as egg-laying workers producing female offspring (Neumann and Moritz 2002). As these bees are all essentially reproductives, there is little or no foraging in these colonies, and they soon run out of nectar and pollen reserves. These colonies then dwindle in size to only a few hundred bees, which then either die-out or invade other *scutellata* colonies, thus repeating the cycle (Allsopp 2004). The spread of the Cape honeybee workers has been facilitated by keeping colonies at high densities enhancing parasite transmission and by frequent transportation enhancing long-distance dispersal.

Competition between managed bees, mainly the honeybee, and wild pollinators has been documented in several contexts. In Kenya it has been observed that honeybees outcompete wild bees specialized on a few plant species during seasons with scarce floral resources (Martins 2004). In South Africa it has been shown that honeybees start foraging early, diminishing the pool of pollen available for dispersal by bird pollinators that start foraging at later time points (Hargreaves et al. 2010). In Mauritius honeybees compete with endemic nectarivorous birds, the Grey White-eye, *Zosterops borbonicus mauritianus* Gmelin, and the Olive White-eye, *Z. chloronothos* Viellot for nectar on two endemic trees (Hansen et al. 2002).

Another invasive alien species is the dwarf honeybee *Apis florea*, introduced via air transport from Pakistan into Khartoum, Sudan in 1985 (Lord and Nagi 1987). Since then, this bee has spread along the river Nile and the Ethiopian highlands up to Djibouti in 2014 (Steiner 2017). A second introduction occurred in 2017 in Egypt (Shebl 2017). In its current distribution range it is coexisting with *A. mellifera* and no threats have been observed. The honeybee *A. mellifera* does not appear to be competing over resources with the dwarf honeybee (El Shafie et al. 2002) and pathgogen spill-over has not been reported. Impact on the wild bee fauna of the invaded areas has not been documented so far.

In Asia, the erosion of traditional knowledge systems (management of local bees, including honey bees and stingless bees), particularly in tropical countries, may contribute to local declines. The management and keeping of the Asian hive bee, *Apis cerana* has revolved around a number of traditional practices, such as the use of wall hives. These practices are now less and less common in the region.

Pesticides

Laboratory studies on the impact of neonicotinoids (e.g. thiamethoxam, clothianidin) on bumblebee species and the honey bee (*Apis mellifera*) increasingly tend to employ field-realistic doses (~1.5-5 ppb) in experimental treatments. This has revealed reductions in feeding rate, lifespan, and reproductive capacity (i.e. ovary development, sperm viability, egg production) of queen and male (drone) bees (Baron *et al.* 2017a, Baron *et al.* 2017b, Straub *et al.* 2016). Semi-field experiments (where insects are subject to a treatment and then allowed to forage freely) continue to reveal complex, subtle effects of neonicotinoids on bumblebee foraging behaviour (e.g. floral choice, foraging duration, pollen collection rates), although the level and directions of effects often varies among studies (Stanley & Raine 2016, Stanley *et al.* 2016, Arce *et al.* 2017).

A controlled landscape experiment implemented across three countries (Hungary, Germany, and the United Kingdom) that employed oilseed rape (canola) treated with neonicotinoids (clothianidin or thiamethoxam) showed wild bee reproduction (*B. terrestris* and *Osmia bicornis*) was negatively related with neonicotinoid residues in the bee nests (Woodcock *et al.* 2017), broadly in agreement with Rundlof *et al.* (2015). Honey bee colony performance varied geographically with negative (Hungary and United Kingdom) or positive (Germany) responses recorded. Only in Hungary did this translate into reduced colony viability (24% decline in workers in the following spring) (Woodcock *et al.* 2017). This complexity in honey bee responses follows overall equivocal results from earlier honey bee field experiments in Sweden and N. America (Rundlof *et al.* 2015; IPBES 2016).

Recent distribution modelling integrating long-term citizen science records of wild bee species occurrence across the British landscape revealed while the bee species benefit from the pollen and nectar provided by the oilseed rape crop, they were on average three times more negatively affected by exposure to neonicotinoids than species that did not forage on the treated crop, leading to increased population extinction rates (Woodcock *et al.* 2016). Similarly, analysis of long-term butterfly population data from Northern California revealed a negative association between butterfly populations and increasing neonicotinoid application, even after controlling for land use and other factors (Forister *et al.* 2016).

Fungicides, in combination with other agrichemicals (e.g. insecticides), have an impact on bee health, for an example by impeding resistance to disease (IPBES 2016). Recent studies have produced further correlative evidence of the role of fungicides in range contractions of N. American bumblebee species, potentially due to interactions with pathogens (*Nosema bombi*) and/or climate (McArt *et al.* 2017b). The risk to honey bees from fungicide exposure extends beyond the cropped area (McArt *et al.* 2017a). Gene function and cellular metabolism of honey bees experimentally exposed to diets containing triazole fungicides is disrupted with implications for the bee's capacity to extract sufficient energy from and detoxify secondary plant compounds in food (Mao *et al.* 2017).

Pervasive neonicotinoid and fungicide contamination of floral resources and honey stores mean it is likely that there is a risk to wild pollinators from sustained exposure to sub-lethal doses of multiple pesticides (David *et al.* 2016; Hladik *et al.* 2016; Mogren & Lundgren 2016; Colwell *et al.* 2017; McArt *et al.* 2017a; Mitchell *et al.* 2017). Experiments show a risk of synergy between chemical and other stressors (nutritional state, pathogen presence) that could have implications for pollinator health and survival e.g. via reduced immunocompetence (Brandt *et al.* 2016; IPBES 2016; Brandt *et al.* 2017; Tosi *et al.* 2017).

Agricultural intensification

There is experimental evidence linking pesticides with negative effects on pollinators in Brazil and Argentina region. For example, *Apis mellifera* is used as the test species in environmental risk assessments, and it is generally considered as extremely sensitive to pesticides. A recent meta-analysis showed that when compared to honey bees, stingless bees (*M. scutellaris, N. perilampoides, S. postica*, *T. iridipennis*, *T. nigra* and *T. spinipes*) are more sensitive to different pesticides than other bee species (e.g., bumblebees, etc.) (Arena and Sgolastra 2014). Nevertheless evidence is based mainly on dose–response toxicity bioassays, with scant information on field sublethal effects of pesticides (del Sarto et al. 2014; Balbuena et al. 2015; Barbosa et al. 2015). Experimental studies performed with the great fruit eating bat (*Artibeus lituratus*) from Brazil indicate that the chronic exposure of fruit bats to environmentally relevant concentrations of endosulfan can lead to significant bioaccumulation, which may affect the health of this important seed disperser in neotropical forests (Brinati et al. 2016).

Genetically modified organisms (GMOs) are herbicide tolerant (HT) or insect resistance (IR), may threat pollinators by lethal or sub-lethal effects on adult insects or larvae (Lima et al. 2011; Paula et al. 2014; Villanueva-Gutiérrez et al. 2014; Ricroch et al. 2017; Pandolfo et al. 2017).. A recent review on honeybees showed no clear negative effects (Ricroch et al. 2017). There are some examples spreading GMOs crops in ruderal and natural habitats as GMO soybean pollen in honey from Yucatán (Mexico) or the spreading of the glyphosate resistant *Brassica rapa* biotypes mediated by pollinators. Evidence from Mexico demonstrates that bees visit the flowers of soybean and pollen of GMO soybean flowers is present in honey harvested by beekeepers (Villanueva-Gutiérrez et al. 2014). The persistence of transgenes in nature was confirmed at central Argetina, implying gene flow and introgression between feral populations of GM *B. napus* and wild *B. rapa* (Pandolfo et al. 2017). The risks may depend on GMO spreading (plants or pollen) their concentration in pollen or nectar and also on the species of pollinators. More studies are needed.

V. RESPONSE OPTIONS TO CONSERVE AND MANAGE POLLINATOR, THEIR HABITATS AND THEIR FOOD AND WATER RESOURCES IN ALL ECOSYSTEMS

5.1 Improving local knowledge on pollination and pollinators

The main gap for underdeveloped countries on shaping land management policy to benefit pollinators is the lack of knowledge of local pollinators and their importance to improve biodiversity and agricultural yields. For most regions, the management of native bees and appropriate beekeeping practices are unknown. This lack of knowledge facilitates the introduction of alien commercial pollinators. The non-managed pollinators, of great importance for the maintenance of biodiversity (as bats, birds, small mammals, moths, butterflies, flies, beetles, etc.) generally are also not considered in their roles as ecosystem services providers.

Most of the knowledge base on understanding land management and pollination services comes from research carried out in the Northern Hemisphere. Often this research is made possible or supported through the scientific collections that provide free information on species identification and distributions. In Latin America, data on pollinators are partially available (also in IABIN and GBIF). Through these data sources, it is possible to identify critical areas that have not been thoroughly sampled and are also under the threat of imminent deforestation by the anthropogenic activities. An example of deforestation in large scale is the Amazon area in last 50 years. In this region, considered strategic for global climate, the arch of deforestation is shown in the map from 1978 until 2015 (Figure 7). Yet there is a dearth of knowledge on pollinators and their role in forest health and integrity, in this region. There is some evidence of how deforestation impacts many bees' richness and healthy populations, which may inform policy. Tropical deforestation has been documented to change bee communities due to foraging characteristics (Campbell et al. 2018; Lichtenberg et al. 2017; Hipólito et al. 2018), yet the specific changes and measures to mitigate these are hampered by insufficient taxonomic knowledge and research findings in the regions where pressures are the greatest.

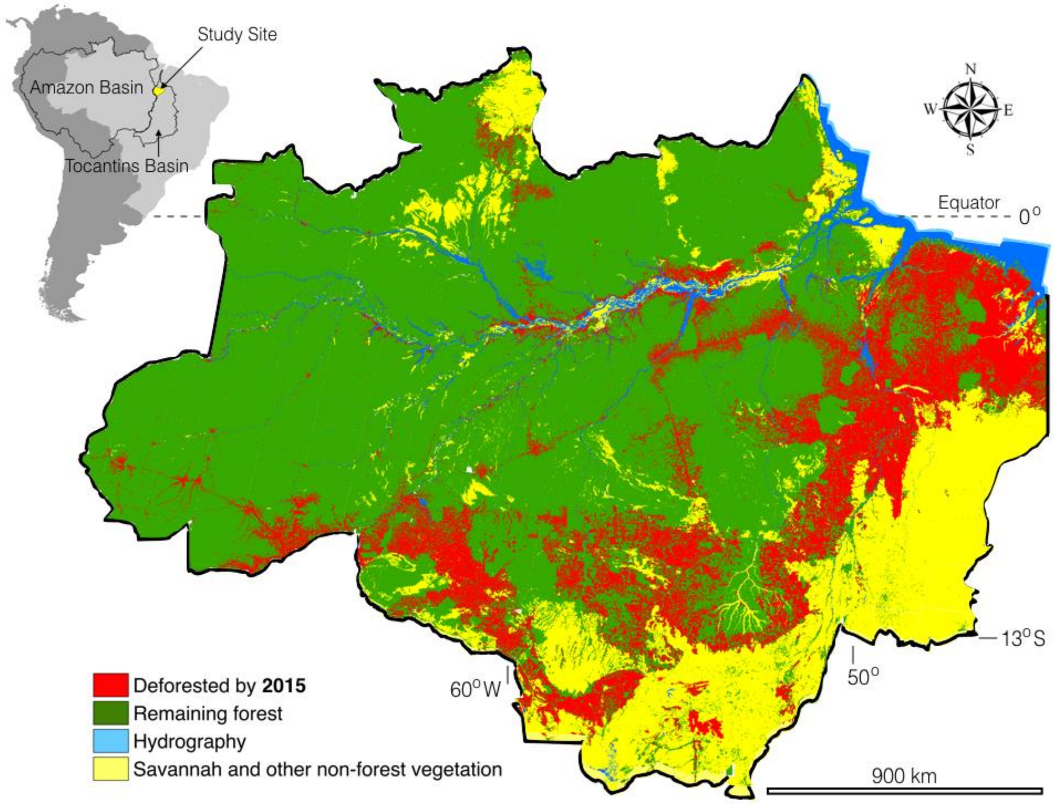


Figure 7. Map of current status of deforestation in the Amazon Basin, provided courtesy of Dr. Luiz Gylvan Meira, Instituto Tecnológico Vale - Desenvolvimento Sustentável

5.2 Landscape approaches to pollinator conservation and climate change

A landscape approach to pollinator conservation can be very useful in combining a land use approach with restoration planning for pollinators (Pufal et al. 2017; Potts et al. 2016). As most natural areas are under pressure by anthropogenic uses - mainly agriculture - the ranges for pollinators activities are changing. The main drivers of change on pollinators and pollination in tropical areas are land use change (a proximate drive), and climate change (a distal drive), both very difficult to mitigate.

Recent studies have recently been carried out considering landscape structure, and showing how pollinator communities are linked to the health of ecosystems (for instance Campbell et al, 2018, a study with assaí palm in Amazon; Hipolito et al, 2018, for coffee crops under forest). For example, the possibility to maintain natural vegetation, corridors between crop fields and diversified production systems could help to conserve pollinators and to sustain pollination services. Where extensive natural area is lacking, small stands of traditional diversified farming systems already present in most regions of Latin America serve as an alternative to support native pollinators.

Another point of concern on significant changes in landscape structure is related to crops such as oil palm that causes a large natural area to be deforested. One policy response to such concerns is the pressure being applied to the buyers of these products to modify their practices to attain a sustainability certification (Carlson et al, 2018).

Also within the landscape approach is the use of landscape genetics, a very powerful tool to determine population characteristics of pollinators, as well as the genetic consequences of bee management in large areas, inside or outside their distribution areas (Jaffe et al, 2016 a, 2016b; Brito et al, 2010; Byatt et al, 2015, Quezada-Éuan et al., 2012). A recent example in a tropical area is the recent analysis using (Chapman et al, 2017), showing the impact on the genetic structure of a stingless bee, *Tetragonula* in Australia, from anthropogenic displacement of colonies among areas, a problem for managing pollinators.

Using collection data it is possible to fit in the landscape the ongoing and future occurrence area for many pollinators, as bats (Costa et al, 2018; Gutierrez et al, 2017) in Brazil and Mexico. With these maps in mind, as well as data on biology of pollinators, it is possible to plan restoration and social impacts, as studied for the stingless bee *Melipona subnitida* who lives in an extreme dry area (Giannini et al, 2017b). The use of landscapes attributes is needed to design measures to buffer the joint effect on habitat configuration and global change (Giannini et al. 2015). Modelling is a very important tool for restoration and also to predict the impact of global change and to define protection policies for pollinators in next years (see Giannini et al. 2013; Giannini et al. 2015; Giannini et al. 2017a e b; Costa et al. 2018).

Global change suggests several mitigation opportunities for pollinators (Settele et al. 2016; Scheffers et al. 2016). One of them is the assisted displacement of managed colonies of social pollinators. The genetic structure of populations was not considered at this time, and this is a constraint for the managed displacement. There are few examples of population genetics of solitary pollinators for sustainable managing (Toledo et al. 2017). This issue needs attention, because most pollinators are wild and for them is it necessary to restore corridors and methods to evaluate if they will find or not a suitable place in the future, due to the effects of global change in the near future for tropical areas of the world. In temperate areas, Kerr et al. (2015) showed the projection of the range of bumblebees into the future.

5.3 Land or habitat restoration with pollinator considerations

Community Indigenous Conservation Areas – Options for pollinator conservation: Natural or semi-natural habitats managed by local communities, particularly indigenous communities by means of customary laws/rules are important conservation hotspots (ICCA 2012). These areas are often outside state-regulated conservation protected area networks and are important biodiversity refuges providing valuable ecosystem services including pollination (Bhagwat 2009, Blitcharska et al. 2013). Bodin et al. (2006) reports that the community managed forests in Madagaskar improved crop pollination in adjacent farming landscape. Sacred groves are excellent examples of such community protected forests and previously extended from Asia, Africa, Europe and Americas but their present occurrence is mostly restricted in Africa and Asia (Chandran et al. 1998). Sacred groves have been extensively documented in Nigeria (Onyekwelu & Olusola 2014), India (Malhotra et al. 2001), Thailand (Wangpakkapattnawong et al. 2010), Philippines (Rebancos & Buot 2007), and several other south and south eastern Asian countries (Ramakrishnan et al. 2012 in Parotta and Trosper 2012). Habitat enhancement might be necessary for improving conditions for pollinators, such as introducing artificial nests (bee hotels). Several reports indicate open forests to harbor larger numbers of pollinator populations compared to closed canopy forests (Taki et al 2013, Wratten et al. 2012, Winfree et al. 2007). Areas where shifting cultivation practices exist, often host a series of forests in different successional stages; their pollinator fauna should to be studied so that suitable management strategies for better pollinator fauna can be informed by such studies. Financial schemes and insurance programs may be conceived that would further incentivize management of such forests by the local communities.

Provision of indirect and direct economic incentives: Incentives for supporting biodiversity in farmlands has been proactively practiced in EU countries through Agri-Environment Schemes. These schemes have so far been the best known examples of protecting biodiversity through providing incentives (Batary et al 2015). A number of studies have linked increased pollinator abundance and/or diversity through such schemes. The US Farm Bill (2008) had made specific economic provision for pollinator conservation when it was further ratified in the 2014 Farm Bill. These models show the potential of conserving non cropped land in agricultural landscape that can go a long way in conserving pollinators (Morandin and Winston 2006). While such schemes are generally non-existent in the developing world, economic incentive models for forest protection have emerged from India. Joint Forest Management (JFM), a participatory forest management program, was launched in the early 1990s in West Bengal, an eastern Indian state that involved local communities in forest protection. The Forest Department extended revenue shares to the forest protection committees constituted by the villagers. This model later spread across other Indian states and has since been adopted in other developing countries too for protection or afforestation (Klooster and Masera 2000; Pagdee et al. 2006; Ellis and Bolland 2008; Porter-Bolland et al. 2012). Although such programs have not explicitly included pollinator conservation as a focal theme, the incentivized community protected forests outside the protected area network do contribute significantly to pollinator conservation as well. Similar models can and should be developed with a more clearly stated objective of conserving pollinators. Maderson and Wynne-Jones 2016 make a point about harnessing bee keepers' knowledge and involving them in participatory bee conservation program. Direct or indirect provisioning of incentives can be a mechanism through which this can work. Village level Biodiversity Management Committees (BMC) are an integral part of India's regulation of biodiversity. These BMCs are the custodians of biodiversity knowledge at the local community level. These bodies can be vehicles for incentivizing the conservation of pollinators and/or their habitats. Similar bodies exists in other developing countries as well.

Disaster risk management: Pollinator conservation can be synonymous with habitat conservation and restoration or regeneration of natural habitats. Habitats in flood plain regions recurrently face flood and the frequency of flood is increasing in the climate change scenario (Arnell and Gosling, 2016) Tree planting is often identified as an important activity in climate change adaptation programs (Pramova, et al. 2012) and pollinator friendly shrubs or trees can be selected in such tree planting programs.

5.4 Managed pollinators: honey bees

Often, managed pollinators are transported over long distances for the purpose of pollination (Cavigli et al. 2016). They are even transported outside of their natural distribution range (e.g. African honey bees into Brasil, European bumble bees into Australia, Asia and South America) (Moritz et al. 2005).

Transportation of managed pollinators: Transport of managed pollinators, whether local, regional or global, appears to be uncoordinated and unregulated. There is an urgent need to set-up and harmonize regulations for the trade of managed pollinators. This should aim at prevention of relocating species or their genetic material from their native range into non-native habitats. There, species might compete and spill-over diseases to native and endemic pollinators setting them and their pollination services at risk. In order to prevent such catastrophic introductions into non-native ranges, some countries have set-up national regulations, e.g. South Africa has restricted imports of non-native species and requires a preliminary environmental impact assessment (NEMBA 2004). Placement of managed pollinators in or close to protected areas should be better understood and carefully planned to protect wild pollinators from competition (Campbell and Motton 1985; Goulson 2003; Lopezaraiza-Mikel et al. 2007), avoid the spread of diseases (Fürst et al. 2014) and hence prevent negative impacts on wild plant pollination.

Regulation of international trade in pollinators: When regulating the international trade with species, colonies and queens (noting that trade with queens is quite common in the honey bee industry) it would be recommended to first promote and support the use of native/local alternatives for pollination purposes (Jaffé et al. 2010), before the need for trade is accepted. Regulations on pollinator movement need to consider the economic value of the pollinator movement, which might represent a trade-off, recognizing that movement of pollinators might be required for pollination purposes in agricultural production, e.g.

pollination of almonds, apples and melons (Kremen 2005). If however, quarantine requirements to prevent spillover and transmittal of bee pests and disease are instituted for importing or exporting pollinators, this might also create incentives to support local wild pollinators. In addition to any regulation, there is need for elaborating a set of best management practices (BMP) for the transportation of bees, which needs to be based on an assessment of the appropriate use of different transportation structures (roadways, railways, as air cargo, etc.).

Best management practices in pollinator management: Aside from large scale transportation of managed pollinators, management within the apiaries of managed bees may play a role in the transmission and spread of diseases between colonies (Brosi et al. 2017). For this purpose best management practices (BMP), similar to the Good Agricultural Practice (GAP) promoted by the FAO, need to be developed that respect local differences in beekeeping and hive management. BMPs have been developed on country-level, e.g. Australia (Somerville 2007), USA (for crop pollination in California) (Heintz et al. 2011), and some other non-compulsive suggestions have been put forward (Ritter 2013), overall lacking international harmonization. Establishment of BMPs should be based on rigorous education of beekeepers and farmers on interactions between managed and wild pollinators and the requirements of wild pollinator communities and wild plant pollination.

Risk management: Preventive measures put in place should be accompanied by a set of effective responses to current and emerging risks, such as diseases, invasive alien species etc. Introductions of non-native species can happen accidentally, but the spread of an invasive species or pathogen might be suppressed by early measures and activities, like the establishment of quarantine areas etc. There are a few regions in the world (parts of Australia, Seychelles), that are not affected by the ectoparasitic mite *Varroa destructor* (Locke 2016), the most detrimental honey bee pest. This mite also acts as vector for a number of bee viruses, which might spill over to wild bee species (McMahon et al. 2015). Effective counter-measures need to be put in place to prevent the invasion of these areas by the mite, if it is ever introduced. Australia has set up an information platform[9], which aims at disseminating information relevant to all relevant stakeholders.

Monitoring: The spread of pests and pathogens amongst managed pollinators and the spill-over to wild pollinators and its prevention and management requires continuous monitoring of both managed pollinators and wild pollinators. The state institutions that are in charge of the monitoring need to be reinforced and maintained. Additionally, a harmonized reporting procedure, data management strategy, data storage including accessibility needs to be established, so that current and emerging risks and threats can be detected almost in real time and across borders, so that respective counter-measures can be activated. Over the last several years, the COLOSS network[10] - whose activities are dependent on voluntary engagement of stakeholders - has set up surveillance networks and task force teams to enforce and standardize research, build capacity, create awareness for sustainable bee management and inform policy makers.

5.5 Knowledge management

There are three key areas where knowledge and data management are crucial to supporting the role of pollinators in conservation and sustainable use of biodiversity in all ecosystems: i) Knowledge about the **status** of pollinators; ii) Knowledge about the **risks to human well-being** generated by pollinator decline and iii) Knowledge about what the best **responses** are, and where and when they should be implemented.

5.5.1 *Knowledge about the status of pollinators*

As described in section 3, there is a severe lack of long-term data on the status of wild pollinator populations and communities in much of the world, especially for invertebrate pollinators. For example, in the European Red List of bees, 57% of the European bee species were classed as ‘data deficient’, with not enough information to assign a threatened status (Nieto et al. 2014). Addressing this data shortfall

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[9] http://beeaware.org.au

[10] http://www.coloss.org

requires a combination of taxonomic training and capacity building, standardized long term monitoring at national scales and accessible, well-curated databases.

An exemplary response has been demonstrated by the JRS Biodiversity Foundation Pollinators Programme[11]. This funds projects in African countries. It aims to increase the accessibility and quality of pollinator biodiversity data, through long-term investment in collecting baseline data, developing technologies and methods to do so, and creating data-sharing platforms relevant at regional and local levels. The programme has already funded projects to develop long-term monitoring and accessible databases.

Many initiatives like IABIN Thematic Pollinators Network, the BIOTA Program, GBIF and Species 2000 have been supporting the digitalisation of pollinator-related biodiversity collections, offering possibilities to open access for biodiversity data. Most tropical biodiversity is poorly known, and an effort to use new tools to access information on pollinators has assisted in this.

Notable tools and effots include DNA barcoding; biodiversity informatics tools; geographical referencing for the museum specimens; and the focus on small collections. As yet, however, open access to data of most of regional collections is not completely available for general public, making it difficult to map the occurrences of better studied groups and to provide evidence on areas not sampled until now.

Taxonomy

The support of basic scientific projects to increase the taxonomic knowledge of pollinators may be important for pollinator conservation; however taxonomy is only a priority for invertebrate pollinators. In comparison, vertebrate pollinators, which include birds, mammals and reptiles (Ratto et al. 2018), are extremely well-known taxonomically.

Reviews of regional conservation needs for native bees have identified a shortage of taxonomic expertise as a constraint to effective conservation action. Many regions 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). There are excellent examples of training courses in bee taxonomy. The American Museum of Natural History has conducted a training course annually since 1999, training >250 people[12]. The European COST Action SuperB[13] has funded three annual European bee taxonomy courses, in 2016, 2017 and 2018, targeted towards European early career researchers. The funding for these courses ends in 2018.

Provision of such courses in developing countries is limited by the availability of funding. Taxonomy training is perhaps best delivered by National Museums with entomology collections. Financial support for these institutions in data-poor regions (Africa, Latin America and Asia-Pacific) could be used to support taxonomy capacity building.

Standardized long-term monitoring

Much of our current understanding of declines in pollinator species comes from analysis of temporal trends in distributional data, gleaned from ad hoc records of species presence, collected by amateur naturalists (Bartomeus et al. 2013; Carvalheiro et al. 2013), or from IUCN Red List Assessments (Nieto et al. 2014; Regan et al. 2015). Standardized long-term monitoring of pollinators and pollination services is urgently needed to provide good quality information about patterns of decline and to target interventions (IPBES 2016a).

The UK Government has funded the development of a Pollinator Monitoring and Research Partnership (PMRAG)[14], which aims to combine improved analyses of long-term records with new systematic survey activity to establish how insect pollinator populations are changing across Great Britain. This is funded

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[11] http://jrsbiodiversity.org/our-programs/pollinators/

[12] https://www.amnh.org/our-research/invertebrate-zoology/bee-course-2018

[13] http://www.superb-project.eu/

[14] https://www.ceh.ac.uk/our-science/projects/pollinator-monitoring

initially for two years (2017 and 2018) as part of the National Pollinator Strategies for Scotland, England and Wales.

The PMRAG recently conducted a stocktaking of pollinator monitoring programmes around the world, which identified 73 monitoring programmes. The vast majority (47) were in Europe, with only 2 in Africa and 1 in Asia. For example, in 2017 JRS Biodiversity Foundation funded the College of African Wildlife Management (CAWM) in Mweka, Tanzania, to develop and implement a monitoring program for Tanzanian bee species[15]. The most frequently included taxa across all these monitoring programmes were bumble bees (*Bombus* sp.) and butterflies. The majority of the initiatives (51/73) depend on data collected by volunteers, or 'citizen scientists'. A number of these used standardised monitoring methods such as transect walks or pan traps. This indicates the feasibility of such monitoring programmes, but a dearth of activity in the data-poor areas of the world.

The majority of existing literature focuses on specific hymenopteran groups, but there is a lack of information on the impact of landscape changes on non-bee taxa (Senapathi *et al.* 2017). Most studies have taken a 'spatial' approach (as evidenced above) while very few studies have assessed changes over time by resampling sites across multiple habitat and land cover types and comparing findings to historical data sets (Burkle *et al.* 2013; Aguirre-Gutierrez *et al.* 2015; Senapathi *et al.* 2015; Aguirre-Gutiérrez *et al.* 2016). Studies combining both spatial and temporal approaches are rare (see, Carvalheiro *et al.* 2013) but are required to obtain a comprehensive understanding of land-use changes on pollinator communities.

Accessible Databases

Good taxonomy, volunteer recording and long-term monitoring of pollinator populations are only useful to policy in the long term if the data they generate are stored and curated in accessible databases.

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) 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) 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).

Three projects funded by JRS Biodiversity Foundation provide Africa-focused examples[16]. The National Museums of Kenya were funded to collect and disseminate data on Lepidoptera in critical habitats in East Africa. This includes digitizing records of species occurrence held by partner institutions across the region, and publishing those data to make them available for use by managers and conservationists. A second project, led by the Royal Museum of Central Africa, is funded to assemble, enhance, and mobilize data on the diversity and distribution of Diptera.

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[15] http://jrsbiodiversity.org/cawmm-2017-announcement/

[16] http://jrsbiodiversity.org/jrs-awards-two-grants-study-africas-wild-pollinators/

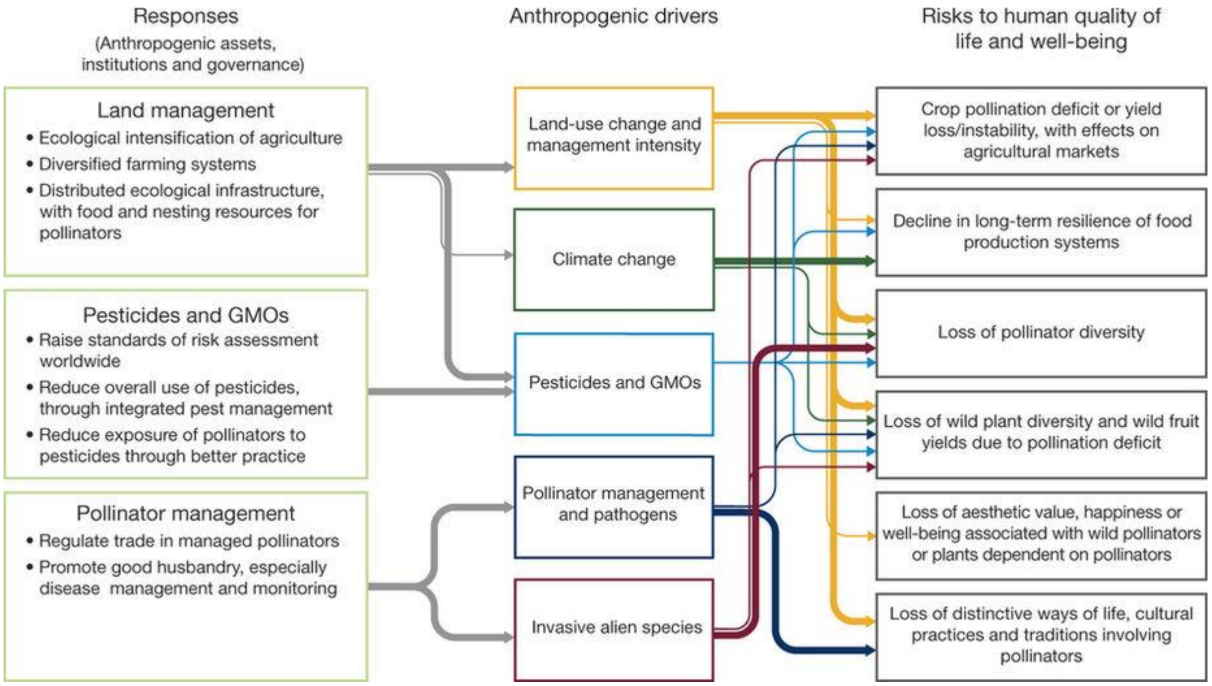

Figure 8: Drivers of pollinator decline (central boxes) relate to the key risks associated with pollinator decline (right boxes), and how these drivers are addressed by three important sets of responses (left boxes) that ultimately reduce the risks. Arrows are thick if there is clear evidence that at least one of the responses can reduce the impact of the driver on pollinators, or clear evidence that the driver generates the impact underlying the risk, at least in some circumstances. Arrows are thin if the link between response and driver, or driver and risk, is suspected or inferred by current evidence, but direct empirical evidence of it taking place is either sparse or lacking. Source: Potts et al. 2016

5.5.2 Knowledge of risks to human well-being

When it comes to knowledge of links between pollinator decline and human well-being, the main challenges for knowledge management are complexity and lack of relevant knowledge in many areas (Figure 8 above illustrates both).

In the case of invasive species and pesticides, risk assessment and tiered approaches to mitigate the risks are well established policy processes (Keeling et al. 2017; Keeling et al. 2017a; IPBES 2016b, 2016a), developed in response to complexity and large uncertainties.

Horizon scanning garners knowledge from across society as broadly as possible, to identify emerging risks that can feed into risk assessment processes (Brown et al. 2016; Roy et al. 2014; Sutherland et al. 2017). This process has identified specific pollinator-related risks, such as the potential for *Bombus terrestris* to invade in southern Africa and Australia (Sutherland et al. 2017)

5.5.3 Knowledge about appropriate responses

Knowledge about appropriate responses to mitigate or reverse pollinator decline, or reduce its impacts on biodiversity and ecosystems, comprises knowledge about i) what responses are available; and ii) how effective those responses are at delivering their specific objectives.

Information about available responses can be generated by sharing best practice guidelines, or case studies, among international networks. Many examples of sharing best practice for pollinator conservation beyond food production are provided by the existing national pollinator strategies. For example the National Pollinators Strategy for England has defined and widely shared ‘Five simple actions for pollinators’[17], and an accompanying set of case studies[18]. Further action to catalyse an international

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[17] http://www.wildlifetrusts.org/bees-needs/fivesimpleactions

[18] http://www.wildlifetrusts.org/bees-needs/casestudies

network that collates best practice guidelines and case studies to support wild pollinators would be valuable in this context.

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. This is where the second area of knowledge about appropriate responses is important. Knowledge about the relative effectiveness of responses at delivering their objectives must be rigorously synthesized and shared as widely as possible.

Systematic review, synthesis and summary of evidence relating to environmental management is rapidly developing (Dicks et al. 2014; Cook et al. 2017; Sutherland et al. 2017). A number of large-scale syntheses relating to pollinators and/or pollination (Scheper et al. 2013; Dicks et al. 2010; Humbert et al. 2012; James et al. 2016; Villemey et al. 2018; Blowers et al. 2017) are already published online and fully open access. For example, Villemey *et al.* (2018) found that globally, terrestrial verges along linear transport infrastructure (road, rail, pipelines and powerlines) host similar or richer pollinator communities than habitats away from linear transport features. These resources, and the rigorous knowledge synthesis methods that underlie them, represent a gold standard for management of scientific knowledge. They are not necessarily appropriate for synthesizing other forms of knowledge, such as Indigenous and Local Knowledge or tacit, experiential knowledge held by practitioners such as land managers and conservationists.

In addition to information about effectiveness of practices at supporting pollinators, relative impacts of pollinator-friendly versus unfriendly practices (or landscapes) on the full range of benefits provided by pollinators should be evaluated. These benefits include cultural (e.g. identity), financial (e.g., honey sales), health (e.g. pharmaceutical properties of bee products), human (e.g., employments in beekeeping), and social (e.g. beekeepers associations) dimensions. The sum and balance of these benefits are the foundation for future development and sustainable rural livelihoods. Therefore, evaluating how the full range of benefits change would be a robust approach to valuing pollinator changes in both monetary and non-monetary terms (Garibaldi et al. 2017; Garibaldi et al. 2016) and allow quantification of the synergies and trade-offs (for example, between financial and natural assets) associated with pollinator enhancement.

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