A REPORT TO THE QUEENSLAND PARKS AND WILDLIFE SERVICE ON THE EFFECTS OFF COMMERCIAL HONEY BEES ON NATIVE FLORA AND FAUNA

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1. EXECUTIVE SUMMARY

1.1 What impacts do European honey bees have on the ecology of Australian ecosystems?

Honey bees (*Apis mellifera*) have been present in Australia for almost 200 years. They are found in all habitats with the exception of extremely arid areas, and are now highly integrated into most of our natural systems.

Honey bees contribute to the pollination of invasive weeds and provide inadequate pollination to some native plants. For other native plant species, honey bees provide pollination services, either supplementing native pollinators or filling niches created by the loss of native pollinators. Native pollinators may be rare or absent due to habitat loss, or historical competition with honey bees and other introduced species.

Honey bees reduce standing crops of nectar and pollen, and therefore compete with vertebrate and invertebrate fauna for food. The evidence that this competition causes changes in the reproductive success of native fauna is equivocal.

The most important impact of honey bees on native ecosystems is competition between feral bees and native birds and animals for nest sites (Oldroyd et al. 1994). Feral bee populations are genetically distinct from commercial populations, demonstrating that feral populations are self-sustaining (Oldroyd et al. 1997, Chapman et al. 2008, Chapman et al. 2015, 2016). Therefore, exclusion of commercial beekeeping from national parks would not materially impact the density of feral bees.

1.2 What specific impacts of the European honey bee can be attributed to managed, commercial populations, as opposed to feral honey bee populations?

There is little evidence that commercial beekeeping negatively impacts native fauna beyond the impacts of feral bees. The impact of commercial honey bees relative to the background level of impact from feral honey bees arises from a sudden, temporary, increase in the number of honey bee foragers in an ecosystem. The density of feral bees in Australian ecosystems varies enormously: 0.5-150 colonies per square kilometre (Oldroyd et al. 1997, Hinson et al. 2015). The density of colonies at a commercial apiary site, assuming a foraging range of 2 km (Visscher and Seeley 1982) and 120 colonies per site, is 10 colonies per km^2. Thus, the expected increase in honey bee forager density when a commercial apiary is established varies between a negligible increase to a 20-fold increase. We suspect that areas targeted by beekeepers have higher densities of feral bees due to the floral resources available, so the expected increase in forager density as a consequence of commercial beekeeping is at the lower end of the range. Furthermore, there is an economic imperative for beekeepers to only use apiary sites when floral resources are abundant, lessening competition.

We conclude the impact of commercial bees over and above the impacts of feral bees is likely to be small in most cases. Despite the above arguments, we emphasise that this opinion is based on an absence of evidence rather than evidence of absence. Only two studies have considered the demographic effects of commercial beekeeping on Australian native fauna. One study found that native bee nests had higher survival at apiary sites relative to control sites (Schwarz et al. 1991). The other found that 23% fewer native bee nests were initiated at apiary sites (Paini and Roberts 2005), though we have some reservations about this study (Table 2).

In terms of the pollination of native and introduced flora, no Australian study has conclusively shown that honey bees (feral or commercial) have a significant negative effect on the reproductive success of native flora. Some studies have shown that honey bees are important pollinators for native plants.

1.3 Are the known ecological impacts of managed, commercial populations of European honey bees sufficient to preclude ongoing commercial beekeeping industry use of national parks in Queensland?

Among the studies that have examined the effects of commercial beekeeping on native fauna and flora, there are no compelling instances demonstrating negative effects. This is largely because the necessary ecological studies are inherently difficult to perform. An ideal study needs an area where 10 or so similar sites are chosen, each more than 4 km apart. A commercially-relevant number of honey bee colonies are then introduced at half the sites for the commercially-relevant time period for that location and then removed. Studies should be conducted over several years to assess seed set and the reproductive success of native fauna.

Existing studies fall far short of this ideal, and therefore lack sufficient statistical power to detect an ecological impact of commercial colonies should there be one, or cannot exclude confounding factors. The choice is to take the precautionary principle and argue that notwithstanding the limited evidence for impacts of commercial bees, at least some impact is likely and so beekeeping should be excluded from national parks. Alternatively, one might argue that, based on the available data, there are unlikely to be measurable impacts beyond those incurred by the presence of feral colonies, and that in the absence of compelling data to the contrary, the economic imposts on individual beekeepers and to agriculture more broadly would be unjustified.

The available data indicate that honey bees are important players in plant-pollinator interaction networks, and that their removal could have effects on natural ecosystems. A primary function of national parks is to conserve nature, and thus it would seem that the removal of foreign species would help achieve this aim. However, to do so is not without risk, as there is an absence of data on what happens when super-generalist pollinators are removed from ecosystems. These sites are accustomed to having commercial loads of honey bees when a major flowering event occurs. Banning commercial loads of bees from such sites may have no effect on pollination, a positive effect or lead to a pollination short-fall if native pollinators are not available.

2. RECOMMENDATIONS

As a result of our review we believe that on the balance of probabilities the presence of commercial colonies is unlikely to pose additional stresses on ecosystems beyond those caused by feral bees. Therefore, there is no compelling reason to exclude beekeeping on the basis of the available ecological data. Again, we emphasise an absence of evidence rather than evidence of absence.

- If a decision is made to exclude beekeeping as a result of the precautionary principle or concerns about commercial activity in parks, then we recommend that the opportunity be taken to assess the effects of commercial beekeeping on native flora and fauna in a rigorous way. We recommend that within any one park only half the sites should be removed from availability. Observations should be made over several years on apiary sites and ex-apiary sites in terms of the reproductive success of native species. If differences are found then exclusion can then be expanded to the remaining control sites. Conversely, if there are no discernible changes in ecosystem function, then bee sites could be restored.
- If beekeeping is continued it should be highly regulated. Beekeepers should be required to specify the target flora and specify the anticipated flowering period of the target species. Hives should be removed from the site outside this period;
- Beekeeper industry associations should be consulted to formulate workable regulations;
- Each park should be considered on a case by case basis to consider its conservation value and importance to beekeepers;
- Exclusion of beekeeping will cause severe economic hardship to a number of beekeepers and reduce the number or quality of colonies available for paid pollination services. Alternative sites should be identified where possible.

3. BACKGROUND

In this report we will use the term 'commercial' to denote any managed honey bee colony living in a human-made hive. Commercial colonies are distinct from feral colonies that live in natural cavities.

Commercial beekeeping plays a central role in Australian agriculture. Honey bees pollinate 53 Australian crops, and bee-pollinated crops contributed $8.53-19.97 billion to the 2014-2015 economy (Karasiński 2018). In the same year, honey, beeswax, queen bees and propolis production contributed $101 million to the national economy (van Dijk et al. 2016). There are over 13,000 registered beekeepers operating 448,000 colonies Australia-wide (van Dijk et al. 2016). Queensland is an important beekeeping state with 5,000 beekeepers and 100,000 registered hives (van Dijk et al. 2016). In Queensland, 320 beekeepers are commercial or semi-commercial, defined as 50 or more hives (Weatherhead 2018). Important horticultural industries in Queensland, particularly the melon and blueberry industries, are dependent on commercial honey bee pollination. Queensland is the least productive state on a per-hive basis (Bennik 2009, Salvin 2015, van Dijk et al. 2016).

Honey bees, *Apis mellifera*, are native to Europe, Africa and the Middle East but have been introduced throughout the world (Ruttner 1988). Honey bees were introduced to Australia in 1822 (Weatherhead 1986). Australia now has a large feral, bee population. Estimates vary from 0.1-1.5 (Hinson et al. 2015) to 50-150 colonies per km^2 (Oldroyd et al. 1997). The feral population is genetically distinct from the commercial population (Chapman et al. 2008, Chapman et al. 2015, 2016). This indicates that the feral population is self-sustaining and not dependent upon supplementation from the commercial populations (Oldroyd et al. 1997, Chapman et al. 2008, Chapman et al. 2016).

Beekeeping is considered an inconsistent use of national parks in Queensland. Despite this, commercial beekeeping currently occurs in 46 national parks, with approximately 1,100 established apiary sites. Beekeeping persists as a legacy activity, related to the transfer of State Forests to national park management under the federal Regional Forest Agreement (RFA) process. Established beekeeping sites in former State Forests were allowed to continue post-transfer to the national park system through special provisions of the *Nature Conservation Act* 1992 and other regulations (Environmental Protection Agency 2007). These provisions are due to expire on 31 December 2024, when all bee sites within Queensland national parks will lapse.

Since the RFA the beekeeping industry has consistently lobbied the Queensland government to maintain access to national parks beyond 2024. The industry argues in support of this concession as follows (Keith and Briggs 1987, Gibbs and Muirhead 1998, Australian Honey Bee Industry Council 2005, Moncur 2005, Somerville 2010, Salvin 2015):

- 80% of Australian honey production comes from native plants;
- National park apiary sites represent approximately one sixth of all apiary sites on public land (state forest and protected areas) in Queensland;
- National park apiary sites are proportionally more valuable than sites on other lands, as they contain unique floral assemblages and high-yield honey sources;
- Apiary sites in areas of high-quality native vegetation, including national parks, are required for the rehabilitation of hives stressed by the provision of pollination services, a function that is often not considered when assessing the importance bee sites to the industry;
- There is insufficient alternative land available to replace national park apiary sites. Studies conducted in the mid-200s identified approximately 18,000 ha of freehold land that could potentially support new commercial beekeeping sites. This is sufficient land to replace only 10% of the existing national park sites, and may not be available;
- Research does not sufficiently demonstrate that commercial honey bees have a significant ecological impact on plants and wildlife;
- Any impact is principally caused by feral honeybees. Commercial hives do not contribute to feral populations, as beekeepers managed their hives to minimise the risk of swarming. Even if swarming did occur, feral honey bees are already well-established through the national park system;
- Sites are used infrequently and for short periods due to the sporadic flower of native plants. Therefore, if there is any effect of commercial beekeeping use it is temporary.

4. SCOPE

The aim of this Review is to examine the literature that relates to the ecological impacts of *commercial* populations of the European honey bee on native pollinators, flora, fauna and on ecosystem integrity and function. Feral bees differ from commercial bees in that feral bees are present in an ecosystem year-round. In contrast, commercial bees are generally placed at a site for short periods, usually 4-6 weeks but up to 3 months (Seeman 1994, Somerville 2010, Salvin 2015). Obviously, commercial colonies do not compete with native fauna for nest sites. Thus, the impacts of commercial bees are likely to be substantially different to the impacts of feral bees.

Unfortunately, the vast majority of ecological work on the impacts of honey bees on native flora and fauna considers the impacts of feral bees rather than commercial bees. This is, most studies have not increased the density of honey bees to 10 colonies per km^2, which is the density generated by commercial bee sites. Rather, most studies are based around field observations of interactions of what are presumably feral honey bees with native fauna and flora.

Much of the available literature on ecological impacts of honey bees concerns Africanized bees in the Americas. Africanized honey bees are hybrids of European and African subspecies of *A. mellifera* which became established in São Paulo, Brazil, in the 1960s (Winston 1992). European honey bees (*A. m. ligustica, A. m. mellifera* and other subspecies) perform poorly in tropical Latin America because they are unsuited to the climate. In an attempt to breed bees that were more appropriate for the Brazilian environment, *A. m. scutellata*, a subspecies that evolved in the tropical savanna region of southern Africa, was introduced to Brazil in 1956. *A. m. scutellata* hybridized with the extant European honey bees that had been naturalized since the 1700s. The resulting hybrids have spread through much of South and North America (Winston 1992). Africanized bees are much more aggressive and produce more swarms than do European subspecies, and are unsuitable for modern beekeeping (Winston 1992).

Concerns over a biological invasion of the United States by Africanized bees led to a massive research effort in the 1980s to better understand the ecology, genetics and behaviour of feral Africanized bees. We have surveyed this literature because it concerns the impacts of honey bees on tropical and subtropical ecosystems, whereas most other studies are from temperate regions. However, we caution that the literature on feral Africanized honey bees is only tangentially relevant to the ecological impacts of *commercial European* honey bee colonies in Queensland national parks.

Various pests and diseases have been, or have the potential to be, transferred from commercial bee populations to feral and native bee populations (Fürst et al. 2014, Evison 2015, Goulson and Hughes 2015). Such studies have not been included within this review as we consider that the presence of commercial colonies in national parks does not pose any additional disease risks to native fauna beyond that posed by the presence of feral honey bee colonies.

5. LITERATURE REVIEW

5.1 Relevant quality research is lacking

Numerous reviews (e.g. Paton 1993, Sugden et al. 1996, Butz Huryn 1997, Manning 1997, Schwarz and Hurst 1997, Goulson 2003, Paini 2004, Mallinger et al. 2017, Wojcik et al. 2018; Table 1) have strongly criticised the literature that explores the ecological effects of honey bees on native plants and wildlife (Table 2). In summary, few studies have measured the effects of commercial bees on native fauna and flora under real-life conditions. That is, there are very few studies where changes in native ecosystems have been documented as a consequence of the presence of commercial honey bees at commercially-relevant densities. Fewer still have done so with control and experimental sites and appropriate replication.

5.2 Competition with native fauna

Honey bees have a number of traits that make them efficient foragers and this raises concerns that they outcompete native nectar and pollen foragers. Honey bees communicate the distance, direction and quality of food sources to their nestmates (Winston 1987). They therefore quickly recruit a large forager force to high-quality resources (Seeley et al. 1991). Honey bees thermoregulate their nest precisely, and can therefore commence foraging earlier in the day than most native insects (Jones et al. 2004). In addition, with their relatively large body size, honey bees are able to forage at lower temperatures than many native insects. Body size and nest thermoregulation means that honey bees have the potential remove substantial resources before native insects commence foraging (Paton 1993).

5.2.1 Abundance correlations and resource overlap

Competition between honey bees and native fauna native fauna is generally measured via niche overlap studies, or by changes in native fauna abundance in areas where honey bees have been introduced. Below we provide a critical evaluation of these methods before consideration of individual studies.

Many studies (Table 2) are ‘correlational’. In these studies researchers set out a number of plots in the field, and then make counts of the number of honey bees, other insects and birds over a period of hours or days. Various inferences are then made from the abundance of honey bees and the native species of interest across sampling units (Sugden et al. 1996, Goulson 2003, Paini 2004, Stout and Morales 2009, Wojcik et al. 2018; Figure 1).

A negative correlation between the abundance of two species across sampling units can be interpreted as evidence for competitive displacement. Conversely, a negative correlation may be interpreted evidence that two species under consideration have different resource requirements and are not competitors (Ludwig and Reynolds 1988) (Figure 1).

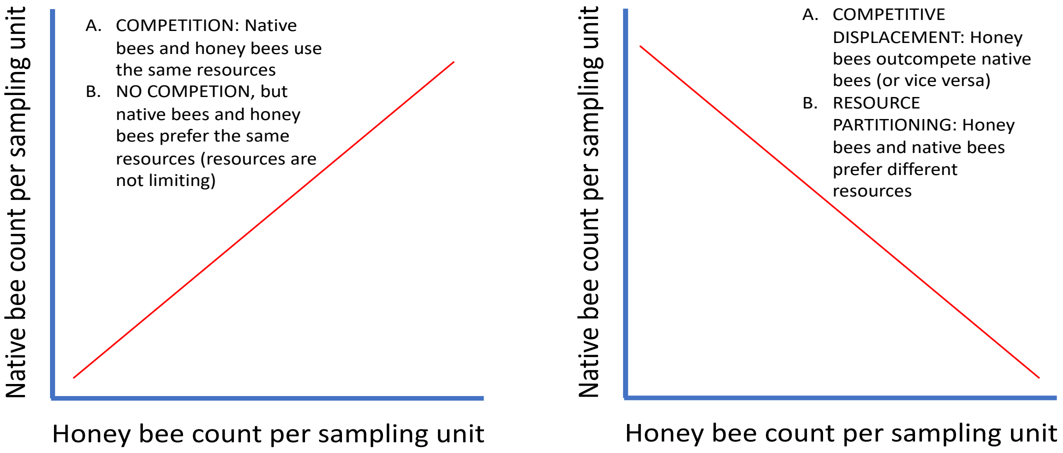

Figure 1. Hypothetical correlations between honey bee and native bee or bird abundance. Positive and negative correlations can be interpreted to give the opposite conclusions

A positive correlation in abundance across sampling units may be interpreted as evidence for inter-specific competition, or conversely, of resource overlap (Ludwig and Reynolds 1988) without any competition (Figure 1).

A lack of association in the abundance of two species might be interpreted as evidence that two species do not affect each other. However, it is quite possible that there is competition, but the study lacks sufficient statistical power to detect it.

There are other ways in which a correlation can be misleading. For example, if removal of resources by honey bees means that native species must visit more flowers over a longer period to collect the same quantity of resource, the number of native foragers counted could increase (Paton 1993). Under this hypothetical example, what is actually competition leads to a positive correlation that could be interpreted as evidence of resource overlap without competition.

We conclude from the above that without manipulation of the study system, it is difficult to come to definitive conclusions about species interactions, because a correlation in species abundance can be interpreted in ways that can produce opposite conclusions. We therefore take the view that studies that merely demonstrate that honey bees and native species use the same floral resources are of limited value for demonstrating competition.

Unfortunately, despite the inherent problems of correlational studies, most ecological studies of native fauna-honey bee interactions are based around field observations of foraging native insects and (presumably feral) honey bees, without experimental manipulation (Table 2).

5.2.2.1 Beyond correlation: selected Australian studies

The following studies examined whether a nectar resource remained available to native fauna in the presence of honey bees. These studies show that temporal variation in resource use or excess floral resources can reduce competition between honey bees and native species.

Honeyeaters foraged on Yellow mallee *Eucalyptus costata* (formerly *incrassata*) (Bond and Brown 1979) and one-sided bottlebrush *Calothamnus quadrifidus* (Collins et al. 1984) earlier in the morning than did honey bees. By the time honey bees commenced foraging the majority of the day's nectar had been consumed by the native species. Here the potential for honey bees to compete with honeyeaters was reduced because native species could utilize the available resource before honey bees had started foraging.

Honey bees commenced foraging on *Eucalyptus costata* before native bees, but nectar supplies remained at midday (Horskins and Turner 1999). This suggests that nectar was not a limiting resource in this ecosystem.

Two studies have demonstrated nectar depletion in the presence of honey bees:

Leatherwood (*Eucryphia lucida*) is a major resource for beekeepers in Tasmania. It produces nectar both during the day and the night. *E. lucida* flowers were depleted of nectar at apiary sites whereas 50% of nectar remained at control sites at 6pm (Mallick and Driessen 2009).

Desert Banksia, *Banksia ornata*, is a major resource for beekeepers in South Australia and Western Victoria. Nectar availability from the desert Banksia was significantly reduced at sites 1km from an apiary versus control sites with no apiary (Paton 1999). However, this depletion was only observed in only one year of a 3-year study, demonstrating that resource abundance varies in time and space.

5.2.1.2 Monolectic species

Native species that are monolectic (i.e. confine their foraging to a single plant species), cannot or do not switch to other resources in response to resource depletion. If resources are limiting then monolectic species are vulnerable to the presence of an exotic competitor. However, we found no such examples in the literature. Seeman (1994) is of the view that Australian fauna are unlikely to be monolectic due to the low frequency and high irregularity of flowering of many native plant species. Monolecty generally arises due to the absence of co-flowering congeners, and is rare (González-Varo et al. 2016).

5.2.1.3 Summary

There is potential for honey bees to remove significant floral resources. In some instances, native foragers are able to remove resources before honey bees commence foraging, or significant nectar and/or pollen is left even in the presence of honey bees. Whether or not resource removal by honey bees reduces the reproductive output of native wildlife can only be determined from direct assessment, not inference from correlations.

5.2.2 Impacts of honey bees on the reproductive success of native bees and birds

Studies that demonstrate that the presence of honey bees impacts the reproductive success of potentially competing native species are the most valuable. Reductions in the number and quality of brood, or the amount of food stored by native bees as a consequence of the presence of honey bees, clearly demonstrate that competition is taking place.

5.2.2.1 Australian bee studies

Two Australian studies have examined the consequences of the presence of commercial honey bees on the reproductive success of native species. These studies used commercially-relevant numbers of honey bee colonies and featured good experimental design with adequately replicated experimental and control sites.

Paini and Roberts (2005) compared nest abundance and offspring mass of the native bee *Hylaeus alcyoneus* at 7 sites with 100 commercial honey bee colonies and 7 (control) sites without commercial honey bee colonies in a beekeeping reserve in Western Australia. The authors report 23% fewer *H. alcyoneus* nests at apiary sites compared to control sites, suggesting competition.

Unfortunately, some apiary sites were considered control sites until honey bee colonies were moved in, presumably by commercial necessity, confounding treatment with temporal factors. The differences between control and treatment sites were mainly driven by differences in the first year of this 2-year study. During the first year, only 30 *H. alcyoneus* nests were established across all 14 sites. Due to the variation in the dates at which apiary sites were occupied, it is unclear how many apiary sites were actually occupied by honey bee colonies during the first two observation periods, when the most *H. alcyoneus* nests were established in that year.

There was no effect when years were considered separately, including in the second year, when 87 nests were established. There was no difference in the number of eggs produced per nest or the weight of emerging offspring, but the statistical power to detect an effect of treatment was low. Despite being relatively well designed, the ability to detect an effect of honey bees on population demography was low, highlighting the difficulties of such endeavours.

In a study in South Australia, using 100 commercial colonies per site, there was no effect of honey bee presence on any measure of reproduction per nest for the native bee species *Exoneura bicolor* and *E. nigrihirta* (Schwarz et al. 1991). However, *Exoneura* nests had higher survival rates at sites with apiaries (n = 4) relative to control sites (n = 4). The authors attributed this surprising result to predator satiation, but gave no evidence for this.

Three more Australian studies considered effects of low honey bee colony densities on the reproductive success of native bees. Spessa (1999) found no difference between apiary sites (n = 4 sites; 6 colonies per site) and control sites (n = 4) in pupal weight, number of brood, survival of brood, colony size or abundance of *Amphylaeus morosus* nests over two years. In one of the years more nests were established at apiary sites.

Paini et al. (2005) assessed nest occupation and the number and weight of offspring of a native bee species (Megachile sp. 323) at a beekeeping reserve in Western Australia over 3 months, at a time when the area is not usually used by beekeepers. The aim was to establish if there is competition with feral honey bees, so honey bee colonies were placed at low density (2 colonies per treatment site). There were 6 control and 5 treatment sites. The number of nests, number of offspring, offspring mass, number of dead progeny, number of nests in diapause and the sex ratio did not differ between treatment and control sites. Again, the statistical power available to detect biological differences of moderate effect size was low.

In south east NSW, more native reed bee (*Exoneura asimillima*) nests were established near one experimental site where honey bee colonies had been added, (10-29 hives, October-March in each of two years) than at 3 control sites, and there was no difference in native bee nest survival between the single apiary and control sites (Sugden and Pyke 1991). Native bee reproductive output per nest was higher at the apiary site than control sites in the first year, but this was likely due to a difference in sampling period. The lack of replication of the apiary site in this study renders the results questionable.

5.2.2.2 An Australian bird study

Paton (1993) suggested that male New Holland honeyeaters increased their territory size and changed their foraging behaviour on *Callistemon rugulosus* at a site in South Australia in response to increased honey bee colony abundance. We note that the male's territory size was estimated based on observations of 10 males for 5 hours. Five of the birds had territories near an apiary of 10 hives, five birds were observed distant to the apiary. Given the short-time frame of observations it is unclear if temporal effects can be differentiated from treatment effects.

5.2.2.3 International studies where honey bees are not native

Thomson (2004) found that the number of offspring produced by native *Bombus occidentalis* colonies in California did not vary with distance from a small number (2-3) of honey bee colonies. The authors suggest that reproductive success (number and proportion of gynes produced relative to the number of males and gyne size) of *B. occidentalis* increased with increasing distance from honey bee colonies, though the differences disappeared when *Bombus* nests damaged by wax moth were excluded from the analysis. The number of males produced is poorly estimated from the number of males leaving the nest during forager observations, with the authors noting that there is a low probability of observing male dispersal flights.

Roubik (1983) did not find any effect of a small number of Africanized honey bee colonies on brood production or food storage of two *Melipona* species in French Guiana. However, there were no control sites and poor replication.

Abe et al. (2008) examined whether resource competition with honey bees, habitat loss, agricultural chemicals or predation affected the distribution of 132 native bee species on 9 Japanese islands that differed in the presence or absence of these factors. While previous studies had shown that competition with honey bees was a factor in native bee decline (Kato 1992, Kato et al. 1999), after controlling for habitat loss, there was no relationship between honey bee abundance and native bee numbers (Abe et al. 2008). Note, however, that an Asian honey bee species, *Apis cerana*, is indigenous to these islands, and so other native bee species should be well-adapted to the presence of honey bees.

5.2.2.4 International studies where honey bees are native

Elbgami et al. (2014) measured colony weight and the number of males and new queens produced by 5 bumble bee (*Bombus terrestris*) colonies placed 5m compared to 1km from a 50-hive apiary in England. The experiment was replicated 2 years later. *B. terrestris* colonies close to the apiary gained less weight and produced smaller queens in both years. The number of males produced per *B. terrestris* colony did not differ between the control and experimental sites, but colonies close to the apiary produced fewer queens in one year. The author's assert that the surrounding forage at the near-apiary site and the distant-from-apiary site did not differ. However, without replication within years it is possible that differences in resource availability between the two sites contributed to the findings. *B. terrestris* colonies tend to specialise in producing either queens or males. The sex favoured is correlated with the length of the mother queen's diapause (hibernation period) (Duchateau et al. 2004). The extreme sex-ratio biases found in colonies near the apiary in one year (11.9:1) may be spurious due to the small number of colonies used. The length of diapause of the queens used for the study was not reported.

In Germany, Hudewenz and Klein (2013) investigated the number of nests of a ground-nesting bee at different distances from the nearest honey bee colony and the number of nests of stem-nesting bee species at sites with or without honey bees. The number of ground nests did not differ with distance from the nearest honey bee colony. Sites with and honey bees had a smaller number of stem nesting species, but the authors argue that the species diversity was best explained by the distance to woody habitat. There was no difference in the total number of stem nests between sites with and without honey bee colonies.

Again in Germany, Hudewenz and Klein (2015) compared the number of brood cells constructed by red mason bees *Osmia bicornis* in flight cages with no honey bees, a small colony of 100 honey bees or a small colony of 300 honey bees. The honey bee colonies had a queen that continued to produced brood throughout the experiment. The honey bees were fed sugar patties throughout the experiment, which may have reduced competition, but likely spurred brood production and thus a greater demand for pollen with which to feed the brood. More *Osmia* brood cells were constructed in cages without honey bees. The experiment is highly artificial. We question whether a honey bee colony comprising 100 or even 300 workers is a viable unit, especially in a cage.

Kühn et al. (2006) found no change in the number of brood cells produced by leafcutter bees (*Megachile lapponica*) before, during and after the introduction of 15 honey bee colonies. The experiment was performed with two temporal replicates of 14 days in Germany. More long-term studies with greater replication are needed to verify these results.

Everts (1995) found that reproductive success of the leafcutter bee *Megachile rotundata* in Germany was higher at a site with no apiary compared to a site without an apiary. Due to the lack of site replication, other factors cannot be ruled out.

Steffan-Dewenter and Tscharntke (2000) found no correlation between the number of honey bee colonies (3-65) and the number of other native bee species occupying trap nests or the number of offspring produced at 15 sites in Germany. Likewise Pechhacker and

Zeillinger (1994) found no difference in occupation rates of trap nests by native bees with distance (100m to 1.5km) from an apiary in Austria.

Goulson and Sparrow (2009) compared the thorax width of up to 10 foraging workers of four native *Bombus* species in 10 areas with honey bees and 10 without honey bees in Scotland. The authors found that workers of all *Bombus* species were smaller in areas where honey bees were present. This result suggests that competition for resources with honey bees resulted in poorly-fed, smaller offspring. Worker size is positively correlated with the length of time since the emergence of the first worker (Shpigler et al. 2013). Therefore, if colonies differed in their development, and more than one worker was sampled from a colony, this may have skewed the results. More work is required to confirm these results.

5.2.5 Summary

Of the 16 studies that have examined the reproductive success and/abundance of native bees in the presence/absence of honey bees, 8 found negative effects of honey bees, 6 showed no effect and 2 showed what might be regarded as positive effects. On this basis one might conclude that there is evidence that the presence of honey bees has adverse effects on native bee populations. However, many studies suffer from a lack of controls and poor replication. Even the best-performed studies lacked sufficient statistical power to detect effects for some measures. For 13 of the 16 studies, the density of honey bee colonies was far less than commercially-relevant densities.

We conclude that there is little compelling evidence that honey bees have detrimental effect on the reproductive success of native fauna at commercially-relevant colony densities beyond the effects of low densities of honey bees.

5.2.3 Aggressive interactions

Some researchers have noted that native bee species are ‘disturbed’ on flowers by honey bees. These interactions are sometimes described as ‘aggressive’ but the actual behaviours are infrequently described and subject to interpretation. No attempt has been made to determine if there was a difference in visitation length between foragers who were and were not ‘disturbed’ by honey bees, or the proportion of foragers that were and were not ‘disturbed’ – were these rare events or only occasional?

5.2.3.1 Australian studies

Gross and Mackay (1998) noted that in 91% of 153 interactions between honey bees and native bees on *Melastoma affine* the native bee was ‘disturbed’. Kinds of disturbance reported included 1) native bees hovering over but not landing on a flower that was occupied by a honey bee (71% of cases). Honey bees in this situation landed 91% of the time; 2) foraging native bees leaving the flower after a honey bee alighted (75% of cases) compared to honey bees leaving when a native bee alighted (6% of cases); and 3) both bees departing (17% of cases). They also report several aggressive interactions in which honey bees pulled native bees from stamens.

Stingless bees (*Tetragonula* spp) were not displaced by honey bees, but small Hylaeine bees flew off when touched by honey bees (Williams and Adam 1997). Taylor and Wheland

(1994) report that honeyeaters are deterred from feeding on Grevillea when honey bees are in high abundance, but they do not quantify or describe this behaviour.

5.2.3.2 International studies

Africanized bees vibrated their wings to repel aggressors (*Trigona*) on feeders in French Guiana, but rarely displayed aggression and only to *Melipona fulva*, polybiine wasps and other honey bees (Roubik 1980). In the main, native social bees outcompeted Africanized bees at feeders (Roubik 1980). Honey bees occasionally bit and stung other honey bees, but the nature of aggression toward other species is not described (Roubik 1980). Africanized honey bees also ‘tackled’ stingless bees in Mexico (Cairns et al. 2005). Africanized bees are known to be more aggressive than European subspecies (Winston 1992).

Dead *Bombus* (8 species) and other native bees were found in honey bee colonies in New York and Arizona, presumable killed while they tried to rob the honey bee colony (Morse and Gary 1961, Thoenes 1993). In New Zealand aggression by honey bees toward large Diptera was reported, but undescribed (Bennik 2009). Attacks by honey bees on other pollinators of *Salvia apiana* in California were also undescribed and unquantified (Ott et al. 2016).

While not aggressive, honey bees have been reported to steal pollen from the bodies of *Megachile montivaga, Melissodes desponsa, Bombus impatiens* (Jean 2005), *Bombus pennsylvanicus* (Laroca and Winston 1978), and *Diadasia enavata* and *Halictus ligatus* (Thorp and Briggs 1980).

5.2.3.3 Summary

Honey bees may be aggressive toward some interspecifics, but only when the interspecific attempts to rob their colony or when the two species forage on the same artificial feeder. Interactions while foraging on flowers are rarely aggressive.

5.3 Pollination

Several studies have quantified the contribution of honey bees to native plant pollination. Quality studies quantify the number of pollen grains deposited by different pollinators or the number of fruit/seeds resulting from single visits by pollinators to individual flowers. This is done by excluding pollinators from flowers until they open, at which time a single pollinator is allowed to visit the flower. The flower is then bagged again, then excluding all other pollinator visits. The number of pollen grains removed or deposited can then be quantified, and fruit, nut or seed set can be determined. In contrast, several studies merely count the number of pollinator visits to a flower and correlate the number of visits with seed set or fruit production. Clearly such studies cannot determine which visitor(s) were responsible for pollination. Nonetheless, we cover these studies below for completeness.

5.3.1 Pollination of native plants

5.3.1.1 Australia studies that considered single visits

Melastoma affine is a cosmopolitan pioneer shrub that is found from India to south east NSW. It only produces pollen (not nectar) and is not a target species for beekeepers. Given its Asian range, it is likely that *M. affine* co-evolved with other honey bee species. (There are 10 Asian species of *Apis*, (Oldroyd and Wongsiri 2006)). Gross and Mackay (1998) found that honey bees deposited fewer pollen grains on the styles of *M. affine* than did native bees. Flowers that were last visited by honey bees had 415 pollen grains compared to 1148 when the native bee *Lestis bombylans* was the most recent visitor. Flowers pollinated by a single honey bee set fruit in 10% of cases, compared to 35-53% when pollinated by one individual from any of four species of native bee. Fruit set was 60-71% where a honey bee had been the last visitor and 81-88% for flowers where a native bee was the last visitor. Since most flowers are visited numerous times the actual effect of the presence of honey bees on pollination rates may be low, though honey bees are more commonly the last visitor.

Dillwynia juniperina is a widespread native shrub of dry sclerophyll forests in NSW and Victoria that requires bee pollination. Gross (2001) found no difference in fruit set between native bee- and honey bee- pollinated *D. juniperina* flowers. They further suggest that honey bees may be necessary to augment pollination. Indeed, honey bees are the sole pollinator of *D. sieberi* in winter (Lomov et al. 2010).

The Grey Mangrove, *Avicennia marina*, is a widespread mangrove species that is found from Africa to Australia and New Zealand. It likely co-evolved with honey bees within its African, Middle Eastern and Asian range. Hermansen et al. (2014) report that that only honey bees were seen foraging on *A. marina* at Australian sites. Honey bees removed 96% of pollen in a single visit. The authors suggest that honey bees have displaced one or more native pollinators. However, it is possible that the native pollinators of *A. marina* were lost due to habitat loss or some other cause, rather than by competition with honey bees, or that *A. marina* does not require insect pollination to set fruit.

Tasmanian blue gum, *Eucalyptus globulus*, is native to Tasmania and Victoria, and is a widely-planted forestry species throughout the world. Hingston et al. (2004b) found that flowers of *E. globulus* produced fewer seeds after a single visit by honey bees than following a visit from the swift parrot, *Lathamus discolour*. No seeds were set after single visits by native bees. To assess parrot pollination, caged parrots were taken to the field. *In situ* tree branches were then presented to the parrots through a door in the cages. The relevance of the findings to the field situation is questionable.

A second study of blue gum pollination (Hingston et al. 2004a) suggested that insects of a similar size as honey bees contribute less than 20% of the maximum seed-set. However, the apertures to the experimental cages were of a size that resulted in pollen being knocked off the pollen basket of foraging honey bees and thus, the study may have underestimated the contribution of honey bees to pollination.

The swift parrot is listed as an endangered species. If the swift parrot is a major pollinator of blue gums in Australia, then reductions in the population of swift parrots may increase the need for honey bee pollination. We also note that blue gums are invasive in South Africa and California where parrot pollination is unlikely, and honey bee pollination is likely.

5.3.1.2 International studies that considered single pollinator visits

Honey bees do not differ from the average non-*Apis mellifera* pollinator in pollination effectiveness. Honey bees are the most important pollinator globally, based on a review of 32 studies, of which two were Australian studies (Hung et al. 2018).

Honey bees improved pollination of the wildflower *Cistus salvifolius*, but lowered seed set in *Cistus crispus* in Spain (Magrach et al. 2017).

5.3.1.3 Australian studies that estimated the effectiveness of honey bee pollination

Jervis Bay Grevillia (*Grevillia macleayana*) has a narrow distribution on the South Coast of NSW. Vaughton (1996) attributed 30% of pollen removal from *Grevillea macleayana* to (presumably feral) honey bees, while England et al. (2001) cite unpublished data indicating that honey bees remove 90% of pollen from this species. Fewer pollen grains were deposited on bird-excluded flowers than open flowers (Wheland et al. 2009). Fruit set was reduced by >50% when birds were excluded (Vaughton 1996). Bird-exclusion cages may reduce insect foraging, somewhat confounding these experiments (Wheland et al. 2009).

The hair-pinned Banksia is a woody understory shrub found in heathland from Victoria to North Queensland. Vaughton (1992) investigated *B. spinulosa* pollination in NSW. Honey bees were uncommon early in the season, but became increasingly common later in the season. Fruit set was higher in the late season, though this could be due to seasonal effects rather than the presence of honey bees. Late-season flowers that were left open achieved higher fruit set than bird-excluded flowers, suggesting that honey bees are less efficient pollinators. Nonetheless, fruit set did not differ between treatments.

Brachyloma ericoides is a heath that is wide spread in the south east of South Australia and south-west Victoria. Honey bees contact the anthers and stigma when foraging on B*rachyloma*. Capsule production was lower when birds were excluded (Celebrezze and Paton 2004).

There was a positive correlation between the number of honey bee visits and fruit production in *Callistemon rugulosus*. However, fruit production was less than when birds were allowed to forage (Paton 1993). Fruit set and pods per fruit were also lower in bird-excluded than open *Correa reflexa* (Paton 1993). There was no difference in nut set between honey bee-excluded and non-excluded macadamia (Heard 1994).

There was no difference in pollen deposition or seed set of *Eucryphia lucida* at apiary compared to control sites, but fruit set tended to be higher at apiary sites (Mallick and Driessen 2009). Caged *Banksia menziesii* had lower fruit production than open flowers (Ramsey 1988). However, honey bees deposited more pollen than is required for fruit set, pointing to other factors, such as pollen removal by beetles.

Desert Banksia, *Banksia ornate,* provides a major honey crop in South Australia and western Victoria, and has been exploited by beekeepers for over 70 years. Seed production of *B. ornata* was higher at sites with honey bees relative to control sites (Paton 1999).

5.3.1.4 International studies that estimated the effectiveness of honey bee pollination

Fruit and seed set of a New Zealand native *Phormium tenax* (Murphy and Robertson 2000) and *Alseuosmia macrophylla* (Pattemore and Anderson 2013) were higher at sites with bird visitation, suggesting that honey bees were less efficient pollinators than native birds.

In the United States the number of floral visits, 87-91% of which were by honey bees, was correlated with seed set of *Triteleia laxa* (Chamberlain and Schlising 2008). Honey bees transferred little pollen to *Impatiens capensis* (Wilson and Thomson 1991).

do Carmo et al. (2004) found a negative correlation between the frequency of honey bee visits to male *Clusia arrudae* flowers and seed production by female flowers that opened on the same day. Native *Eufriesea nigrohirta* foraging for resin on flowers that had previously been visited by honey bees carried less than 0.1% of the pollen grains that they did when foraging on an unvisited flower. Honey bees discard pollen from *Merremia aegyptia* in Brazil, but their contribution to pollination was not quantified (Pick and Schlindwein 2011).

In Spain two plant species (*Echium wildpretii* and *Spartocytisus supranubius*), species that are highly visited by honey bees, produced more fruit, but fewer seeds per fruit when honey bee colonies were present. Three other plant species (*Erysimum scoparium, Scrophularia glabrata, Adenocarpus viscosus*) are less-frequently visited by honey bees (Valido et al. 2019). At the level of the plant rather than individual fruit, it is likely that the number of seeds produced by *E. scoparium* and *A. viscosus* was unchanged by honey bee foraging.

5.3.1.5 Other Australian studies

Morning iris, *Orthrosanthus multiflorus*, is native to Kangaroo Island. Honey bees likely provide significant pollination to *O. multiflorus* as plants bloom for one day only, and there are not sufficient native bees to visit all flowers (Paton 1993).

Diospyros pentamera and *Neolitsea dealbata* are rainforest trees distributed from NSW to north Queensland. Honey bees are unlikely to pollinate *Diospyros pentamera*, but may pollinate *Neolitsea dealbata* (House 1989).

Honey bees are unlikely to pollinate *Grevillea* as honey bees are too small (Taylor and Wheland 1994). Honey bees contact the anthers or stigmata of *Calothmnus quadrifidus* on 42% of foraging visits (Collins et al. 1984). Only pollen-foraging honey bees (79% of visits) contacted the pollen presenters of *Grevillea macleayana* (Vaughton 1996, England et al. 2001). Honey bees collected previously-deposited pollen from *Melastoma affine*, never gathering pollen from the anthers (Gross 1993). Honey bees contact the stigmas of *Eucalyptus costata* (Horskins and Turner 1999).

5.3.1.6 Other international studies

In Peru honey bees use holes in *Sparattosperma leucanthum* made by native robber-bee species to rob nectar. They may accidentally pollinate the flowers, but this has not been quantified (Polatto et al. 2012). In Brazil, honey bees removed all the pollen from native passionfruit, but as they rarely touch the stigmas, they are unlikely to pollinate this species (Yamamoto et al. 2012). Honey bees deposit more pollen than native bees in *Metrosideros polymorpha* in Hawaii (Junker et al. 2010).

5.3.1.7 Summary

Australian and international studies suggest that pollination effectiveness can be lower when performed by honey bees relative to native species. No studies show that the presence of feral honey bees alters the reproductive success of native plants, despite their lower pollination effectiveness. That is, frequency of visits may compensate for reduced pollen transfer per visit.

There is limited evidence that a small number of native plants may now be dependent upon honey bees for pollination. These studies also suffer from a lack of evidence for changes in reproductive success.

5.3.2 Hybridization of native plants

Inter-specific plant hybridization is a relatively common event in nature (Grant 1981). Concerns have been expressed that honey bees increase the frequency of hybridisation of native plants, resulting in sterile offspring (Matthews 1984, Paton 1993, Butz Huryn and Moller 1995, Butz Huryn 1997). Honey bees have strong species fidelity when foraging (Free 1963, Percival 1974, Butz Huryn 1997), reducing the likelihood of inter-specific pollination relative to the foraging activities of native birds and bees. We therefore suspect that honey bees do not increase the frequency of plant hybridization over natural levels. Nonetheless, as several studies have claimed this (or at least claim that honey bees carry pollen from more than one plant species). We briefly survey these studies below.

5.3.2.1 Australian studies

Both native insects and honey bees have been recorded carrying pollen of more than one plant species when foraging on the following plant genera: *Persoonia* (Bernhardt and Weston 1996), *Acacia* (Bernhardt and Walker 1985, Bernhardt 1987), *Neolitsea dealbata* and *Diospyros pentamera* (House 1989). Honey bees carried a large proportion (88-99%) of a singular species of two mangroves (*Avicennia marina* and *Aegiceras corniculatum*) and a salt marsh plant *Sarcocornia quinqueflora* (Hermansen et al. 2014).

5.3.2.2 World-wide studies

Honey bees were more likely to carry mixed pollen loads than native bees in Hawaii (Miller et al. 2015), and interspecific pollen transfer was largely driven by honey bee visitation patterns (Johnson and Ashman 2019). However, there is only one native bee genus in Hawaii, Hylaeus, consisting of at least 63 species. Honey bees did not contribute to hybridization of milkweed in the USA, *Bombus* being the primary agent (Stoepler et al. 2012).

5.3.2.3 Summary

Hybridization requires cross pollination by closely-related plant species that are nonetheless able to form a viable hybrid. Butz Huryn (1997) argues that honey bees are unlikely to increase plant hybridization due to their strong floral constancy in comparison to numerous other taxa and we support this view. We conclude that it is extremely unlikely that commercial honey bees would increase the frequency of plant hybridization above background levels in any circumstance.

5.3.3 Outcrossing of native plants

Only 10-15% of plant species are predominantly self-fertilizing (Goodwillie et al. 2005). Plants that are self-compatible may suffer from inbreeding depression (Wright et al. 2013), while plants that are not self-compatible are severely affected by a lack of suitable pollination. There has been little work comparing the distances and frequency at which honey bees and native pollinators move between plants (Richardson et al. 2000).

5.3.3.1 Australian studies

Honey bees moved twice as frequently as birds between plants of *Brachyloma ericoides*, but when birds do change plants they tend to move further away (Celebrezze and Paton 2004). Outcrossing was lower when birds and mammals were excluded from *Grevillea macleayana* (England et al. 2001), likely because they move further between plants (Wheland et al. 2009). A site that had more frequent bird visitation had higher outcrossing, however this site was of better quality that the other two sites, which may explain some of this variation (Wheland et al. 2009). Honey bees are less likely to move between plants of *Dillwynia sieberi* than the native bees *Lasioglossum clelandi* and *Megachile chrysopyga* (Lomov et al. 2010).

5.3.3.2 International studies

Honey bees and bumble bees mostly deposited self-pollen on *Phormium tenax* in New Zealand, but outcrossing rates did not differ between populations where exotic social bees were common versus uncommon (Howell and Jesson 2013). Honey bees visited 1.5 times more flowers within plants of *Dicerandra immaculata* than native pollinators in the USA, but the authors did not investigate whether this led to higher rates of inbreeding (Richardson et al. 2016). Honey bees facilitated geneflow between fragmented populations of *Dinizia excelsa* in Brazil (Dick 2001, Dick et al. 2003). Honey bees tended to visit a single flower or a single branch of the cactus *Carnegiea gigantea* in Arizona, while native bees visit different trees (McGregor et al. 1959). Similarly, honey bees visited more flowers on the same *Echium wildpretii* plant than did native bees in the Canary islands (Dupont et al. 2004). There was no difference between honey bees and *Osmia cornuta* in the frequency of inter-row flights in pears in Spain (Muñoz et al. 2014).

5.3.3.3 Summary

Outcrossing rates within a species may be elevated, reduced or unchanged by honey bee pollination, depending on the system.

5.3.4 Pollination of introduced weeds

Opinions are mixed on whether or not honey bees prefer foraging on introduced plants to native plants. If honey bees preferentially forage on introduced plants, this may reduce competition with native fauna for native plant resources (Aizen et al. 2014, Aslan et al. 2016). Second, if honey bees forage on invasive weeds this may contribute to increased dominance of these problem plants in our ecosystems (Beard 2015). However, native insects and birds may also contribute to the pollination of weedy species (Butz Huryn and Moller 1995). Again, studies that quantify pollination after single controlled visit provide higher-quality information than studies that simply correlate the number of bee visits with seed set.

5.3.4.1 Single-visit studies in Australia

Scotch broom, *Cytisus scoparius*, is an important weed of high altitude national parks in NSW. 84% of honey bee visits resulted in fruit set in *C. scoparius*, while native bees did not pollinate this plant (Simpson et al. 2005). The number of seeds per fruit did not differ between open flowers and those that received a single visit by honey bees.

5.3.4.2 Correlational studies in Australia

Seed set of invasive *Lantana camora* (Goulson and Derwent 2004) was higher at sites that had more honey bee visits. This study can be criticised in many ways. First, sites were across a 2000 km transect from Brisbane to Daintree. There was a strong latitudinal gradient in the incidence of honey bees and fruit set, and the authors conclude that the causative factor was foraging honey bees. However, while sites may have varied in the number of honey bee visitors to flowers, they undoubtedly differed in ecological variables other than the abundance of honey bees. More importantly, the number of honey bees observed was very low at all sites and rare or absent at the majority of sites. Across 29 sites, a total of 11,600 *L. camora* inflorescences were observed but only 171 honey bees were recorded foraging on those inflorescences across two months of observations. In other words, honey bees and native bees mostly ignore *L. camera*, especially in the tropics north of 20^{o}S.

Honey bee site visitation was correlated with seed set in invasive *Lupinus arboreus* in Tasmania (Stout et al. 2002), but only at sites where flower visits were dominated by invasive *Bombus terrestris* were excluded. A similar study found no correlation (Goulson and Rotheray 2012). Native insects were unlikely to pollinate *L. arboreus* (Stout et al. 2002).

5.3.4.3 Correlational international studies

Introduced honey bees and bumble bees are likely to be the only pollinators of the invasive Banana Vine, *Passiflora tripartite*, in New Zealand. However, their relative contribution to pollination was not quantified (Beavon and Kelly 2012). Seed production of Scotch Broom *C. scoparius* was positively correlated with the number of honey bee visits to sites (Paynter et al. 2010).

In California there was a positive correlation between the number of honey bees at a site and seed set of *Centaurea solstitialis* (Barthell et al. 2001).

Honey bees were the primary visitors to *Lespedeza cuneata* in Kansas (Woods et al. 2012) and *Eichhornia crassipes* in China (Liu et al. 2013).

5.3.4.4 Summary

Honey bees often forage on introduced plants (Telleria 1993, Goulson et al. 2002, Morales and Aizen 2006, Miller et al. 2015). Only one study has directly investigated whether honey bees contribute to weed spread, finding that they pollinate Scotch broom (*Cytisus scoparius*). Australian native bees are unlikely to pollinate Scotch Broom because they do not trip the flowers (Simpson et al. 2005).

5.4 Ecosystem integrity and function

5.4.1 Plant-pollinator networks

The study of plant-pollinator networks is an emerging area of research. These studies record which pollinator species forage on which plant species within an ecosystem, and then build a network of relationships between them (Bascompte et al. 2003). A general weakness of plant-pollinator network studies is that whether or not flower visits result in actual pollination is often not considered (Ballantyne et al. 2015). These studies are essentially abundance correlations performed at a larger scale, considering multiple plant and floral visitor species. Indeed, many are based upon previously-collected datasets that recorded plant visitation.

A network that comprises generalist pollinators that interact with many different plant species is known as a ‘nested’ network (Bascompte et al. 2003). As the number of unique interactions between plant species and pollinator species increases, the nestedness of network also increases.

Connectedness is the level of generalization in the interactions at the community level, and refers to the number of observed interactions compared to the number of possible interactions in a network (Landi et al. 2018).

Modularity occurs when some species are more highly connected to each other than to the rest of the network. Networks with a large number of plants tend to be modular (>150), while networks with fewer plant species (<50) are not (Olesen et al. 2007).

Only 15% of species are structurally important to their network (Olesen et al. 2007). Removal of generalist pollinators, that is those with the most connectedness, is expected to lead to the most extreme and rapid extinction of plant species (Memmott et al. 2004). Generalist species, both plant and pollinator provide redundancy in the network (Fortuna and Bascompte 2006). Such species tend to increase nestedness, and more nested networks tend to be the most robust to extinction (Memmott et al. 2004, Burgos et al. 2007)

5.4.1.1 International studies

Santos et al. (2012) examined six plant-pollinator networks in Brazil. Africanized honey bees occupied a central position in all networks, interacting with a large number of plant species. Connectance was low in all networks, while nestedness was low to intermediate. Simulations that removed honey bees from the networks found that there was a strong decrease in nestedness, an increase in modularity and no effect on connectance. It also resulted in the removal of 5 plant species that were connected only to honey bees.

Giannini et al. (2015) found that honey bees had a strong positive effect on nestedness in 21 plant-pollinator networks in Brazil, and was fundamental to the maintenance of the whole network.

Watts et al. (2016) found that honey bees acted as hubs in the networks of eight of nine valleys in Peru. Modularity was lower in networks where more honey bee observations were made, though the nine valleys under study differed in many aspects.

Magrach et al. (2017) compared plant-pollinator networks in nine woodlands near high density orange groves and seven woodlands near low cover-orange groves, during and after orange flowering. After flowering there was a decrease in the number of interactions per pollinator species (i.e. some pollinators changed their behaviour and foraged on fewer plant species). Likely, these changes are driven by honey bee abundance.

Norfolk et al. (2018) compared bee-plant interactions at plots in Egypt where native honey bees are present (low mountains) or rare (high mountains). The low mountain network was more nested, although conclusions are difficult to draw due to the many factors that differed between the regions. For example, less than half of plant species were found in both the low- and high-mountain sites. Honey bees had resource overlap with range-restricted bees and rarely foraged on range-restricted plants. Simulated removal of honey bees from the low mountain network decreased pollinator generalization, increased plant specialization and decreased nestedness.

Hung et al. (2018) created a global dataset using 80 published plant-pollinator networks. 5% of plant species are visited exclusively by honey bees, but nearly half of all species are not visited by honey bees. Honey bees may disrupt interactions between plants and pollinators, including when they are only modestly abundant. The authors conclude that honey bees are the single most important pollinator species across natural systems.

Valido et al. (2019) studied plant-pollinator networks in a high-altitude national park in the Canary Islands. Honey bees are either native to the Canary Islands or were established either hundreds or thousands of years ago. Teide National Park is used by beekeepers every summer, with over 2000 hives introduced during the main flowering season. Valido et al. examined pollination networks prior to and after introduction of the commercial colonies. Clearly such an analysis is flawed, since flowering and insect emergence is correlated with season. Nonetheless the authors report that:

'*The onset of the beekeeping period triggered considerable shifts between the pre- and [post-apis] periods, leading to a reduction in the number of pollinator species but also in interaction links. For example, we did not record 8 (in 2008), then 13 (2009) pollinator species through the [post] apis-period that were already observed in the pre-periods*.'

With no contemporary control sites, this conclusion is unconvincing. However, in one of the years of the study, one corner of the park was not used by beekeepers. This area showed a greater diversity of non-honey bee flower visitors that the sites where honey bees were present. The authors did not give a reason as to why honey bees were not present in the area, but potentially it was because the area was unsuitable for beekeeping in that particular year, rather than an experimental manipulation.

This study is notable in that it is the first to examine the effects of removing commercial honey bees on a pollination network rather than simulating their removal. Despite the obvious methodological flaws, the study suggests that honey bees have measurable impacts on pollination networks and that they contribute to pollination of a wide variety of plants.

5.4.2 Summary

Despite the generalization that nested pollination networks are more stable, it is important to realize that not all plant species can be pollinated by generalist pollinators. Such species and their pollinators may suffer adverse effects in the presence of exotic honey bees if the honey bees displace native species, or rob floral resources without affecting pollination. Honey bees can cause significant changes in the structure of pollination networks.

Honey bees may fill empty pollination niches (Traveset and Richardson 2006, Hung et al. 2018). The effect of their removal from areas where they have been long established is poorly understood.

6. CONCLUSIONS

This report has considered the findings of over 200 papers (Table 1, Table 2) that investigate the effect of honey bees on native animals and both native and non-native plants.

Abundance correlations and resource overlap do not conclusively demonstrate that competition is taking place, (5.2.1), but can be suggestive. There are demonstrated instances where native fauna are able to remove large proportion of the available floral resources before honey bees commence foraging, or even while they forage concurrently – that is the native species can compete and co-exist with honey bees. Conversely, other studies are suggestive of competition and alteration of pollinator networks.

The results of studies that tried to ascertain whether the demography of native fauna populations is affected by the presence of honey bees are mixed (5.2.2). Many of these studies do not have appropriate controls and replication, do not significantly manipulate the number of honey bee colonies at sites and suffer from low power to detect the effects of honey bees. As such, these studies do not provide compelling evidence that honey bees affect the reproductive output of native species.

Honey bees may be aggressive toward native fauna in limited instances (5.2.3) mostly involving inter-specific nest-robbing by Africanized honey bees.

The response of native and non-native plants to pollination by honey bees varies (5.3). Honey bees may be less efficient pollinators of some native plants, but honey bees are equal to the average pollinator. Some native plants have become dependent upon honey bees for pollination (5.3.1). Honey bees usually show plant constancy, and are generally unlikely to increase hybridization of native plants above that of other pollinators (5.3.2). This plant constancy sometimes results in reduced outcrossing from honey bee foraging, though in other instances outcrossing may be increased (5.3.3).

Honey bees pollinate some invasive weeds which are unlikely to be pollinated by native fauna, though native fauna also contribute to the pollination of some weeds (5.3.4).

As a generalist pollinator, honey bees are strong players in plant-pollinator networks and can change the structure of such networks (5.4). Networks with honey bees have higher nestedness, which may contribute to the stability of the network. Simulation studies suggest that the loss of honey bees will result in the extinction of a small number of plants, while a single empirical study suggests that pollinator diversity may be decreased when honey bees are present versus absent.

We conclude that there is an absence of evidence that commercial honey bees have ecological impacts beyond that of feral bees to the extent that commercial beekeeping should be excluded from all national parks in Queensland. If the precautionary principal is to be applied then sites should be assessed for their conservation and beekeeping value and commercial loads of bees excluded from only half of them and monitored for a number of years to ensure that there is not a shortfall in pollination at excluded sites.

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Table 1: Review articles

Reference	Conclusions	Effect
(Keith and Briggs 1987, AHBIC 2005) Beekeeping Industry policy document not peer reviewed	The impact of managed *Apis* on Australian flora and fauna is insignificant	±
	Evidence does not support exclusion of managed *Apis* from conserved forests	±
	Competition has been suggested when resources are limiting, but not that this leads to a reduction in reproductive success	±
	No additional long-term stress is caused above that of feral *Apis*	±
	Sites are used intermittently and for short periods	±
	Apis are only placed under conditions of nectar abundance and research has been done when nectar is limiting	±
(Alaux et al. 2019) Peer reviewed	Limiting beekeepers to agricultural areas would result in colony losses due to lack of forage and pollution and loss of revenue due to the poor market for honey produced from crops	±
	Extensive use of commercial *Apis* may be problematic for conservation and the number of hives should be controlled	-
(Aslan et al. 2016) Peer reviewed	The impact of *Apis* on pollination in natural areas is context dependent	±
	Apis occupies a central role in plant-pollinator networks	±
	In many cases *Apis* is an imperfect replacement for native pollinators	±
(Beard 2015) Government report not peer reviewed	Despite the lack of conclusive evidence of negative impact of *Apis* in New Zealand, they pose a threat to native biodiversity	-
	Access to native forests are required for building hive strength and health	±
	Native floral resources (particularly manuka) underpin the industry	±
	Care should be taken to avoid placing bees at critical life-stage of indigenous fauna	-
	Exclude managed bees from high conservation value areas where there has been no historic use	-
	Secure some *Apis* free areas	-
	Exclude *Apis* from areas where problem weeds occur	-
(Butz Huryn 1995) Peer reviewed	*Apis* may have little direct effects on plants through competition with other visitors	±
	There is overlap in resource use with other pollinators	±
(Butz Huryn 1997) Peer reviewed	*Apis* unlikely to increase hybridization of native flora and probably contributes little to the success of most weeds	±
	Apis alters the foraging behaviour and abundance of some fauna on flowers	-
	Studies are often confounded with habitat changes	±

	Apis are floral parasites of 4 native Australian plants	-
(Butz Huryn and Moller 1995) Peer reviewed	Many weeds are important for honey production worldwide	-
	Apis are important pollinators of some weeds (main pollinator 3%, medium level of influence 9%)	-
	Apis probably contribute little to the reproductive success of few weeds (minimal influence 36%, no influence 27%)	±
	Native birds and bees likely also contribute to pollination of some weeds	±
(Cane and Tepedino 2017) Peer reviewed	Value of *Apis* as pollinators declines when they are deprived good nutrition outside pollination contracts	±
	Smaller more widely spaced apiaries dilute competition	-
	Proposes that pollen and nectar depletion from too many *Apis* increases forage range and time of native bees, which reduces the number and size of offspring, skews sex ratios as males are smaller and increases parasitism. The papers cited as proof don't address the effect of *Apis* on these factors.	-
(Dohzono and Yokoyama 2010) Peer reviewed	Bird-pollinated plants are not impacted by *Apis* as birds can pollinate in the presence of *Apis*	±
	Bee-pollinated plants may be impacted	±
	Some studies have shown temporal or spatial resource partitioning but for most studies the underlying causes are unknown	±
	Effects on native plant reproduction is likely if pollen is limited	-
(Donovan 1980) Peer reviewed	*Apis* mainly forages on introduced plants	±
	Native bees are dominant on native and some introduced plants	±
	Peak native bee activity occurs seasonally when forage is plentiful and thus competition is reduced	±
	Habitat destruction, creation of nest sites and plant introductions have more impact than introduced bees	±
(Douglas 1977) Opinion (no references) not peer reviewed	*Apis* should be excluded from natural areas with 'viable' populations of native insects and plants	-
	Native insects commence foraging later in the day than *Apis* and thus there may be no forage available for them	-
	Feral *Apis* are not a serious problem because their lives are hazardous (water, predation, food)	±
	Apis becomes essential if native pollinators disappear	+

(Eickwort and Ginsberg 1980) Peer reviewed	*Apis* influences foraging patterns of native bees through competition (correlational)	-
	Most studies are inconclusive as competition is difficult to prove	±
(Geldman and González-Varo 2018) Peer reviewed	*Apis* competes with wild pollinators (correlational)	-
	Managed *Apis* should not be placed in protected areas	-
	Management plans should ensure that *Apis* do not spillover to surrounding areas once a crop has finished flowering	-
(Gibbs and Muirhead 1998) Beekeeping Industry report not peer reviewed	Effects on insect pollinators are absent or minor	±
	If nectar is limiting, bird behaviour, pollination and seed set may be affected	-
	No adverse effects on birds and mammals when nectar is not limiting	±
	Bee sites are only used when nectar is abundant	±
	Sites are used infrequently for a short time	±
	No evidence that *Apis* increase plant hybridisation	±
(Goulson 2003) Peer reviewed	Correlational evidence suggests *Apis* competes with native pollinators	-
	Studies have not demonstrated long-term population reduction of native fauna, because it is difficult to do so rather than because there is no effect	±
	Competition is unlikely when there is a nectar flow	±
	Native bees which forage on only one plant family are more likely to experience completion than generalists	-
	Many weeds are visited by *Apis*	-
	Where native pollinators have declined due to habitat loss *Apis* provide pollination services to at least some native plants	+
	Apis should not be placed in sensitive areas	-
(Hatfield et al. 2018) Not peer reviewed?	*Apis* competes with native bees for resources	-
	Few studies have addressed population effects on native bees	±
	Apis are inadequate pollinators for some plants	-
	Apis may increase the population of invasive weeds	-
(Hanley and Goulson 2003) Peer reviewed	Non-native bee visitation increased seed set of introduced plants	±
	There is a positive link between introduced bees and weed spread	-
(Hill 1982) Not peer reviewed? Few references	*Apis* deprives native animals of food	-
	Apis discourage native animals by aggressive and interference competition	-
	Apis fails to pollinate native plants	-
	Apis damages native plants	-

(Hung et al. 2018)	*Apis* does not differ in pollination effectiveness from the average floral visitor	±
	Apis was 75.6% as effective as the best floral visitor	-
	Apis is the most important pollinator	+
	5% of plant species are visited exclusively by *Apis*	+
	Apis may disrupt interactions between plants and pollinators even when they are only modestly abundant	-
(Mallinger et al. 2017) Peer reviewed	52% of studies on competition report negative effects, 25% report no effect and 23% mixed effects, but most are correlational	-
	39% of studies on plant communities reported positive, 34% reported negative, 7% no effect and 20% mixed effects	±
	Managed bees in their native range had lesser competitive effects	±
(Manning 1997) Peer reviewed?	There is no conclusive proof that *Apis* have a significant effect on wildlife	±
	Most studies are flawed	±
	Australian flora produce nectar in superabundance	±
	Long-term studies are needed	±
	Site use is sporadic and for short periods	±
(Matthews 1984) Magazine article not peer reviewed	Nectar is always limited	-
	Apis competes with native pollinators for resources (correlational)	-
	Apis may be inefficient pollinators of native plants or cause hybridization	-
	All evidence points to severe detrimental effects at all times	-
(Moller and Butz Huryn 1996) Department of Conservation report not peer reviewed Viewed abstract only	No quantified evidence that New Zealand plants are affected	±
	There are few potential effects of fauna	±
	There is no evidence that loss of *Apis* would limit weed populations	±
	There is no reason to further restrict beekeeping in conservation areas	±
(Moncur 2005) Beekeeping industry report, not peer reviewed	Research on competition is inconclusive and poorly conducted	±
	Presence of *Apis* reduces predation of native bees	+
	Beekeepers only work areas with surplus nectar	±
	Site usage is several weeks every 1-4 years	±
	Seasonal variation in resource availability leads to changes in native fauna abundance and distribution. Therefore, short-term studies do not reflect the broader context	±
(Moritz and Härtel 2005) Peer reviewed	There is little evidence that *Apis* has any impact on native bee survival, fecundity or population density	±

	There are no reports of *Apis* causing extinction of other bees	±
(Paini 2004) Peer reviewed	Resource overlap, changes in native bee visitation rates and resource collection are taken as evidence for a likely impact of *Apis* on native bee population densities rather than demonstrating this directly	±
	Research to assess *Apis* competition with native bees has problems with sample size, confounding factors or data interpretation	±
	68% of studies use 1-2 sites	±
	11% of studies were confounded	±
	32% of studies considered direct effects on native bees, too few to make definite conclusions	±
	Vulnerable native bees may already have disappeared	-
(Paton 1993, 1996) Peer reviewed/ Report for Australian Nature Conservation Agency	*Apis* remove a large proportion of nectar from a number of native Australian plants	-
	Apis forage at lower temperatures and hence can start foraging earlier in the day and deplete resources before native bees commence foraging	-
	Apis starts foraging after honeyeaters	±
	Apis alters the foraging behaviour of honeyeaters	±
	It is assumed that increased numbers of *Apis* will decrease the number of native fauna visiting a resource if there is competition. However, they may need to spend longer per foraging trip to collect the same amount of resource. This would lead to native counts increasing. A true impact would not be expressed until the next generation, if competition resulted in fewer offspring	±
	Apis can perform pollination of native plants whose natural pollinators have declined or disappeared	+
	Research should first determine if resources are limiting, for this will affect the ability to detect competition	±
	Most experiments are poorly performed and poorly interpreted	±
(Paton 2000) Peer reviewed	In some systems *Apis* has a negative effect, while in others they can be beneficial	±
(Pyke 1990, 1999) Peer reviewed?	A precautionary principle should be adopted rather than an innocent until proven guilty approach	-
	Resource overlap constitutes competition (correlational) and results in reduced abundance of native fauna	-

	Excess resource production is evidence that *Apis* has disrupted plant-pollinator interactions, plants would evolve to produce only that which is necessary *Apis* likely changes the abundance of many native plants and animals and may affect future evolution, this does not need to be, and cannot be, conclusively demonstrated Research is never perfect and can be criticised by both sides Research cannot be completed on every system Scientific studies have established deleterious effect of *Apis* Reduction of the feral *Apis* population will improve conservation Sites where conservation gains from *Apis* removal should be located and there are likely sites that would not benefit appreciably from a reduction nor be adversely impacted by an increase in the number of hives	- - - ± - - -
(Roubik 1988, 2009) Unpublished/ Peer reviewed	Africanized *Apis* has not negatively impacted either solitary or eusocial native bees Native bees compensate for competition by altering foraging time or the species they forage on	± ±
(Russo 2016) Peer reviewed	Many studies are inconclusive or contradictory There is some empirical support for negative impacts Impact on native pollinators may be confounded with human disturbance	± - ±
(Salvin 2015) Rural Industries Research and Development Corporation Report not peer reviewed	Apiaries are generally placed for a 2-4 month period coinciding with a peak in flowering plants Sites are not profitable every year The impact of managed *Apis* in Australian ecosystems varies with time, place and research methods It is unlikely that a consensus in favour of manged bees will be reached through future research Areas where risks are minimal likely exist and can be identified, as with high risk areas that should be protected A decision framework should help to identify these areas	± ± ± - ± ±
(Schwarz and Hurst 1997) Peer reviewed?	Studies have been inconclusive as they are inadequately designed and floral resources were in abundance Effects are likely greater on native bees with restricted forage range	± ±
(Seeman 1994) Unpublished?	Beekeepers use sites for 6 weeks on average but up to 3 months Competition is unlikely if resources are in excess	± ±

	Australian fauna likely does not rely solely on any particular eucalypt due to their infrequency of flowering	±
	Apis reduces available resources, if competition is persistent there are likely long-term effects	-
	Apis are poor pollinators of some plants	-
	Commercial *Apis* are a source of new feral colonies	-
	Any inimical effects are temporary	±
(Stout and Morales 2009) Peer reviewed	Most studies on resource overlap (correlational) found negative or neutral effects of introduced bees on native bees	-
	Mixed results on spatial correlations of native and invasive bees	±
	There is no indisputable evidence that alien bees have a substantial and consistent effect on native bees	±
	Most indirect evidence shows that negative impacts might be strong	-
	Habitats rich in native flowering species should be protected	-
(Sugden et al. 1996) Peer reviewed	Most or all studies are incomplete or flawed and cannot provide useful conclusions	±
	Apis may impact bees under some circumstances	-
(Vergara 2008) Book	Mixed results of studies on pollination	±
	Some Australian plants may depend on *Apis* for full pollination due to declines in native pollinators from habitat loss and degradation	+
	Effects of *Apis* on native pollinator population density are required to prove competition	±
(Wojcik et al. 2018a, Wojcik et al. 2018b) Peer reviewed	Foraging patterns do not necessarily correlate with fitness outcomes	±
	Without monitoring fitness or reproduction little can be inferred about competition	±
	Majority of studies have been performed in the absence of potential conflict and not in natural landscapes where *Apis* use is practiced or considered	±
	9 of 19 studies showed no competition effect of *Apis* on native or wild bees, or had low power to detect an effect	±
	10 of 19 studies showed exploitative competition	-
	6 of 7 studies with direct measures found negative fitness and reproductive outcomes in native bees in the presence of *Apis.* This number is too limited to provide confidence and clarity for management decisions	-

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Table 2: Research papers that examine the effect of *Apis mellifera* on flora and fauna and the strength of the conclusions based on the methods used

Reference	Country/ecosystem	Methods	Conclusions	Effect	Strength
Australia - Introduced					
(Bailey 1994) * description from Paton 1996	Western Australia	Counted visits of insects to 1 bush of *Leucopogon propinquus* for three days when *Apis* had access and three days when they were removed (from flowers)	Number of visits by two native bees and a Diptera increased on days that *Apis* were removed	-	Weak - correlational
(Bernhardt 1987)	Victoria	Collected insects on 8 species of *Acacia* at 9 sites and identified pollen	21-91% of bees carried pollen of 2 or more plants, *Apis* among them	-	Adequate
(Bernhardt and Walker 1985)	Grampian Mountains, Victoria	Collected insects on *Acacia retinodes* and identified pollen	82% of bees carried pollen of at least 1 sympatric plant, *Apis* among them	-	Adequate
(Bernhardt and Weston 1996)	New South Wales and Victoria	Collected insects on *Persoonia* at 17 sites and identified pollen	28% of bees carried pollen from more than 1 *Persoonia* spp. At 2 sites, *Apis* among them	-	Adequate
(Bond and Brown 1979)	Wyperfeld National Park, Victoria - sand dune adjacent to a flood plain	Measured daily nectar availability and concentration of *Eucalyptus incrassate* using exclusion experiments and recorded floral visits at 1 site in 1 year	*Apis* forage later than honeyeaters, when nectar production has dropped *Apis* cannot access young flowers Honeyeaters harvest most of the nectar	± ± ±	Adequate - there could be variation between sites or years due to variation in resource availability. Does not demonstrate a population demographic effect
(Celebrezze and Paton 2004)	Cox Scrub Conservation Park, South Australia - woodland	Recorded visits, fruit set and seed production in native *Brachyloma ericoides* in three treatments: exclusion of birds and *Apis*, exclusion of birds only and no exclusion at 4 sites	Exclusion of birds reduced fruit set and seed production Outcomes still low under no exclusion, authors suggest that this is because *Apis* deplete pollen	- -	Weak - cause of poor fruit set and seed production not adequately examined. Amount of pollen removed by *Apis* not examined, nor the number of pollen

			Apis flew between plants 2x as much as birds, but when birds changed plants they move further	±	grains required for adequate pollination
(Collins et al. 1984)	Wongamine Nature Reserve, Western Australia - heathland	Nectar availability and foraging activity on *Calothamnus quadrifidus* was measured through the day. 1 site, 1 season	Honeyeaters remove significant resources before *Apis* commence foraging *Apis* contacted anthers or stigmata on 42% of nectar-foraging visits *Apis* moved between bushes more frequently than honeyeaters	± + +	Weak to adequate - presence/absence, there could be variation between sites or years due to variation in resource availability
(England et al. 2001)	Jervis Bay, New South Wales - woodland	Compared outcrossing in flowers where birds and mammals were excluded or not excluded from self-compatible *Grevillea macleayana* at 3 sites	90% of pollen was removed from bird and mammal-excluded inflorescences (unpublished data) Less pollen is transferred to stigmas when birds and mammal are excluded Outcrossing is lower when birds and mammals are excluded Outcrossing was low across all 3 sites, whereas sites had previous varied *Apis* was observed nectar robbing	- - - - -	Adequate - presence/absence, seed set was low and difference, if any, between *Apis* and bird pollinated flowers was not reported on
(Goulson and Derwent 2004)	Queensland	Insect visitation and fruit set of an invasive weed *Lantana camara* were recorded on 400 inflorescences at each of 63 sites along a 2000km transect. The transect was completed twice to try to control for latitude and season	*Apis* performed 63% of visits Seed set was positively correlated with *Apis* abundance at sites	± -	Weak - *Apis* contribution to pollination was not measured directly but by correlation with the number of *Apis* counted on 400 inflorescences at sites

(Goulson and Rotheray 2012)	Tasmania	Estimated population size, insect visitation and seed set of an invasive weed *Lupinus arboreus*, comparing results of this study (2010) with a those of another (Stout et al 2002) in 1999	*Apis* were absent from most sites Seed set was not correlated with visitation rates	± ±	Weak - see Stout et al 2002
(Goulson et al. 2002)	Tasmania wide	Compared floral visitations between sites with and without *Apis* and *Bombus.* 10 garden and park sites, 32 pasture, weeds and waysides sites and 25 native scrub or forest sites over 2 months	Native bees were >3 times more abundant at sites that *Apis* were absent from Flower-visiting insects were more abundant at sites where *Apis* were abundant No difference in species richness between sites with and without *Apis* and *Bombus* *Apis* preferred introduce plants There is niche overlap	- ± ± ± ±	Weak - correlational, short duration
(Gross 2001)	Northern Tablelands, New South Wales - fragmented woodland	Examined effect of *Apis* presence on a native legume *Dillwynia juniperina* by recording visits at 2 sites for 3 years. Fruit set was determined after single visits by *Apis* and native bees (*Leioproctus* spp.). Pollen limitation was tested by hand pollination	Native bee presence is negatively correlated with *Apis* There was no difference in fruit set between native bee (25%) and *Apis* (14.5%) pollination *Apis* may be necessary to augment pollination at some sites at some times Flowers were not pollen limited	- ± + ±	Weak to robust - correlational, fruit set component robust
(Gross 1993, Gross and Mackay 1998)	Mount Spec, Queensland - disturbed area adjacent to	Examined the number of pollen grains deposited and pollination success of the pioneer shrub *Melastoma*	*Apis* deposits less pollen on stigmas and removes more pollen from stigmas than native bees	-	Adequate

	montane tropical rainforest	*affine* (36 plants) after single visits of *Apis* versus 4 native bee species and interactions between *Apis* and native bees. Not all species were used in all experiments.	Fruit set and seed set were lower in plants that were visited last by *Apis* than by native bees	-	
			Apis is more likely to be the last visitor	-	
			Native bees were more abundant most of the time	±	
			Native bees were disturbed from foraging by *Apis*	-	
			Suggest that pollen limitation caused by *Apis* increases native bee visitation	-	
			Apis should not be allowed in montane rainforest	-	
(Heard 1994)	Australia - macadamia orchards (native)	Compared fruit set between caged and non-caged flowers, native stingless bees could enter the cage but not *Apis.* 10 orchards, one site was visited 8 times and the others once.	Most *Apis* forage for nectar while stingless bees foraged for pollen	±	Adequate
			Nectar foraging *Apis* still contact the stylar	+	
			There was no difference in nut set between *Apis* excluded and non-excluded treatments	±	
(Hermansen et al. 2014)	Georges River and Kiama Downs, New South Wales - temperate estuaries	Observed floral visitors to a native mangrove *Avicennia marina* at 2 sites for 3 months for 3 years. The proportion of pollen removed and deposited by *Apis* in a single visit was determined. Foraging fidelity of *Apis* was determined by identification of pollen from 60 bees on *A.*	38 floral visitors were identified	±	Weak to adequate - native pollinators loss could be due to other factors than *Apis*, such as nesting habitat loss. Contribution of *Apis* to fruit or seed set was not determined.
			Apis was the most common visitor	±	
			Only *Apis* performed significant pollination, only one beetle and one ant species carried pollen	+	
			Apis remove 96% of pollen from *A. marina* in a single visit	±	
			Apis carried 89-95% *A. marina* pollen	±	
			Apis carried 88-94% *A. corniculatum* pollen	±	

		marina, 30 on *Sarcocornia quinqueflora* and 40 on *Aegiceras corniculatum*.	*Apis* carried 99% *S. quinqueflora* pollen *Apis* has displaced one or more native pollinators	+ -	
(Hingston et al. 2004b)	Derwent Estuary, Tasmania	Measured number of capsules and seeds produced by 4 trees of *Eucalyptus globulus* after single visits of *Apis,* other insects and the swift parrot *Lathamus discolor*	Only single visits of *Apis, Bombus terrestris* and *L. discolor* resulted in seed production Fewer seeds were produced per flower when pollinated by *Apis* than by *L. discolour*	+ -	Weak to adequate - the parrots were caged, thus estimates of their effectiveness may not be accurate
(Hingston et al. 2004a)	Derwent Estuary, Tasmania	Flowers of *Eucalyptus globulus* within 5m of the ground were placed in cages of varying apertures and floral visitors and seed production were recorded on 12 trees. Nectar consumption was measured the following year at a different site, as trees at the original sites did not flower that year	*Apis* are inefficient pollinators of *Eucalytpus globulus* *Apis* may displace birds from flowers *Apis* consume most of the nectar within 5m of the ground	- - -	Weak - pollen was removed from *Apis* while trying to exit the mesh and thus pollination efficiency is poorly estimated. Birds seldom foraged on experimental branches, making estimates of pollination efficiency of open and caged inflorescences inaccurate. Amount of nectar consumed by *Apis* was not measured, but since few other visitors were recorded it can be assumed that they consumed most of the nectar
(Horskins and Turner 1999)	Wyperfeld National Park, Victoria	Floral visitors and nectar availability were recorded at	*Apis* commenced foraging, mainly for pollen, prior to native bees	±	Weak to adequate - results may differ under resource

		3 times of day on 9 days over 4 weeks on *Eucalyptus costata* at 1 site	Nectar supplies remained at midday Pollen covered *Apis* contacted receptive stigmata The insect community differed from a study conducted in 1979	± + ±	limitation and between sites and years, presence/absence
(House 1989)	Atherton Tableland, Queensland	Visitors to native *Neolitsea dealbata* and *Diospyros pentamera* were captured using sticky traps and their pollen loads identified and measured. Number of sites not recorded	*Apis* foraging may be restricted to a single tree or gender of tree in *N. dealbata.* If they do move between tree genders they are expected to pollinate efficiently as they would likely contact the stigmas *Apis* are unlikely to pollinate *D. pentamera* as they are only partially in contact Only 8% of individuals carried pollen of a single species, *Apis* among them	- - ±	Weak to adequate - pollination effectiveness not measured
(Lomov et al. 2010)	Cumberland Plain, New South Wales - revegetated pasture and remnant endangered eucalypt woodland Introduced	Observed floral visitors to a perennial native shrub *Dillwynia sieberi* in remnant woodland (2 sites) and surrounding land undergoing restoration (2 sites) once per month for 4 months in year 1 and 6 months in year 2	*Apis* are less likely to change plants than 2 native bee pollinators *Apis* were the sole pollinator in winter	- +	Adequate
(Mallick and Driessen 2009)	Waratah and Queenstown, Tasmania	Floral visitors to *Eucryphia lucida* were recorded at apiary sites (7) and control sites (6) which were at least 2km from the nearest apiary for 1 month of two non-consecutive years. Nectar	*Apis* visits were 3 times as high at apiary sites Few feral *Apis* were observed at apiary sites Presence of apiaries did not affect visits by native insects	± ± ±	Weak to adequate - some aspects correlational. Black bees were assumed to be feral and yellow bees assumed to have come from apiaries. Observations were

		production and consumption and seed set, fruit set and fruit weight were measured. Apiaries had 50-120 hives	No difference in pollen deposition or seed set in apiary and control sites Fruit set tended to be higher at apiary sites Nectar was depleted at apiary sites Control sites had 50% nectar remaining No competitive effects detected	± + ± ± ±	restricted to within 2m of ground level. Large variation in native insect abundance between sites may affect results
(Moncur et al. 1993, not peer reviewed, Moncur et al. 1995) *Moncur et al. (1993) not available		Seed set was compared in Eucalypts between years with and without *Apis*	Placing *Apis* in *Eucalyptus globulus* seed production areas increased seed quality and quantity Outcrossing of *E. globulus* was unaffected *E. nitens* seed production was unaffected *E. nitens* outcrossing was increased *E. camaldulensis* increased seed production *E. camaldulensis* outcrossing was increased	+ ± ± + + +	Weak - seed set could vary between years for numerous reasons
(Paini and Roberts 2005)	Northern Beekeepers Nature Reserve, Western Australia - low heath	Compared fecundity of a native solitary bee *Hyleus alcyoneus* in trap nests at 7 control and 7 apiary sites (100 colonies) over 2 years. Determined resource overlap by sampling pollen from native bee nests, honey from *Apis* nests and using pollen traps on *Apis* nests. Sites were at least 1.5km apart	Pollen resource overlap varied between 0.52-0.97 *H. alcyoneus* only collected pollen from one plant species whether *Apis* were present or not There were 23% less *H. alcyoneus* nests at apiary sites, mainly driven by data collected in the first year There was no difference in *H. alcyoneus* egg production between treatment and control sites	± ± - ±	Adequate to robust - correlational resource overlap backed up with population reproductive measurements. Power was low for many analyses. Apiary sites were not populated simultaneously and were considered to be control sites until they were populated. It is unclear how many

			There was no difference in male or female *H. alcyoneus* mass between treatment and control sites	±	treatment sites there were at any given time and how this effects results, particularly in the first year
(Paini et al. 2005)	Northern Beekeepers Nature Reserve, Western Australia - low heath	Assessed trap nest occupation and number and weight of offspring produced by *Megachile* sp. 323 at 11 sites over 3 months. 2 *Apis* colonies were introduced to 5/11 sites after 3 months and remained for 3 months. Resource overlap was assessed by collecting pollen from honey from 1 native bee colony per site and from pollen traps on *Apis* colonies	Niche overlap was 0.74 Sex ratio of the native bee species unchanged There was no difference in reproductive success between treatment and control sites	± ± ±	Adequate - commercial colonies are usually not placed during this period due to low resource availability. *Apis* colonies did not produce honey in the final 8 weeks.
(Paton 1993)	Scott Conservation Park, South Australia for *Callistemon rugulasus* and Flinders Chase National Park, South Australia for *Correa reflexa*	Observed floral visitors to *Callistemon rugulosus* and compared territory size of honeyeaters before and after introducing 10 *Apis* hives. Compared fruit production on inflorescences that birds were and were not excluded from.	Honeyeaters made fewer visits per flower and changed behaviour to visit inflorescences within the canopy more frequently when *Apis* were foraging Honeyeaters increased their territory size with increasing *Apis* abundance When *Apis* were infrequent honeyeaters foraged throughout the day, when *Apis* were abundant honeyeaters foraged more intensively early in the morning	- - -	Weak to adequate - 1 site, territory size was based on only 5 hours of observations on 5 birds at a control site and 5 at a treatment site. Not a realistic apiary size. Contribution of *Apis* to pollination was not quantified

			Only 4.4% of nectar-foraging *Apis* struck the stigma of *C. rugulosus,* and 16.7% of pollen-foraging *Apis* There was a positive correlation between number of *Apis* visits and fruit production in flowers that birds were excluded from but lower than that of uncaged flowers of *C. rugulosus* Fruit set and pods per fruit are higher for open than bird-excluded *Correa reflexa* *Orthrosantus multiflorus* flower for one day and *Apis p*rovide significant pollination	- - - +	
(Paton 1999, not peer reviewed)	Ngarkat Conservation Park, South Australia - heathland	Introduced *Apis* colonies (40-100) to some sites and not others in an areas where there had previously been no commercial use and few feral *Apis.* 5/15 sites were stocked in the 1990, 5/15 sites were stocked in 1992 and 9/14 in 1993. No sites received colonies in 1994. Measured flower-visiting fauna populations, documented floral resources and seed production of *Banksia ornata* in sites with and without *Apis* along transects. Sites (29) were	There was much variation in nectar production and fauna abundance between years and sites Sites on transects close to apiaries (100-300m) had less nectar than sites further from an apiary and control sites In one year nectar was significantly reduced at sites 1km from an apiary versus no apiary Pollen quantity was lower at sites with *Apis* Pollen and nectar availability declined in presence of *Apis*, but was in excess (some left at end of day)	± - - - ±	Weak to adequate - abundance is correlational, *Apis* contribution to seed production not quantified directly. Highlights the variation between years and how important long-term studies are.

		variously affected by fire, drought and loss of access and therefore were not used in all 4 years	No difference in honeyeater abundance between sites with or without *Apis* No difference in small mammal abundance between sites with or without *Apis* No difference in abundance of native bees at sites with and without *Apis* No difference in abundance of native ants at sites with and without *Apis* Increased seed production in areas with *Apis* in most years No detrimental effect of *Apis* on flora and fauna	± ± ± ± ± ±	
(Paton 2008, not peer reviewed)	Mt Lofty region, South Australia	Measured nectar throughout the day and counted floral visitors to 18 plant species at 6 sites, often at multiple months/years	Flora produced excess nectar *Apis* are unlikely to be affecting the ability of native fauna to harvest nectar	± ±	Weak to adequate - resources may not always be in excess. Didn't consider night pollinators
(Pyke and Balzer 1985)	8 parks in New South Wales	Compared native bee abundances over 10 minutes at sites where *Apis* foragers were and were not removed. Counted the number of *Apis* and native bees arriving at a resource at different distances from an apiary. Counted the number of native bees along	More native bees foraged when *Apis* were removed *Apis* density was highest close to the apiary Native bee densities on transects were higher close to the apiary, however they declined from that seen before *Apis* was introduced No relationship between *Apis* and native bee abundance on flowers	- ± - ±	Weak - correlational, contradictory results. *Apis* visits were similar at experimental and control plots, indicating that treatment was not effective. Removing *Apis* during a 10-minute period does not seem an effective

		transects and at census plots before and after introducing 30 *Apis* hives			way of reducing density or to effect nectar availability. Foraging range is a factor of resource availability and weather conditions. Sampling protocols poorly described
(Ramsey 1988)	Perth, Western Australia - open woodland	Visitation rates and foraging behaviour of birds, beetles and *Apis* were recorded in *Banksia menziesii* and contrasted with fruit set at 1 site over 3 days. Pollen removal was assessed on 3 caged (no birds), 3 bagged (no birds or *Apis*) and 6 open inflorescences. Pollen deposition was assessed on 10 inflorescences for each treatment. Fruit set was measured on 15 inflorescences per treatment. Effect of cages on *Apis* foraging was assessed using 3 caged and 3 open inflorescences.	38% of open inflorescences received adequate pollen, while only 9% of cages inflorescences did Fruit set was 87% on open inflorescences and 40% on caged inflorescences Pollen deposition by *Apis* exceeded that required for fruit set - therefore factors other than pollen deposition limit fruit set There was no difference in number or length of *Apis* visits to caged and uncaged inflorescences *Apis* do not affect reproductive success, but they may indirectly if they alter the foraging behaviour of honeyeaters	- - ± ± ±	Weak - the low sample size may have precluded the ability to detect differences in behaviour of *Apis* foraging in cages versus open inflorescences. Beetles remove a significant amount of pollen and likely have a greater effect than *Apis*
(Schwarz et al. 1991) *Source not available, description taken from Paini (2004) and	Australia	Compared brood mass and number and number of adults in colonies of *Exoneura bicolor* and *E. nigrihirta* in the presence (6	No effect of *Apis* on any measure of *Exoneura* reproduction *Exoneura* colonies had higher survival at sites with *Apis*	± +	Adequate

Schwarz & Hurst (1997)		or 100 colonies; 4 sites) and absence (4 sites) of *Apis*			
(Simpson et al. 2005)	Barrington Tops, New South Wales	Observed floral visitor on invasive broom *Cytisus scoparius* at 3 sites. Determined fruit set and seed production after a single *Apis* visit	84% of *Apis* visits result in fruit set compared to 35% of flowers that were left open (not all open flowers get visited) The number of seeds per fruit did not differ between those that had a single *Apis* visit and those that were left open Native bee visits outnumbered *Apis*, but they did not pollinate	- ± ±	Robust
(Spessa 1999, thesis) *only abstract available	Australian Capital Territory	Compared pupal weights, survival and frequency of *Amphylaeus morosus* nests with adult females of a native bee at 4 sites with 6 *Apis* colonies or 4 sites with no colonies over two years	There was overlap in resource use No impact of *Apis* pupal weight, number of brood, brood survival or colony size In one year there were more new nests in the presence of *Apis*	± - +	Adequate
(Stout et al. 2002)	Tasmania	Observed insect visits to an invasive weed *Lupinus arboreus* and determined seed set at 20 sites with a range in density of *Bombus*	No relationship between *Apis* visits and seed set. When sites dominated by *Bombus* were excluded there was a positive correlation Insect visitation is necessary for pollination Native insects are unlikely to perform pollination	- - ±	Weak - correlational, seed set relationships were determined indirectly, by correlating the proportion of flowers setting at each site against the number of bees per flower and the proportion of flowers visited. There were 3 sites without *Bombus*, far removed from the rest of the sites.

(Sugden and Pyke 1991)	Nadgee Nature Reserve, New South Wales - grass-tree plains	Measured the number and reproductive output of native bee *Exoneura asimillima* colonies at 1 experimental and 3 control sites over 2 years. The 3 control sites were 7-8km from the control site. The experimental site had 0-29 hives.	Reproductive output was higher at the experimental site in year 1	+	Weak - no replication, sampling period differed between experimental and control sites by 2 weeks in year 1. Differences in number of adults could be from dispersal. Variability between sites could explain results. Conclusion of negative effects based on a lack of any negative results
			More colonies were established near the experimental site in year 2	+	
			Survival of colonies was similar between control and treatment sites	±	
			There were fewer adults at colonies in the experimental site in year 2	±	
			Concludes there is a negative effect of *Apis*	-	
(Taylor and Wheland 1994)	Bargo, New South Wales - Eucalypt woodland	Observed *Apis* foraging on 500 *Grevillea x gaudichaudii* flowers for 2 hours over 2 days. Examined 10 *Apis* for pollen grains	*Apis* transferred little pollen to stigmas	-	Weak - small sample size and short observation period, deterrence of honeyeaters not described
			No *Apis* carried *Grevillea* pollen on their body (the only way likely to affect pollination)	-	
			Apis returning to the hive carried only one type of pollen in their corbicula, which was not *Grevillea*	±	
			High abundance of *Apis* deterred honeyeaters (unpublished and undescribed data)	-	
(Vaughton 1992)	New England National Park - New South Wales - dry sclerophyll forest	Observed visitors to *Banksia spinulosa* for 9-18 days in each of 3 years over 5 months, with exclusion experiments	*Apis* only visited on warmer days late in the flowering period, past peak flowering	±	Weak - cannot differentiate seasonal effects from effects of increased *Apis* abundance, effect of single visits on pollination not examined
			Pollination was higher in the late period (64-73%) than the earlier period (22-27%)	±	
			38% of bird-excluded flowers were pollinated, less than that of open flowers, during the late season	-	

			Fruit set did not differ between open and bird-excluded flowers during the late period *Apis* represented 90% of insect visitors, with other insects not providing pollination Only pollen-foraging *Apis* (35%) contact the stigma	± ± -	
(Vaughton 1996)	Jervis Bay, New South Wales - heathland	Observed floral visitors to *Grevillea macleayana* with exclusion experiments at 1 site	Bird-excluded flowers had 50% less fruit set than open flowers Bird and *Apis*-excluded flowers had higher fruit set than open flowers suggesting that *Apis* prevented self-pollination through pollen removal Only pollen-foraging (79%) *Apis* contact pollen presenters Approximately 30% of pollen removal is attributed to *Apis*	- - - -	Adequate - effect of single visits on pollination not examined
(Whelan et al. 2009)	Jervis Bay, New South Wales - disturbed shrubby heathland and a woodland	Compared visitor frequencies and movement patterns to *Grevillea macleayana* at 3 sites over 2 years (6 days in spring, 6 in winter) with exclusion experiments	*Apis* were the most frequent visitors Pollen was removed equally from open and bird-excluded flowers, suggesting *Apis* remove most of the pollen Fewer pollen grains were deposited on stigmas of bird-excluded flowers Birds were more frequent visitors at a more highly outcrossed site, moving further between plants and visiting fewer inflorescences per plant *Apis* contacted the stigmatic region on 4.4-20.5% of visits	± - - - -	Weak to adequate - the more outcrossed site was of better quality (woodland, larger population, larger plants), correlational, effect of single visits on pollination not examined

(Williams and Adam 1997)	Manning Valley, New South Wales - lowland subtropical rainforest remnants	Recorded bee visits to mass-flowering rainforest trees at 9 sites	*Apis* foraged on most sampled plant species *Apis* was common at all sites *Apis* disturbed small native Hylaeine bees No interaction, displacement or avoidance of *Trigona* by *Apis*	± ± - ±	Weak to adequate - correlational, presence/absence
(Wills et al. 1990)	Northern Sandplain, Western Australia - shrubland	*Apis* and other pollinator visits were recorded along transects of 90 sites in a beekeeping reserve over 30 months. Flowering species were recorded, along with fire history and regeneration mode after fire (reseeding or sprouting)	*Apis* visited 30% of the 413 plant species identified *Apis* favoured species that were widespread and/or locally abundant 70% of the species used by *Apis* were also used by native bees Resource overlap indicates potential resource competition Post-fire reseeding species are of major importance to *Apis* and native bees Increased fire frequency may be leading to lower abundance of post-fire reseeding species If *Apis* sub-optimally pollinate these species than these effects may be compounded *Apis* can play a role in conservation More research is needed on the efficacy of *Apis* pollination of native plant species	± ± ± - ± ± - +	Weak - correlational
New Zealand - Introduced					
(Beavon and Kelly 2012)	New Zealand - broadleaved vegetation, gorse,	Observed visits to an invasive vine *Passiflora tripartite.* Compared fruit-	Floral visitors were almost exclusively *Apis* and *Bombus.* Native bees were observed twice. Damage	-	Adequate - relative contribution of *Apis* and *Bombus* could be

	scrubby mid-successional trees	set in bagged and unbagged flowers. 2 sites	indicated flowers were robbed by birds and mammals Fruit-set was increased when flowers were left open to receive visitors	-	determined by bagging flowers after their visit
(Bennik 2009) Thesis	New Zealand - manuka	Recorded visits to manuka plants at 18 study sites. Exclusion experiments were conducted at 4 sites to determine capsule and seed set. Pollen limitation was tested through hand pollination. Nectar was limited, but not pollen	Diptera abundance was negatively correlated with *Apis* abundance There was no correlation with any other pollinators *Apis* were the strongest factor influencing guild abundance There were some instances of aggression from *Apis* toward large Diptera Capsule and seed set was unaffected	- ± - - ±	Weak to adequate - correlational, nature of aggression was not described or quantified
(Howell and Jesson 2013)	New Zealand	Floral visits by birds and bees (*Apis* and *Bombus*) to native *Phormium tenax* were observed at 4 sites. Outcrossing rates were analysed using genetics and compared between sites where bees (including *Apis*) were and were not common	Bees (*Apis* and *Bombus*) are inefficient pollinators, rarely contacting the stigma Most pollination deposition by bees is self pollen No difference in outcrossing between sites where bees were common versus uncommon Pollen foraging by bees may reduce pollen available for deposition by birds	- - ± -	Adequate
(Iwasaki et al. 2018)	New Zealand - montane-alpine grassland	Plant and insect surveys were conducted along transects at 3 sites over 2 years	Introduced species (*Apis, Bombus*) prefer different flora than native bees Few *Apis* were present in the study area	± ±	Weak - correlational

(Markwell et al. 1993)	New Zealand-honeydew beech forest	*Apis* and *Vespula* wasp visits to a native tree *Nothofagus solandri* were recorded. 1 site with 4 apiaries, samples were taken within 1km of an apiary, over 5 years	Competition between *Apis* and introduced wasps may be used to reduce the number of wasps There were aggressive interactions between the 2 species	+ ±	Weak - correlational, nectar was in abundance most of the year
(Murphy and Robertson 2000, not peer reviewed)	New Zealand	*Apis* distribution was estimated using sugar-water feeders in 2 summers. Floral visits to heather (invasive), flax (*Phormium tenax*), manuka (*Leptrospermum scoparium*) and *Hebe stricta* were recorded at 18 sites, but the timing and length of these observations is not recorded. Fruit and seed set determined for each site. Standing nectar was record at each site	Fruit and seed set of *P. tenax* were higher at sites with bird visitation Abundance and diversity of insect visitors to manuka and *H. stricta* were negatively correlated with *Apis* activity Standing nectar was highest at sites visited by *Apis* but not birds, therefore *Apis* do not deplete nectar	- - ±	Weak - correlational, abundance and diversity of pollinators varies between sites, day and time of day, contribution to pollination by different species should be measured following observed visits rather than indirectly inferred from average fruit and seed set at sites, nectar quantity may differ between sites regardless of bird visitation
(Pattemore and Anderson 2013)	New Zealand	Compared fruit set, pollen-limitation and floral visits of a native shrub *Alseuosmia macrophylla* at 1 site with and 1 without native birds a in 3 years (2005, 2008, 2009) with exclusion experiments	Fruit set was higher on open than bird-excluded flowers *Apis* may play a role in pollination at sites without native birds *Apis* rob nectar	- + -	Weak to adequate - limited observations in some years. Sites likely differ in numerous ways other than presence/absence of bird species, contribution of *Apis* to pollination not investigated, no comments on the behaviour of *Apis* i.e. whether they make contact with reproductive parts

(Paynter et al. 2010)	New Zealand	Floral visitors to invasive *Cytisus scoparius* were recorded. Excluded pollinators from some flowers. 7 sites over 5 years	The most common pollinator was *Apis* (70% of visits) Seed rain was correlated with number of *Apis* and *Bombus* visits Absence of *Apis* due to colony losses from *Varroa* mites reduces pollination and seed set to levels at which biocontrol via seed destruction can be effective *C. scoparius* invasions may persist in some areas due to commercial beehives that are treated for mites	- - - -	Weak - correlational
(Rader et al. 2009)	New Zealand - pak choi (*Brassica rapa* var. *chinensis*) crop	Measured pollen transfer during single visits in 11 fields	*Apis* transferred more pollen per stigmatic contact than 4 species and did not differ from 3 species *Apis* contact the stigma more than 4 species and did not differ from 3 species	+ +	Adequate - pollination outcomes not investigated
North America - Introduced					
(Barthell et al. 2001)	California, USA	Recorded visits to a weed *Centaurea solstitialis* along transects at 3 sites. Excluded *Apis* from some flowers with mesh	There was a positive correlation between *Apis* visitation and seed-set *Apis*-excluded flowers produced more seeds	- -	Weak - would be strengthened by comparing seed-set when flowers were bagged after pollination by different visitors. Some other pollinators were also excluded by the mesh
(Batra 1999)	Maryland, USA - Research station and suburbia	Recorded floral visits to a native tree *Nyssa sylvatica* near an apiary on approximately 20 trees over 3 weeks	No evidence that *Apis* displace native bees	±	Weak - only two *Apis* were seen on trees

(Bruckman and Campbell 2014)	California, USA - coastal sage scrub	Observed floral visits to a native herb *Phacelia parryi* at one site for 17 days over 3 months. The number of pollen grains deposited after a single pollinator visit was determined at a single site the following year. The year after that seed-set was determined after a single pollinator visit at the same site.	*Apis* deposits less pollen than native pollinators *Apis* pollination resulted in reduced seed set *Apis* made 83% of floral visits, and is thus an important pollen vector	- - ±	Adequate - there could be variation between sites or years due to variation in resource availability
(Cane and Schiffhauer 2003)	New Jersey, USA - cranberries (native)	Determined the number of pollen tredrads deposited in single visits by 3 native bees (*Bombus affinis, Megachile rotundata, Megachile addenda*) and *Apis* and compared fruit set, mass and seed set	*Apis* deposited the least pollen *Apis*-pollinated flowers produced less fruit than *Bombus*-pollinated flowers, but not of *Megachile* spp. *Apis*-pollinated flowers produced less fruit mass than *Bombus*-pollinated flowers, but not of *Megachile* spp. *Apis*-pollinated flowers produced less seeds than *Bombus*-pollinated flowers, but not of *Megachile* spp. *Apis* pollination improved all these measures over unpollinated flowers	- - - - +	Adequate
(Champerlain and Schlising 2008)	California, USA - savanna and grassland	Observed visits to and seed set in a native plant *Triteleia laxa.* 2 sites, 2 months in 2 years	*Apis* was the most abundant visitor (87-91%) and performs most of the pollination There was a correlation between the number of floral visitors (including *Apis*) and seed set	± +	Weak - seed set due to different pollinators was not measured directly, but by correlation to number of visits. There was lots of variation between sites - distance to agriculture,

					vegetation, non-bee pollinator abundance
(Dieringer 1992) *only abstract viewed	USA	Compared number of pollen grains deposited and number of seeds produced per flower from single visits of *Apis* and *Bombus pennsylvanicus* on *Agalinis strictifolia*	Number of pollen grain deposited did not differ Number of seeds per flower did not differ	± ±	Adequate
(Gillespie and Elle 2018)	Canada - oak-savannah fragments	Surveyed flowering of native *Camassia quamash* and *C. leichtlinii* and invasive *Cytisus scoparius* and abundance of *Apis* and four native *Bombus* to these plants along transects at 18 sites for 2 years, 6 visits per site per year	No evidence of direct impacts of *Apis* on native *Bombus* in either diet or visit rates Diet overlap between pollinators is driven by native plant resources rather than non-native plant abundance	± ±	Weak - correlational
(Ginsberg 1983)	New York, USA - old field	Collected pollen-collecting insects from flowers along transects at a single site	*Apis* outcompeted native bees at large clusters of attractive plants (apple and yellow rocket)	-	Weak - correlational, single site
(Hung et al. 2019)	California, USA - coast sage scrub	Floral visitors were collected along transects at 12 sites. 16 surveys were conducted every 4-5 days at a single site with 5 transects. Six surveys were conducted biweekly at 11 sites over 6 months.	*Apis* visits increase more rapidly than non-*Apis* visitors with increasing flower abundance *Apis* could disproportionately impact the most abundantly blooming plant species and their pollinators as they are disproportionally attracted to high floral density	- -	Weak - correlational
(Javorek et al. 2002)	Canada - blueberry crop (native)	Determined the number of pollen grains deposited to *Vaccinium angustifolium*	*Apis* and *Megachile* spp. pollinated fewer flowers than *Bombus, Andrena* and *Halictus* species	-	Adequate

		after single visits at 1 site over 3 years (7-40 individuals per species)	*Apis* and *Megachile* spp. deposited less pollen than *Bombus, Andrena* and *Halictus* species *Apis* pollination was more effective than no pollination	- +	
(Jean 2005)	Indiana, USA - black oak savannas	Observed pollinators on *Cirsium discolor*	*Apis* stole pollen from *Bombus* and two native bee species, *Megachile montivaga* and *Melissodes desponsa*	-	Robust presence /absence
(Keys et al. 1995)	Arizona, USA - upland Sonoran desert	Determined pod production of native *Prosopis velutina* after single visits on 7 trees at 2 sites (8-62 individuals per species)	Pod production after single visits by *Apis* did not differ from that of *Chalicodoma, Perdita, Volucella* or Colletidae species	±	Adequate
(Kremen et al. 2002)	California, USA - watermelon crops	Counted pollinator visits to watermelon on organic and conventional farms near or far from natural habitat (there were no conventional farms near natural habitat) over 2 years (14 farms year 1, 16 farms year 2)	No evidence that native bee abundance and diversity declined with increasing *Apis* abundance	±	Weak - correlational
(Krend and Murphy 2003)	Colorado, USA	Observed floral visitors in patches of native, *Heterotheca villosa* and *Grindelia squarrosa,* versus introduced (*Cirsium arvense* and *Centaurea diffusa*) plants. Each plant species was observed 4-6 times, duration and number of sites unreported	*Apis* may prefer introduced plants (P=0.056) *Apis* may contribute to spread of introduced plants Native bees were 7-8 times more abundant on both native and introduced plants	± - ±	Weak - not significant

(Laroca and Winston 1978)	Kansas, USA - university campus	Observed interactions between *Apis* and a native *Bombus* on thistle at 1 site on 1 day	*Apis* stole pollen from the body and appendages of male *Bombus*	-	Robust presence/absence
(McGregor et al. 1959)	Arizona, USA	Observed floral visits to a native cactus *Carnegiea gigantean* on 2 days	*Apis* may collect an entire pollen load from a single flower Nectar foraging *Apis* are coated in pollen A single black *Apis* was observed to make several visits to the same flower, suggesting that *Apis* may be a poor pollinator unless there are so many of them that they require multiple flowers to obtain a load of food	- + -	Weak - It is assumed that the same black *Apis* was seen multiple times, marking it would have been preferable. Actual pollination by *Apis* was not tested. Observations were only made from 0500-1245 on 2 days
(Minckley et al. 1999, Minckley et al. 2003)	Southwestern USA - desert	Bees were sampled on a native bush *Larrea tridentate.* Pollen availability was measured at each site. Bee biomass was estimated at each site. Conducted over 3 years. Only 6/47 sites were sampled more than once	*Apis* were collected at 66% of sites No relationship between *Apis* density and native bee abundance and diversity Bee populations are rarely pollen limited If competition occurs it is likely short-term and uncommon For many species population size lags a year behind resource availability Most studies are insufficient to determine if *Apis* compete with native bees Long-term population studies are needed	± ± ± ±	Weak - correlational, effect on native bee population measures not investigated

(Morandin and Kremen 2013)	California, USA	Compared native bee abundance and diversity and foraging decisions of native bees and *Apis* on exotic versus native plants in mature and newly-planted hedgerows. 4 sites for each type of hedgerow for 1 season	*Apis* preferred native plants at mature hedgerows *Apis* exhibited no preference at newly-planted hedgerows *Apis* abundance did not differ between treatments	± ± ±	Weak - length of time between assessments differed between treatments and study period varied between treatments
(Ott et al. 2016)	California, USA - open grassland, coastal sage scrub, sandy open areas, garden	Observed visitors to a native sage *Salvia apiana* with hand-pollination experiments 4 sites for 2 months	*S. apiana* suffers from pollinator limitation *Apis* is a poor pollinator due to its small size, however its high frequency likely makes it the main pollinator *Apis* attack other visitors	+ - -	Adequate - pollinator contribution would better be estimated by bagging flowers after single visits and comparing seed-set. No description or quantification of the attacks
(Park et al. 2016)	New York, USA and Canada - apple orchard	Compared fruit set and seed set of apples pollinated in single visits by *Apis, Andrena* and *Bombus*	*Apis* deposited less pollen to stigmas Fruit and seed set did not differ	- ±	
(Pleasants 1981)	Colorado, USA	Floral visitors were counted at 2 sites over 2 years	*Apis* numbers were reduced in the second year, while *Bombus* species that overlapped in resource usage were more abundant that year. This is taken as evidence for resource competition	-	Weak - correlational, perhaps *Bombus* had been foraging elsewhere the previous year
(Rader et al. 2013)	Pennsylvania & New Jersey, USA - watermelon crops	Determined the number of pollen grains deposited on stigmas during single visits to watermelon on 18 farms	*Apis* deposited less pollen than wild bees	-	Adequate - pollination outcomes were not investigated

(Richardson et al. 2016)	Florida, USA - scrub	Observed floral visits to an endangered native mint *Dicerandra immaculata* over 2 weeks in 2012 and 2014 at 3 sites	*Apis* visited 1.5x more flowers within plants than native pollinators	-	Weak - pollination efficiency, outcrossing and seed set not examined
(Rogers et al. 2013)	North Carolina - USA	Artificial flowers were placed in flight cages along with either *Apis, Bombus* or a combination of the two. *Bombus* colonies were used in multiple setups, but *Apis* colonies were used only once. Foraging was observed for 30 minutes in 5 replicates	Most *Bombus* stopped foraging at a plant after encountering *Apis* There were 30 instances of *Apis* encountering *Bombus* Foragers encountering another bee are more likely to move away than those not encountering another bee, regardless of species *Apis* were not aggressive	- ± ± ±	Adequate but highly artificial. *Bombus* colonies may have learned the setup, since colonies were used more than once individuals were not naive
(Schaffer et al. 1979)	Arizona	Plant density, nectar and pollen production in *Agave schottii* were measured and floral visits by *Apis* and native *Bombus* and *Xylocopa* recorded at 6 sites over several days	There was resource partitioning by time of day, plant density and nectar concentration Native bee visits were negatively correlated with *Apis* abundance	± -	Weak - correlational
(Schaffer et al. 1983)	Arizona USA	Visitation rates to and nectar availability of a native plant *Agave schottii* were recorded. Density of *Apis* and ants was manipulated and their effect on *Bombus* and solitary bees determined. Ants were excluded to some or all plants at the site. A single	Removing ants from some flowers increased *Apis* and *Bombus* abundance Removing all ants increased *Apis* and solitary bees Increasing the number of *Apis* hives on the site from 2 to 4 had no effect on the number of *Apis* visits Removing the *Apis* hives increased the number of feral (dark) *Apis*, and	± ± ± ±	Weak - correlational, population densities may have changed due to some other variable, particularly given the increase and then decline after *Apis* were removed

		site censused 7/day over ~ month	the number of solitary bees and *Bombus* increased and then declined		
(Stimec et al. 1997)	Canada	Collected pollen from 8 *Apis* colonies over 16 weeks using pollen traps	75% of identified plant species were introduced taxa *Apis* forage on introduced taxa even if native plants are available	± ±	Adequate
(Stoepler et al. 2012)	Virginia, USA	Recorded the number of pollinaria removed and inserted in single visits to *Asclepias exaltata, A. syriaca* and their hybrid at 1 site	*Apis* did not differ from *Bombus* or *Epargyreus* in the number of removed pollinium Hybridization is largely performed by *Bombus*	± ±	Adequate
(Tepedino 1981)	Utah, USA	Counted the number of pollen grains deposited and resulting fruit set and development time after single visits of *Apis* and native *Peponapis pruinosa* to *Cucurbita pepo*	Fruit set and mean developmental time did not differ between *P. pruinosa* or *Apis* visits *Apis* deposited a similar number of pollen grains as *P. pruinosa*	± ±	Adequate
(Tepedino et al. 2007)	Utah, USA - orchards surrounded by national park	Assessed *Apis* and native bee activity in apricot and cherry orchards at various distance from *Apis* hives. Recorded floral visitors to orchards of 4 fruit species over 2 years	Neither native bee nor *Apis* activity in orchards was correlated with distance to *Apis* hives *Apis* were more abundant on apricot, apple and cherry while native bees were more abundant on pears	± ±	Weak - correlational
(Thoenes 1993)	Arizona, USA	Attached dead bee traps to 36 *Apis* colonies for 8 months at one site	147 *Bombus* and 5 carpenter bees were killed trying to enter *Apis* colonies	-	Adequate - presence/absence
(Thomson 2004)	California, USA - coastal scrub	2-3 *Apis* colonies introduced to 3 sites for 4 months for each of 3 years and native *Bombus* colonies introduced	*Bombus* colonies near *Apis* colonies had lower foraging rates *Bombus* colonies near *Apis* colonies had decreased proportion of pollen	- -	Weak - some *Bombus* colonies were killed by predation or damaged by wax moth. Some pollen

		along a 1km transect. Observed foraging activity, counted pupal cocoons	foragers, suggesting nectar limitation Number of individuals produced did not vary with distance from *Apis* colonies Reproductive success (gyne number & ratio, gyne size, male sightings) increased with increasing distance from *Apis* colonies	± -	foragers likely also foraged for nectar. Pollen or nectar amount/concentration was not measured, and thus resource intake was not truly quantified. Gyne numbers and ratio is related to male investment, which cannot be differentiated from workers. Number of males produced was estimated from the number leaving nests during forager observations, rather than directly. There is low probability of observing male dispersion flights. Number/ratio of gynes did not vary when damaged nests were removed from analysis. *Bombus* colonies were well established upon placement, many nest fail to establish and thus *Apis* competition on this aspect is unstudied
(Thomson 2006)	California, USA - coastal scrub	As for Thomson 2004	*Apis* and *Bombus* forager numbers were correlated in only 1/7 months Niche overlap varied, but was higher during resource scarcity	± ±	As for Thomson 2004, correlational

(Thomson 2016)	California, USA - coastal scrub	Counted *Apis* and native *Bombus* foragers at 4-10 patches for one month in 11-13 years over a 15 year period	*Apis* abundance negatively correlated with *Bombus* abundance the following year Resource decline was negatively correlated with *Bombus* abundance	- ±	Weak - correlational
(Thomson and Goodell 2001)	California & New York, USA - apple and almond orchards	Counted number of pollen grains depositved on stigmas and removed from anthers folling single visits of *Apis* and *Bombus* spp.	*Apis* and *Bombus* remove similar amount of apple pollen *Apis* depositis less pollen to apple stigmas *Apis* and *Bombus* remove and deposit similar amounts of pollen to almonds	± - ±	Adequate - pollination outcomes not tested
(Thorp and Briggs 1980)	California, USA - sunflower farm	Recorded foraging of *Apis* and native bees on sunflowers	*Apis* stole pollen from the scopae of native bees *Diadasia enavata* and *Halictus ligatus*	-	Adequate - presence/absence
(Wilson and Thomson 1991)	New York, USA	Quantified pollen removal and transfer from single visits of floral visitors to *Impatiens capensis*	*Apis* perform little pollen transfer *Apis* remove a large quantity of pollen in one visit	- -	Adequate - amount to pollen required for adequate pollination not examined
(Wist and Davis 2013)	Canada	Determined the number of retracted styles, pollen grains per stigma on retracted styles and pollen grains germinated per stingma on retracted styles of *Echinacea angustifolia* after single visits by insects at 2 sites	Pollen grains per stigma that germinated on retracted styles was highest for *Apis*	+	
(Woods et al. 2012)	Kansas, USA - tallgrass prairie	Observed floral visitors to 3 native and a single invasive plant species (*Lespedeza*) at	*Apis* was the primary visitor to the single invasive species *Apis* was never observed on the 3 native species	± ±	Weak - correlational, pollination effectiveness not demonstrated

		6 sites, at least 2 sites per species over 2 years			
(Young et al. 2007)	Vermont, USA	Determined the amount of pollen removed and deposited and seed production of *Impatiens capensis* when pollinated by single visits of *Apis* and *Bombus impatiens*	There was no difference in the number of pollen grains removed There was no difference in the number of pollen grains deposited There was no difference in seed production	± ± ±	Adequate
North, Central and South America - Introduced Africanized					
(Aizen and Feinsinger 1994)	Argentina - subtropical dry forest	Compared *Apis* and native bee visitation rates within fragmented forest on two native tree species *Prosopis nigra* and *Cercidium australe.* 4 sites, 10-12 days per species in 1 year	Native bee visits are negatively correlated with the number of *Apis* visits, but may be due to *Apis* preferring smaller forest fragments	-	Weak - correlational, confounded by differences in patch size
(Andena et al. 2012)	Brazil - transition zone between cerrado and Atlantic forest	Collected foraging bees at plants along 3 transects at 1 site twice a month for an unknown period	*Apis* have the broadest niche and are the most abundant bee *Apis* have not altered the structure of the bee assemblage as intra-specific competition is stronger than inter-specific competition	± ±	Weak - correlational, unable to compare pre and post-*Apis*
(Anna-Aguayo et al. 2017)	Mexico - desert	Recorded interactions between *Apis* and a native bee *Lithurgus littoralis* on a native cactus *Opuntia huajuapensis* over 5 days at the beginning of the flowering season at 1 site	No agonistic behaviour by *Apis* Female native bees did not visit flowers with *Apis* present	± -	Weak - resource availability may affect behaviour, something that may not be experienced in the short duration of the experiment at a single site

(Badano and Vergara 2011)	Mexico - coffee	Counted visitation rates to 4 coffee plants at each of 12 coffee plantations. 1 day per plantation. Distance from native forest and abundance of floral resources were included as variables	Native pollinator diversity was negatively associated with *Apis* abundance	-	Weak - correlational
(Brizola-Bonacina et al. 2012)	Brazil - suburbia	Bees were collected from a native tree *Tibouchina granulosa* at 3 times of day, sampling sites and frequencies undescribed	A stingless bee *Trigona spinipes* is aggressive toward *Apis*, effectively excluding them There was resource partitioning by time	± ±	Weak - correlational, could be improved by testing the concentration and volume of nectar/pollen available throughout the day
(Cairns et al. 2005)	Mexico - patchwork forest and farming	Floral visitors were recorded along 3 transects in each of 3 areas that differed in ecosystem disturbance	*Apis* was observed attacking stingless bees *Apis* was the dominant species in all but the least disturbed habitat	- ±	Adequate - the frequency of attacking behaviour may vary with resource availability, an effect on fitness has not been demonstrated
(Canto-Aguilar and Parra-Tabla 2000)	Mexico - squash crop (native)	Compared pollen deposition of *Apis* and *Peponapis limitaris* to *Cucurbita moschata* in single visits	Female *P. limitaris* deposits 2 times as much pollen as *Apis* Deposition does not vary between *Apis* and male *P. limitaris*	- ±	Adequate
(Carbonari et al. 2009)	Brazil - secondary forest fragment	Counted the number of perforated (robbed) flowers of a native plant *Pyrostegia venusta* (100 samples) Compared number of perforations in abortive and non-abortive buds	84% of flowers were robbed in the peak blooming period Aborted buds were more likely to have been robbed This may affect regeneration of forest fragments as this species is a pioneer plant	- - -	Weak - It was assumed that all robbing was performed by *Apis* due to other visitors being uncommon. However other studies established that the main nectar robber is likely *Trigona*
(Carneiro and Martins 2012)	Brazil - tropical rainforest	Recorded visits and amount of pollen removed by *Apis*	*Apis* deplete pollen *Apis* is one of the main pollinators	- +	Weak - correlational

		from native *Spondias mombin* over 3 years, excluded early *Apis* visits using gauze	Native bee abundance increased threefold when *Apis* were prevented from early foraging	-	
(Cortopassi-Laurino and Ramalho 1988)	Brazil - university garden	Pollen was collected from new storage pots of *Trigona spinipes* and from pollen traps placed on *Apis* colonies over 13 months	The species differ in preferred pollen sources	±	Weak - correlational
(de Menezes Pedro and de Camargo 1991)	Brazil - cerrado/savanna	Floral visitors and plant species were recorded at 1 site for 12 hours per fortnight for 1 year	*Apis* abundance varied between 0-61.4% of samples Most plants don't permit exploitation by *Apis* Little resource overlap between *Apis* and native bees Interference by *Apis* on food niche is minimal	± ± ± ±	Weak – correlational, 1 site
(Dick 2001, Dick et al. 2003)	Brazil - tropical rainforest	Pollination data were collected on a large Amazonian tree *Dinizia excelsa* in pasture, forest fragments and pristine continuous forest	*Apis* were not observed foraging in continuous forest *Apis* are important pollinators in degraded tropical forest *Apis* enable geneflow between fragmented populations due to their foraging range Trees in pasture and forest fragments produced 3x number of seeds which is attributed to *Apis*	± + + +	Weak to adequate - differences in seed production may be due to differences in soil quality etc
(do Carmo et al. 2004)	Brazil - campos rupestres, gradient of field to chaparral-like vegetation	Pollinator visits and behaviour to a native tree *Clusia arrudae* were recorded over 7 months. The amount of pollen	*Apis* remove most of the pollen from male flowers *E. nigrohirta* has less than 0.1% pollen grains on their body when they forage on a flower previously	- -	Weak to robust - pollen transfer experiments had poor replication. The amount of pollen required for adequate pollination

		removed by *Apis* was assessed with exclusion experiments. Amount of pollen transferred to native orchid bee *Eufriesea nigrohirta,* which is the primary pollinator, was assessed with exclusion experiments in 3 replicates. *Apis* effect on seed set were estimated by correlating the frequency of *Apis* visits to male flowers with seed production on female flowers open on the same day.	foraged on by *Apis* compared to an unvisited flower Honey bee visits reduced seed production There was no aggressive behaviour by *Apis* *Apis* were expelled from flowers by *Trigona spinipes* *Apis* avoided flowers with other pollinators on them *Apis* did not remove the resource (resin) that *E. nigrohirta* is foraging for	- ± ± ± ±	was not assessed. There was poor replication of pollen removal by *Apis* and visits were made by other species, which were considered inconsequential. *Apis* effects on seed production were measured indirectly, though only 2 visits were recorded to female flowers, neither of which included contact with the stigma
(Fagua and Ackerman 2011)	Puerto Rico - subtropical dry forest	Compared number of fruit and seeds produced by native *Melocactus intortus* when pollinated in single visits by *Apis,* ants (*Solenopsis*) and hummingbirds (*Anthracothorax dominicus*) at 1 site	Hummingbirds attacked *Apis* and drove them away (undescribed) Seed set did not differ between pollinators Fruit set and number does not seem to differ between pollinators (statistics not provided)	± ± ±	Adequate
(Faria and Araujo 2015)	Brazil - savannah fragments	Observed floral visits to 3 native *Psychotria carthagenensis* populations in 1 season. Fruit production was determined after single visits by *Apis* or native *Augochloropsis* bees	Fruit set from *Apis*-visited plants was 10-33% and did not differ from those visited by *Augochloropsi* spp. (27-30%) *Apis* is the main pollinator of *P. carthagensis*	± ±	Adequate

(Franco et al. 2009)	Brazil - rocky fields	Collected *Apis* and native *Bombus* on flowers along a 1.5km trail once a month for 6 months	Niche overlap between the bee species varied from 0.09-0.50	±	Weak - correlational
(Freitas and Paxton 1998)	Brazil - cashews (native)	Comparison was made between *Apis* and a native bee *Centris tarsata* pollination of native cashews over 2 years with exclusion experiments to enable comparison of single visits	*C. tarsata* is more efficient at removing pollen *C. tarsata* deposits a greater proportion of pollen onto stigmas Similar seed setting efficiency	± - ±	Adequate
(Fumero-Cabán and Meléndez-Ackerman 2007)	Puerto Rico	Counted the number of pollen grains deposited to native *Pitcairnia angustifolia* in single visits by hummingbirds (*Anthracothorax viridis, Chlorostilbon maugaeus*), *Apis, Coereba flaveola*	There was no aggression by *Apis* *Apis* did not rob flowers, but *C. flaveola* and *C. maugaeus* did *Apis* deposited less pollen than *A. viridis* *Apis* deposited more pollen than *C. flaveola* and *C. maugaeus*	± + - +	Adequate - did not examine fruit or seed production
(Giannini et al. 2015)	Brazil	Built interaction networks between plants*, Apis* and a native stingless bee *Trigona spinipes*. Included surveys which were conducted at least monthly for 1 year	Bee species had similar abundance, were linked to a similar number of plants and were of similar importance in their networks The *Apis* network is more nested - has a small group of highly connected species to which the rest of the network depends on. This helps maintain the whole network	± +?	Adequate
(Jha and Vandermeer 2009a, b)	Mexico highlands - coffee plantations	Recorded visits to feeders in shaded and unshaded coffee plantations	*Apis* and social native bees increased their foraging force on the second day of feeder provision, while non-	-	Weak - correlational and artificial

	and a forest reserve		social native bee and wasp visits to feeders reduced. Suggest this is due to resource competition		
(Martins et al. 2013)	Brazil - grassland	Collected all bees visiting plants and the plant species monthly for 1 year. Collected meteorological data and determined increase in urbanization. Compared results with similar surveys in 1967 and 1990 which are not available (Sakagami et al. 1967, Bortoli and Laroca 1990)	Competition with *Apis* contributes to native bee declines, as does habitat loss, introduced plants and climate change Potential and real impacts of *Apis* on native bees in the Neotropics has not been conclusively shown	- ±	Weak - don't quantify the effect of *Apis* on native bee declines
(Macias-Macias et al. 2009)	Mexico - native tomato (*Solanum lycopersion*) and pepper crops (*Capsicum chinense*)	Compared fruit weight and number of seeds after single visits to *S. lycopersion* and *C. chinense* by *Apis* and two native bees (*Exomalopsis* and *Augochloropsis*)	Fruit set was lower for *Apis*-pollinated tomato flowers Fruit weight was lower for *Apis*-pollinated tomato Fruit set was lower for *Apis*-pollinated pepper flowers Fruit weight was lower for *Apis*-pollinated peppers	- - - -	Adequate
(Morales and Aizen 2006)	Argentina - temperate forest	Compared plant-pollinator webs between the 13 introduced and 15 native plants that were most abundant at 4 sites along a 50km transect	*Apis* was the species most closely associated with introduced plant species	±	Adequate
(Osorio-Beristain et al. 1997)	Mexico - tropical dry forest	Floral visitors to *Kallstroemia grandiflora* were compared before (1 day in 1989) and after (2	Pollinator composition has changed since the introduction of Africanized *Apis*	-	Weak - differences in pollinator composition may be due to the introduction of *Apis* or may represent

		days in 1994) the arrival of Africanized *Apis.* Number of pollen grains deposited on bodies and from bodies to flowers was compared between *Apis* and a native stingless bee *Trigona nigra*	*Apis* transfers 2.5 times less pollen than a native bee *Apis* visits flowers 2.65 more frequently than a native bee Overall *Apis* is as efficient at pollination as native bees	- + +	natural fluctuations, short sampling period
(Percival 1974)	Jamaica - coastal scrub	Observed floral visitors	There are few *Apis* in Jamaica *Apis* visited 21 plant species, of which they did not provide pollination to 3 *Apis* works on some of the same species of plant as native pollinators *Apis* commenced foraging earlier than butterflies and solitary bees at a site where observations began at sunrise	± ± ± ±	Weak - correlational
(Pick and Schlindwein 2011)	Brazil - Caatinga	Observed floral visitors to a native climbing plant *Merremia aegyptia* at 1 site for 3 months in each of 2 years	*Apis* foraged only for nectar, discarding pollen grains, resulting in almost 50% pollen loss *Apis* contacted stigmas in 2/3 flower visits *Apis* flew between different plants	- + +	Adequate - amount of pollination performed by *Apis* not quantified
(Pinkus-Rendon et al. 2005)	Mexico - squash and watermelon crops	Observed floral visitors at 1 $100m^2$ field for each crop. Observed encounters between *Apis, Partamona bilineata, Peponapis limitaris* and *Augochlor nigrocyanea*	Floral resources are not used concurrently by different bee species *Apis* displaced bees more frequently than it was displaced	± -	Weak to adequate - correlational, no replication, presence/absence
(Polatto et al. 2012)	Brazil, secondary forest fragment	Observed visits to native *Sparattosperma leucanthum*	*Apis* may accidentally pollinate *S. leucanthum*	±	Adequate

			Apis use holes made by other (native) robber species to rob nectar, which does not result in pollination	-	
(Polatto and Chaud-Netto 2013)	Brazil - degraded secondary forest fragment	Floral visits to 19 native plant species were recorded over 1 year	*Apis* were the most common visitor *Apis* foraged on the most attractive plants	± ±	Weak - plant attractiveness was defined by the number of floral visitors. Given that *Apis* were the dominant species their preferences contribute more to 'attractiveness'. These plant species may not be attractive to other foraging species, which may partially explain their lack of abundance
(Romero and Quezada-Euán 2013)	Mexico	Determined fruit set and fruit and seed weight of native *Jatropha curcas* after pollination by *Apis* and native bee *Frieseomelitta nigra* at 1 site	Fruit set, fruit weight and seed weight did not differ between treatments	±	Adequate
(Roubik 1978)	French Guiana - forest-savannah transition zone	Introduced and removed *Apis* hives and recorded floral visits on 4 plant species over 8 months, with 4-27 days for each plant species	Stingless bees were less abundant in the presence of *Apis* on *Melochia* Native bee density was unchanged on *Rhynchospora, Borreria* and *Mimosa* No aggression was observed	- ± ±	Weak - correlational, *Apis* density was low and thus competition may have been too low to detect, particularly if confounded with climatic variation
(Roubik 1980)	French Guiana - forest-savannah transition zone	Recorded visits to honey-water feeders of varying concentration and volume. *Apis* colonies (1-7) were	Two normally aggressive *Trigona* species abandoned feeders visited by *Apis*	-	Adequate - artificial feeders were unusually rewarding

		placed near a colony of a stingless bee species. Data were collected until one species displaced others from all 4 feeders or the species partitioned the feeders. 1 year at 9 sites over a 20km transect	*Apis* abandoned feeders visited by two other *Trigona* species *Apis* dominated more feeders *Apis* exhibited low levels of aggression toward one *Melipona* bee and one polybiine wasp	± - -	
(Roubik 1981)	Costa Rica - Riparian forest	Observed native *Trigona corvina* and *Apis* on native weedy plant *Baltimora recta*, before and after removing approximately 1km^2 of the plant	*T. corvina* attacked *Apis* *Apis* abundance on remaining *B. recta* increased by 44% after resource removal compared to 17% for *T. corvina*	± ±	Weak to adequate - correlational, presence/absence
(Roubik 1983)	French Guiana - lowland forest	Introduced (1 month) and removed *Apis* colonies (5-15) and measured brood production and food storage by 2 native bees *Melipona favosa* and *M. fulva* at 2 sites. Site 1 had 2 colonies of each *Melipona* and site 2 had 2 *M. fulva* and one *M. favosa*	*Apis* had no effect on the number of brood produced by native bee colonies *Apis* had no effect on food stored by native bee colonies	± ±	Weak - poor replication, no control site
(Roubik et al. 1986)	Panama - lowland forest	Monitored pollen, resin and nectar quality and quantity in returning foragers of 12 native bee species, comparing days when 22 Africanized *Apis* colonies were or were not allowed to forage	*Apis* foraged on the same pollen and nectar resources as native bees Native bees switch resources in the presence of *Apis* There were fewer native bee foragers when *Apis* were foraging *Apis* gathers more nectar and pollen that all stingless bees combined	± - - -	Weak - correlational, no species have gone extinct

			Calculations based on colony populations, food stores and flight range predict that some stingless bee species may go extinct	-	
(Roubik and Villaneueva-Gutiérrez 2009)	Mexico	Trap nests were provided for solitary bees over a 80km transect. Pollen was collected from trap nests and identified. Ran over an 18 year period, spanning pre-*Apis* invasion (1988-1991, although *Apis* first arrived in 1989) to established-*Apis* (2001-2005)	Solitary bees shifted to different resources after invasion by *Apis* The solitary bee community changed after the arrival of *Apis*, but did not decline *Apis* possibly led to increased floral resources through pollination	- ± +	Weak - the period prior to *Apis* arrival is short, and thus it is difficult to conclude that foraging resources changed, rather than there being extreme resource variation in the year preceding arrival. There was low abundance of solitary bees in the year preceding and 1991 (hurricane, low rainfall). This points to perturbation in the 'before' period. The study was performed in a reserve, but landscape changes in the surrounding area are unknown
(Roubik and Wolda 2001)	Panama	Measured bee demography using two light traps on a single tree. Traps were emptied weekly over 17 years (1977-1993) *Apis* abundance was further measured as the number of colonies destroyed. *Apis* arrived in 1984	Native bee abundance did not correlate with *Apis* abundance Native bee abundance did not decrease Some native bees increased in abundance	± ± +	Weak - correlational, no site replication, interactions with rainfall may limit the ability to detect an effect of *Apis* on native bee abundance

(Sanguinetti and Singer 2014)	Argentina - transition between sub-antarctic and patagonic provinces	Observed floral visits to 2 native orchids *Chloraea virescens* and *Brachystele unilateralis* over 3 months in each of 2 years	*Apis* is likely a less efficient pollinator than *Bombus*, but does pollinate both species	±	Adequate
			The majority of pollination is performed by 2 introduced *Bombus*	±	
(Santos et al. 2012)	Brazil - Caatinga	Performed network analysis of plant-pollinator interactions. Fortnightly - monthly sampling for one year. Included data from prior studies	*Apis* should make plant-pollinator networks more robust to extinctions as lost interactions are backed-up	+	Adequate
			Apis cause strong changes in the structure of native bee-plant networks	±?	
			Apis occupied the highest functional role in all networks	-?	
			Pollination networks may look healthier but be dominated and degraded by *Apis*	-	
			Simulated removal of *Apis* decreases nestedness and increases modularity	±	
(Telleria 1993)	Argentina - pampa	Pollen traps were placed outside *Apis* hives over 8 months	*Apis* prefer non-native plants	±	Adequate
(Watts et al. 2012)	Peru - forest	Pollination effectiveness of hummingbirds, native bees and moths and introduced *Apis* on a native shrub *Duranta mandonii* were compared at 9 sites over 2 months with exclusion experiments	Main natural pollinators are large native bees	±	Adequate
			Apis are as effective at pollination	±	
(Watts et al. 2016)	Peru - highlands	Assessed the structure of plant-pollinator networks in nine valleys along 90	*Apis* dominated the bee fauna and were hubs of 60% of networks	±	Adequate

		transects at different altitudes and life zones	Networks differed between sites where *Apis* are and are not dominant	±	
(Wilms et al. 1996)	Brazil primary rainforest with patches of secondary growth	Collected bee visitors to flowers over 4 years at one site 5km in diameter	*Apis* had the largest niche breadth Mass-flowering food sources should enable stingless bees to avoid competition with *Apis*	± ±	Weak - correlational
(Wilms and Wiechers 1997)	Brazil - tropical rainforest	2 colonies each of 2 native stingless bees and Africanized *Apis* were place at one site. Pollen was collected from *Apis* using pollen traps. Honey was sampled by removing 2 combs per hive per month. Pollen was collected from native bees by closing the entrance to the colony for 5 minutes every 30 minutes and removing returning foragers' pollen load. Pollen and honey were also collected from new storage pots.	Niche overlap between *Melipona* and Africanized *Apis* was stronger for nectar than pollen Stingless bees forage on fewer species than *Apis* All species were observed visiting the same flowering patches, therefore there was not physical exclusion The peak in newly filled pollen pots in stingless bees coincided with a period of low potential competition with *Apis.* Thus, there is indirect evidence of competition for food	± ± ± ±	Weak - no replication. The longer period spent collecting pollen from *Apis* increases the likelihood of collecting more species and provides more accurate estimates of resource use. Using pollen in honey to identify the nectar source may not be accurate. Peaks in stingless bee collection may be due to natural seasonal population peaks in both stingless bees and floral resources, rather than from release from competition with *Apis*
(Wolda and Roubik 1986)	Panama - tropical lowland forest	Two light traps were place on one large tree at different heights (3 and 27m). Funnel traps were placed on a ridge. Only bee species were examined. Study conducted over 7	*Apis* has produced no noticeable changes in abundance of other bees	±	Weak - correlational, as only 1 *Apis* was collected (in 1983) it was likely too soon to conclude that *Apis* was not affecting native bee abundance

		years (1977-1983). Africanized *Apis* arrived in 1982			
(Yamamoto et al. 2012)	Brazil - Passionfruit crop	Recorded visitation at 4 sites over 4 years	*Apis* remove all pollen from passionfruit (native) without pollinating them as they rarely touch the stigma	-	Robust
Asia Pacific - Introduced					
(Abe et al. 2008)	Japan	Tested if resource competition with *Apis*, habitat loss, agricultural chemicals or predation affects native bee numbers using a collection of 9 islands which differ in presence/absence of these factors.	Competition with *Apis* cannot explain loss of native bees	±	Adequate - natural experiment
(Aizen et al. 2008)	Southern Andes/ Oceanic islands	Analysed 10 paired plant-pollinator webs that had varying numbers of alien species. Findings are based on many species, but *Apis* are included	Systems with large number of alien species have decreased connectivity among native plants and pollinators, making them highly dependent of generalist alien species	-	Robust - meta-analysis
(Hanna et al. 2013)	Hawaii	Compared insect visitation rates and fruit production in a native tree *Metrosideros polymorpha* between control plots (4) and plots where an invasive robber wasp had been removed (4) over 2 years	*Apis* and native bee *Hylaeus* visitation and fruit production increased at experimental plots *Apis* appear to be acting as a substitute pollinator by replacing extinct or threatened bird species	± +	Weak - fruit production was not measured for pollination by *Apis* and other insects separately

(Johnson and Ashman 2019)	Hawaii - dry tropical forest	Constructed pollen transfer networks by collecting flowers and stigmas from 23 native and introduced plants 2 transects at 13 sites over 2 months. Pollen loads carried by *Apis* were characterised from 10 individuals at 9 sites	Interspecific pollen transfer is largely driven by *Apis* visitation patterns	-	Adequate - small sample size
(Junker et al. 2010)	Hawaii - national park	Compared pollinator effectiveness on a native tree *Metrosideros polymorpha* whose traditional pollinator (birds) became rare	Pollen but not nectar-collecting *Apis* made stigma contact and deposited more pollen per stigma than native bees (*Hylaeus* spp.), which are ineffective	+	Adequate
(Kato 1992)	Japan, Ogasawara Islands	Current range of 9 endemic species was compared to historic surveys	Native bees have declined because of habitat loss and competition with *Apis*	-	Weak - habitat loss not controlled for, see Abe et al. 2008
(Kato and Kawakita 2004)	New Caledonia	Observed floral visitors to 99 plant species at 31 sites over 4 months	*Apis* is the most common floral visitor. On this basis they suggest that native bees are endangered and plant-pollinator interactions altered	-	Weak - habitat loss not controlled for
(Kato et al. 1999)	Japan, Ogasawara Islands	Surveyed plant and pollinator abundance on several islands	*Apis* are dependent on introduced flora to survive year round Native bees are rare on islands where *Apis* is in large numbers, although there may also be an effect of habitat destruction	- -	Weak - habitat loss not controlled for, see Abe et al. 2008
(Liu et al. 20103)	China	Observed floral visitors to an invasive aquatic weed *Eichhornia crassipes* at 3 sites for 6 days over 2 months	*Apis mellifera* is the major pollinator of *E. crassipes*, on the basis of the number of visits and rarity of other pollinators contacting stigmas	-	Adequate

(Miller et al. 2015)	Hawaii - pasture to forest restoration	Collected native and non-native bees on flowers and with pan traps, weighed and identified pollen at 36 sites in mature forest, 16 in grassland, 20 in planted patches, 59 in planted corridors and 30 in old remnant corridors. Conducted over 4 years	*Apis* were more likely to carry mixed pollen *Apis* was more likely to forage on invasive plants *Apis* pollinates some native plants	- ± +	Weak - pollen was removed from bodies rather than the corbitula. Mixed samples in *Apis* could therefore have been picked up in the hive
(Mu et al. 2014)	China - alpine meadow	Nectar volume and concentration and morphological measurements of a native lotus *Saussurea nigrescens* were compared along transects varying in the distance (1-6km) to 3 apiaries. Seedlings were transferred to a common garden and the same measurements were taken	Nectar volume is >60% lower at sites close to apiaries. Common garden experiments demonstrate a genetic response Plants on sites close to apiaries are taller and produce more flowers and seeds than those on sites far from apiaries Overall nectar production per plant is likely the same	±? ±? ±	Adequate - it cannot be ruled out that there is other natural variation along the transects that result in different plant morphologies. Sites close to apiaries have a higher density of seed predators, which may drive this response
(Pan et al. 2013) *abstract only	China - desert	Quantified the number of pollen grains deposited in single visits to native *Hedysarum scoparium* by 3 floral visitors	The number of pollen grains deposited did not differ between *Apis, Megachile* and *Amegilla* species	±	Adequate
(Xia et al. 2007, Sun et al. 2013)	China - subalpine	Compared pollination during peak flowering of a native annual herb *Pedicularis densispica* in the absence (2002-2003, 2005, 2010-	Seasonal introduction of *Apis* resulted in virtual absence of native and non-native *Bombus* in 2004	-	Adequate - a large effect, but no replication. Fruit-set may have been higher that year for other reasons

		2011) and presence (2004) of *Apis* (*mellifera* and *cerana*) at 1 site. Compared pollen removal and deposition by different bee species after one visit	*Apis* introduction results in 2x visitations and 70% increase in seed-set Fruit-set was highest in 2004	+ +	
(Zhang et al. 2015)	China - peach crop (native)	Measured pollen deposition, pollen-tube growth rates, ovary development and fruit set of *Prunus persica* after single visits by *Apis* and *Bombus patagiatus*	*Apis* depositis less pollen Fruit development time is longer for *Apis* pollination Ovary growth is longer for *Apis* pollination Less fruit is produced for *Apis* pollination	- - - -	Adequate
Europe - Native					
(Albano et al. 2009)	Portugal - strawberries (native)	Counted the number of fertilized ovules after single visits by *Apis*, another native bee (Halicitdae) and Syrphidae (*Eristalis*) over 3 months at 1 site. Sample size 14-30 per treatment.	There was no difference in pollination success performed by the various pollinators	±	Adequate
(Alomar et al. 2018)	Spain - almonds	Compared visitation rates in 18 almond fields that varied in surrounding 'natural' areas. Sites were >850m apart. 2 flowering seasons.	Pollinator diversity negatively affected by *Apis* abundance	-	Weak - correlational
(Cayuela et al. 2011)	Spain - within or close to range of two endangered animal species	Quantified fruit set along transects in 3 native fruiting plants, wildcherry, hawthorn and bilberrys. Half of the samples were bagged to prevent insect pollination	Distance from an apiary was negatively correlated with fruit set in 2 of the 3 (wildcherry, hawthorn, bilberry) native plants These plants are important food for endangered species	+ +	Weak - correlational

			Traditional apicultural practices at low densities should be encouraged	+	
(Elbgami et al. 2014)	UK - farmland	5 bumblebee *Bombus terrestris* colonies were placed close (5m) and far (1km) from a 50-hive apiary. Males and new queens were counted, weighed and measured. Number of flights were recorded on video. 1 site, experiment was replicated 2 years later	Colonies close to the apiary gained less weight in both years	-	Weak - poor replication. Authors claim that the distance between sites means that forage availability was unchanged. Depends on average trip distance of *Bombus.* Results differed between years
			Colonies close to the apiary produced fewer queens in one year	-	
			Colonies close to the apiary produced smaller queens in both years	-	
			The number of males did not differ	±	
			Male to queen ratio varied extremely between years and sites in one year	±	
			The number of workers and foraging activity did not vary early in the season (data one year only)	±	
			Later in the season more foraging trips were made by colonies away from the apiary, likely due to a difference in the size of colonies	±	
(Evertz 1995) *source not available	Germany	Compared reproductive success of a leafcutter bee at a site with and without *Apis*	Reproductive success was higher at sites without *Apis*	-	Poor - no site replication, site specific factors cannot be ruled out
(Forup and Memmott 2005)	England - dry lowland heaths	Sampling at flowers along transects at 19 lowland heaths, each transect sampled once. 4 sites were sampled 8 times over 7 months	*Apis* abundance is negatively correlated to *Bombus* abundance at 19 sites sampled once	-	Weak - correlational, relationship found only with poor sampling
			No abundance correlation at the 4 more intensely sampled sites	±	

(González-Varo and Vilà 2017)	Spain - orange and berry orchards with surrounding fragmented woodlands	Sampled *Apis* in woodland surrounding areas of either high or low orange tree density during and after orange flowering over 3 years, using 14-17 sites per year	*Apis* densities were 3.7 times higher in areas with lots of orange trees than areas with few orange trees *Apis* densities after orange flowering were 8 times higher in woodland surrounding high numbers of orange trees than areas with low numbers of orange trees. *Apis* densities reduced by 2.5 times after orange flowering in woodland surrounding low numbers of orange trees. Therefore *Apis* introduced for crop pollination spillover to natural habitats	± -	Adequate - *Apis* should be removed after crop flowering
(Goras et al. 2016)	Greece	Gradually increased the number of *Apis* colonies and measured visitation frequency and duration of pollinators to a shrub *Cistus creticus* at 12 sites, 6 sites per year	Visitation frequency of other pollinators was unaffected Visitation duration of other pollinators was increased by *Apis*	± ±	Weak - correlational, up to 8 hives were introduced per site. This is higher that the European average of 3.1 colonies/km^2, but is lower than an apiary.
(Goulson and Sparrow 2009)	Scotland- lowland pasture/arable Unknown if they are native	Compared thorax width of foraging workers four native *Bombus* species in locations with (10) and without (10) *Apis.* 10 individuals were collected per site	*Bombus* workers of all species are smaller in areas with *Apis*	-	Weak - correlational, worker size varies through the season and the duration of the study was short (22 days) in an attempt to control for this. Colonies may differ in development and hence worker size, thus if more than 1 worker was collected per colony this

					may skew results. The number of brood, and particularly reproductive individuals would better demonstrate competition
(Henry and Rodet 2018)	France - Mediterranean scrubland	Quantified nectar and pollen foraging by *Apis* and wild bees on rosemary at 60 sites along a transect of variation in apiary size and proximity (10m-4km) over 2 years. There were 28 apiary sites that were not in use the entire period.	Nectar crops reduced with decreasing distance to an apiary for both wild bees and *Apis* Pollen foraging success was lower for *Apis* caught closer to apiaries, but there was no effect on wild bees Wild bee occurrence decrease with closeness to an apiary and increased *Apis* density There is a lag for the effect of occurrence suggesting that wild bee fitness is decreased around apiaries Distance from nearest apiary was a better predictor of competition than colony density To allocate half an area to wild bee conservation apiaries should be spaced at least 3.1km apart Apiaries should not be allowed in pristine areas or areas with endangered plant or bee species Apiary free years may boost resilience in wild bees	- ± - - ± - - -	Weak - correlational, the frequency of nectar crop and pollen load measurements in unclear, but likely varies depending on resource availability, competition effects on wild bee population measures were not investigated
(Herbertsson et al. 2016)	Sweden - agricultural land	Bumblebee abundance was measured at sites with and without *Apis* colonies in heterogeneous (>4%	*Apis* reduced (81%) bumblebee density in homogeneous but not heterogeneous landscapes	-	Weak - correlational. Number of *Apis* hives in treatment sites is unclear but may be low (2)

		grassland) or homogeneous (<2% grassland) agricultural landscapes. 5 *Apis* sites in homogenous landscapes, 4 *Apis* sites in heterogeneous landscapes, 6 control sites in homogeneous landscapes and 4 control sites in heterogeneous landscapes. Control sites were searched for bee hives in the surrounding 1.2km radius	Proportional abundance of *Bombus* species with small foraging ranges was lower at sites with *Apis* in heterogeneous landscapes In homogeneous landscapes one bumblebee species was dominant regardless of *Apis* presence or absence Number of *Apis* hives did not correlate with number of foraging *Apis*	- ± ±	
(Hudewenz and Klein 2013)	Germany - nature reserve	Floral visits were observed along transects at sites with and without (>500m from a hive) *Apis.* Counted the number of wild bee ground-nests at 10 sites that were 110-1229m from *Apis.* Compared the number of stem-nests at 5 sites with *Apis* hives versus 4 sites at least 500m from *Apis* hives	Wild bee visits were reduced in sites containing *Apis* Fewer stem-nesting bee species were found in sites containing *Apis* There was no relationship between the number of ground nests and distance to the nearest *Apis* hive Sites with and without *Apis* did not differ in the number of stem nests	- - ± ±	Weak to robust - correlational data backed up with study of reproductive success, a significant effect may have been found under different resource conditions
(Hudewenz and Klein 2015)	Germany	Red mason bees were kept in 12 flight cages with various densities of *Apis* (0 *Apis* (4 replicates) or 1 colony with 100 (4 replicates) or 300 workers (4 replicates). Flower visitations were recorded and reproductive success	Red mason bees visited fewer flowers when *Apis* were present Niche breadth reduced with increasing *Apis* density Reproduction was reduced when *Apis* were present	- - -	Weak to adequate - correlational data backed up with study of reproductive success, feeding of *Apis* colonies reduced competition and colony size was unrealistic, use of a flight cage is highly artificial

		was measured as the number of nests. *Apis* colonies were fed sugar patties throughout			
(Kühn et al. 2006) *only abstract available in English	Germany	Determined seed set after single visits of *Apis, Bombus, Megachile lapponica* and syrphid flies to *Epilobium angustifolium.* Recorded floral visits and number of *M. lapponica* brood cells produced per day before, during and after introduction of 15 *Apis* hives in 2 trials of 14 days	No change in number or duration of visits by *M. lapponica* with increased density of *Apis* No change in the number of *M. lapponica* brood cells with increased density of *Apis* *Bombus* and syrphid flies were more efficient pollinators than *Apis* and *M. lapponica*	± ± -	Adequate - low replication, no control site
(Lindström et al. 2016)	Sweden - oilseed rape crop	Compared wild insect densities in oilseed rape fields with (12 sites in year 1, 11 in year 2) and without (10 fields in year 1, 11 in year 2) *Apis*. Fields were used once only. 2 *Apis* colonies per hectare were placed on treatment sites, the density recommended for oilseed rape. Flying and flower-visiting insects were recorded on transects at 3 distances (100, 200, 300m) from the hives	*Apis* depresses the densities of wild insects	-	Weak - correlational
(Magrach et al. 20107)	Spain - orange and berry orchards	Recorded insect visitors to flowers in woodland	After crop flowering *Apis* presence doubled in surrounding woodland	-	Weak to adequate - some aspects correlational

	with surrounding fragmented woodlands	surrounding either high (9 sites; 28-57%) or low (8 sites; 0-8%) cover orange groves, during and after crop flowering. Surveyed 4 times per year for 2 years. Determined the reproductive success of the two most common wildflowers *Cistus crispus* and *C. salvifolius* after a fixed number of *Apis* visits during and after crop flowering.	Other pollinators shifted and narrowed their diet *Apis* dominated *C. crispus* after crop flowering, which experiences decreased seed set when pollinated by *Apis.* *C. salvifolius* has improved pollination with *Apis*	- - +	
(Montero-Castaño and Vilà 2017)	Menorca, Spain - shrubland	Removed flowers from an introduced plant *Hedysarum coronarium* and compared floral visitors with nearby plots where flowers were not removed. All pollinator species were native. Conducted in spring of 2 years at 3 sites	93% of visits to the introduced plant were by *Apis* *Apis* visits were lower to plots that had introduced plants removed Changes in foraging behaviour of *Apis* were associated with rewiring of plant-pollinator interaction webs, probably due to pollinators avoiding competition with *Apis*	± ± ±	Weak - correlational, however floral resources were limiting
(Monzón et al. 2004)	Spain - pear crop	Observed visits and counted the number of seeds produced by pears after single visits by *Apis* and *Osmia cornuta*	51.8% of *Apis* pollen/nectar foragers touch the stigma 19% of *Apis* nectar foragers touch the stigma Fruit set of nectar-foraging *Apis* was half that of pollen/nectar foraging *Apis* and *O. cornuta*	+ - -	Adequate

			Fruit set did not differ between flowers foraged by *O. cornuta* and pollen/nectar-foraging *Apis* There was no difference in inter-row flight between *Apis* and *O. cornuta*	± ±	
(Nielsen et al. 2012)	Europe (Leeds & Reading UK, Gottingen Germany, Bologna Italy, Tartu Estonia, Levos Greece)	Used visitation rates to 10 plant species to determine if *Apis* affect the rest of the pollinator community. 6-8 sites per plant species visited 3-6 times over a month. Sampling season varied with site/species	Visitation frequencies of *Apis* had an effect on visitation frequencies of all other pollinator groups The effect was strong for solitary bees and hoverflies but weak for *Bombus* For solitary bees the effect of *Apis* was negative in 5, positive in 3 and neutral in the remaining 2 plant species For hoverflies the effect of *Apis* was negative in 4, positive in 1 and neutral in the remaining 5 plant species	±? ±? -? -?	Weak - patch size and density had an effect on results. Plant and pollinator species diversity differs between populations
(Pechhacker and Zeillinger 1994) *conference proceedings	Austria - alps	Nesting sites for solitary bees were placed along a transect from an apiary (100-1500m). Pollen was collected from brood cells of solitary bees (*Osmia rufa, Anthidium manicaturr, Chelostoma campanularum, Megachile* spp.) and by using pollen traps on *Apis* colonies. Conducted over 3 years	No relationship between distance from the apiary and nest site occupation Pollen plant use overlapped, particularly for common plants with large amounts of pollen No competition between *Apis* and solitary bees	± ± ±	Weak to adequate - correlational resource overlap data supported with nest-site occupation

(Steffan-Dewenter and Tscharntke 2000)	Germany - fragmented grassland	Recorded floral visitors, number of occupied trap nests and number of brood of wild bees at 15 sites over 5 months. The number of *Apis* colonies varied from 3 to 65.	Wild bee species richness and abundance were not associated with density of *Apis* colonies Number of brood cells was unaffected No correlation between the number of nests and distance from apiaries	± ± ±	Weak to robust - correlational results backed up with population demographic data
(Torné-Noguera et al. 2016)	Spain - scrubland	Measured rosemary and thyme flower density, wild bee abundance and resource consumption in 21 plots located at varied distance (262-5122) from apiaries	Plots close to apiaries had lower wild bee biomass due to lower abundance of large wild bees *Apis* was the most common visitor to both plants *Apis* consumed the most floral resources	- - -	Weak - correlational
(Walther-Hellwig et al. 2006)	Germany - agricultural	0, 1 or 10 *Apis* colonies were placed on a single site on some days over a period of 1 month, with colonies being moved to a location 18km away as required. Wild *Apis* could be differentiated from the introduced colonies by colour. Counted forager visits on 4 plants	Placing one *Apis* colony did not change *Apis* forager density, but placing 10 colonies did Short-tongued *Bombus* foraged further away in response to increased *Apis* density Long-tongued *Bombus* changed which species of plant they foraged on	- - -	Weak - correlational
(Willmer et al. 1994)	Scotland - raspberry crop	Compared pollen on bodies and amount of pollen deposited by *Apis* and 5 *Bombus* spp.	*Apis* had less pollen on their bodies than *Bombus* *Apis* carried less non-raspberry pollen *Apis* deposited less pollen than *Bombus*	- + -	Adequate - fruit set not investigated
(Willmer and Finlayson 2014)	Scotland	Determined number of pollen grains deposited to	*Apis* deposits less pollen than 2 *Bombus* spp.	-	Adequate - pollination outcomes not determined

		Geranium sanguineum by *Apis* and 5 *Bombus* species in single visits			
(Kraemer and Schmitt 1997)	Canary Islands - montane semidesert (introduced)	Floral visits were recorded and nectar concentrations measured through the day on a native plant *Echium wildpretii* at 1 site for 3 days	*Apis* is a poor pollinator, contacting reproductive organs in 18% of visits, but some other foragers provide no pollination It is assumed that birds do not visit because of nectar depletion by bees (not just *Apis*), or that other bees excluded birds and they in turn have been excluded by *Apis*	- -	Weak to adequate - nectar depletion due to *Apis* is not differentiated, and the nature of relationships between past and current pollinators is conjecture
(Valido et al. 2002)	Canary Islands - sub-alpine zone (introduced)	Observed bird and insect visits to a native plant *Echium wildpretii* 1 site for 1 month	*Apis* were the most common visitor late in the season, birds are most common early in the season and native bee abundance is static Commercial *Apis* were introduced from 5-11 May and the last bird visitor was recorded on 8 May	± -	Weak - poor replication Previous and unpublished work suggests *Apis* deplete nectar, but due to differences in the timing of studies, bird visitors may have been missed in these other studies. It is suggested that *Apis* excludes other pollinators, but seasonal bird abundance could be caused by other factors. Comparison of sites with/without *Apis* or exclusion experiments required
(Valido et al. 2019)	Canary Islands - high-altitudinal plateau	Introduced up to 1,200 *Apis* colonies over 3 years to a 6 ha park. Compared the pre-	The number of floral visitor species reduced in the presence of *Apis*	-	*Apis* were only excluded from 1 portion of the park in 1 year. There may be

		and-post introduction plant-pollinator network. In the 1st year beekeepers did not install *Apis* in one section of the park (control). Compared fruit set and seed production in 5 plant species with and without *Apis* and relative to distance to an apiary in 1 species.	Plant-pollinator networks differed between pre- post- and during *Apis* presence, being less nested during *Apis* presence 2 plant species highly visted by *Apis* (*Echium wildpretii, Spartocytisus supranubius*) have lower seed production per fruit when *Apis* colonies are present More fruit are produced by these 2 plants during this period *S. supranubius* near apiaries produced fewer, heavier seeds There was no effect on the other 3 plants (*Erysimum scoparium, Scrophularia glabrata, Adenocarpus viscosus*)	- - + - ±	annual or site-specific effects that are missed as a result. Abundance is correlational. It seems that the total number of seeds per plant may be unchanged for *E. wildpretii* and *S. supranubius,* given that there are more fruit but less seeds per fruit
Africa - Native					
(Geerts and Pauw 2011)	South Africa - fynbos	Compared bird and bee visits to a native plant *Promerops cafer* at 3 sites with and without added *Apis.* Sites were >0.7km apart. Fauna abundance was recorded for 4 days before *Apis* placement and for 4 days at least 3 days after they were placed	Increased *Apis* stocking rates do not deplete floral resources One of two bird species had a negative correlation with *Apis* abundance	± -	Weak - correlational, fairly short distance between sites, *Apis* stocking rates were low
(King et al. 2018)	Kenya - agricultural land adjacent to national parks	Comparison of bee communities on farms with and without *Apis*	There was little difference in abundance, number of species and community composition of wild bees	±	Weak - correlational

(Tropek et al. 2018)	Cameroon - montane forest remnants	Recorded *Apis* and native stingless bee visits to 7 plant species for 3 months for two years over 16 15m transects	There was resource partitioning	±	Weak - correlational
(Hansen et al. 2002)	Mauritius - upland heath (introduced)	Quantified pollination of two endemic trees *Sideroxylon cinereum* and *S. puberulum* by *Apis* and two native birds *Zosterops borbonicus mauritianus* and *Zosterops chloronothos* at 1 site and 1 season per plant species. Mesh was used to exclude birds but not insects	*Apis* only touch the stigma in 7.5% of visits Birds remove all nectar from a flower while *Apis* removes a portion of nectar Birds stopped foraging prior to *Apis* depleting nectar	- ± ±	Weak - it is unclear if birds would stop foraging at this time in the absence of *Apis*, resources may vary between sites and years, thus competition may vary. Fruit set and size was only compared between bird and insect-visited and insect-visited flowers, so the contribution of *Apis* is unclear
(Norfolk et al. 2018)	Egypt - arid (introduced)	Recorded plant-pollinator interactions at 42 sites 4 times over 4 months. *Apis* were common at low altitude but rare at high altitude	*Apis* tends to visit widespread plants *Apis* only visits a subset of rare specialist plants of which many had restricted ranges *Apis* increases network generalization Range restricted plants may be more vulnerable to insufficient pollination Simulated removal of *Apis* from the low altitude network decreased nestedness	± ± ± ± ±	Poor - low- and high-mountain networks vary in many factors other than *Apis* abundance, which is not controlled for
(Welsford and Johnson 2012)	South Africa - grassland	Determined fruit set and seed set of native *Wahlenbergia cuspidate* and *W. krebsii* after single visits	There was no difference in fruit or seed set	±	Adequate

		by *Apis* and *Lipotriches* at 2 sites for each species			
Israel - Native					
(Shavit et al. 2009)	Israel	Assessed floral visitation rates of native bees and *Apis* within *Apis* native range	Inconsistent between years and plant and native bee species	±	Weak - correlational
(Watts et al. 2013)	Israel - coastal plain (introduced)	Observed floral visitors to a native plant with exclusion experiments for 2 months for 3 years	Main natural pollinator is male eucerine bees but *Apis* were as effective at pollinating Male native bees carried more pollen in populations free of *Apis*, inferring pollen depletion by *Apis* *Apis* are high removal-low deposition pollinators, while eucerine bees are low removal-low deposition pollinators	+ - -	Weak - correlational, difference in pollen counts on male eucerine bees may not be solely due to the absence of *Apis* - this site has higher genetic diversity and the site is bigger

Works are peer reviewed research unless otherwise noted

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