Non-target effects of insect biocontrol agents and trends in host specificity since 1985

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Abstract

Non-target impacts of parasitoids and predaceous arthropods used for classical biological control of invasive insects include five types of impact: (1) direct attacks on native insects; (2) negative foodweb effects, such as competition for prey, apparent competition, or displacement of native species; (3) positive foodweb effects that benefited non-target species; (4) hybridization of native species with introduced natural enemies; and (5) attacks on introduced weed biocontrol agents. Examples are presented and the commonness of effects discussed. For the most recent three decades (1985–2015), analysis of literature on the host range information for 158 species of parasitoids introduced in this period showed a shift in the third decade (2005–2015) towards a preponderance of agents with an index of genus-level (60%) or species-level (8%) specificity (with only 12% being assigned a family-level or above index of specificity) compared with the first and second decades, when 50 and 40% of introductions had family level or above categorizations of specificity and only 21–27 (1985–1994 and 1995–2004, respectively) with genus or 1–11% (1985–1994 and 1995–2004, respectively) with species-level specificity. In all three decades, 11–12% of introductions could not be classified in this manner due to lack of information. Recommendations for future actions to improve this record are made: (1) distinguish host use from population-level impact; (2) develop country-level online summaries of relevant information; (3) plan biological control projects with conservation partners; and (4) conduct post-release comparisons of actual impact to predicted risk based on quarantine studies.

Keywords: Non-target impacts, Parasitoids, Predators, Apparent competition, Hybridization, Indirect effect, Trends in host specificity

Review Methodology: The article was designed based on personal knowledge of the discipline based on 40 years of work in biological control of insects by senior author (RVD), using literature accumulated across that period, together with input from second author (MH). Additional information was assembled, especially for Tables 1 and 2, using the CAB abstract library service to assess globally available knowledge concerning hosts of parasitoids or predators listed in tables. Details in tables were further checked whenever possible by email correspondence with researchers conducting the introduction (see Acknowledgments). Comments from Dan Simberloff were used to revise text. Figure 1 and statistical analysis between decades were done by second author (MH).

Introduction

Whether, when and how frequently introductions of biological control agents have important population-level effects on non-target species is a question of continuing importance to both biological control scientists and conservation biologists. This issue was first raised by Howarth [1], who outlined evidence for significant non-target impacts from biological control agents. While breaking new ground in raising the issue, this paper was, in our opinion, flawed. First, rather than assessing whether non-target impacts had occurred regularly or to what degree on average, Howarth [1] advocated strongly for the proposition that they had occurred, supporting the article’s
assertion by selectively assembling instances of possible impact. While it alerted society to this unintended risk, Howarth [1] did little to objectively assess the magnitude of the problem posed by natural enemy introductions. Second, the author grouped vertebrate introductions, some made as far back as the 1700s by farmers, with introductions of biological control agents made by government scientists after biological control began to develop as a science (post 1920s). This greatly enhanced the perceived negative impact of biological control as most vertebrate introductions for pest control did cause ecological damage. Third, the paper did not adequately differentiate between simple use (feeding or parasitism to any degree) and evidence-based, population-level impacts on non-target species. Fourth, Howarth [1] greatly overstated the perceived negative impact of biological control as most vertebrate introductions for pest control did cause ecological damage. Third, the paper did not adequately differentiate between simple use (feeding or parasitism to any degree) and evidence-based, population-level impacts on non-target species. Fourth, Howarth [1] greatly overstated the risk of extinctions from introductions, by emphasizing the effects of vertebrate and mollusk introductions, as opposed to arthropods (e.g., herbivores, parasitoids and predators) that are used most commonly for biocontrol of pest plants and arthropods. While this article opened a conversation on the potential environmental effects of classical biological control, it did not provide a definitive answer. Further discussion of this issue ensued in the following decades [2–12].

Here we focus on potential non-target impacts of parasitoids and predacious arthropods introduced as classical biological control agents. The impacts of these agents are less understood than those of herbivorous insects and pathogens released against invasive plants. Releases of plant biocontrol agents are well documented [13], and population-level impacts of herbivorous biocontrol agents on native plants have been rare [12]. Analysis of all 512 species known to have been released for weed biocontrol worldwide found no evidence of impact for 99% of the agents. Of the few known cases of impacts, most (>90%) were only of minor importance, without long-term harm to non-target plant populations. Important population-level effects are known only in the cases of some thistle and cacti-attacking insects, principally Cactoblastis cactorum (Bergroth) on some native cacti [14, 15] and Rhinocyllus conicus Fröelich on some native thistles [16, 17]. In contrast, for parasitoids and predators, whose actions are generally invisible to any but specialists, we have less information on population-level impacts. This has led to speculation that non-target impacts are high, based largely on extrapolation from several cases of likely or presumed high-level impact, especially the coccinellid beetles Harmonia axyridis (Pallas) and Coccinella septempunctata (L.) [18, 19] and the tachinid

Figure 1. Index of host specificity for parasitoids introduced in three decades for biological control of arthropods, showing use of more specific agents in the third decade (2005–2015). A one-factor logistic regression model (Wald’s Chi-square test and pairwise contrasts) was used to determine if significant differences in agent host specificity by decade existed at the Family, Sub-Family/Tribe, Species and unknown level. Fisher’s Exact test was used to test for differences at the Order and two Family level of specificity because of zero counts for some decades. Significant differences were detected across decades for natural enemies specific to the level of Family ($\chi^2 = 6.67$, df = 2, $P = 0.04$) and Genus ($\chi^2 = 14.22$, df = 2, $P = 0.001$) only. Significant differences across decades within a host specificity class are represented with different letters (A, B for Family-level differences and a, b, for Genus-level differences). Bars lacking letters across decades within a host specificity class indicate no significant differences. All tests were conducted at the 0.05 level of significance.
flies *Compsilura concinnata* (Meigen) [20] and *Bessa remota* (Aldrich) [21, 22], as discussed below.

Past summaries of impacts of parasitoids and predators on non-target insects and mites include a mini-review for the island of Guam [23], global literature reviews [24, 25] and a detailed analysis of releases of both weed and insect biocontrol agents in Florida [26]. Lynch and Thomas [24] state that non-target effects are recorded for 1.7% of the ca 5000 recorded cases of parasitoid or predator introductions (species × country releases of about 2000 natural enemy species), as detailed in the database ‘BioCat.’ Of these 87 records (87/5000 = 1.7%), most were recorded as causing only minor effects (that is ‘host use’ but not ‘population-level impact’). Seventeen cases (17/5000 = 0.34%), however, were classified as causes of population reductions or other severe impacts. (However, below, we show that some of these cases were in fact of no ecological concern.) No credible cases of extinction were found; one such case is claimed by Howarth [1], but see Hoddle [22]. For introduced parasitoids and predators successfully established in Florida [26], grouping cases by 20-year intervals (data from Table 4 in Frank and McCoy [26]), there was no detectable trend in either the average severity of impacts (categories 1–6) or the the frequency of instances in high-impact categories suggestive of population-level effects; there were 2–5 such events per 20-year period. No further reviews of non-target effects of insect biocontrol have been published since 2007. Here we discuss known or alleged cases of non-target impacts of parasitoid or predator introductions and review trends in host specificity of agents since 1985 (Tables 1 and 2). We conclude with some caveats and recommendations.

**Types of Impacts**

Several types of parasitoids and predators on non-target arthropods have been discussed: (1) direct attacks on native insects, (2) negative foodweb effects, such as competition for prey, apparent competition or displacement of native species, (3) positive foodweb effects effects that benefited non-target species, (4) hybridization of native species with introduced natural enemies and (5) attacks on introduced weed biocontrol agents. After discussing these categories as concepts, we describe instances of each in the section ‘How Common Have Population-Level Nontarget Effects Been?’.

**Type 1. Direct attacks on native insects**

The concept

Direct attack by a parasitoid introduced for biological control is shown by measuring rates of parasitism in a non-target native species by the introduced natural enemy. Estimating the population-level consequence of various levels of parasitism is not easy [27], but rates below 10% are probably of little importance, while high rates (>50%) may reduce populations. Actual impacts on long-term population densities, however, may vary depending on other factors present in the life system of a particular host and may vary among hosts, years or geographic regions. For introduced predators, rates of predation are more difficult to determine because there is usually little evidence remaining of attacks (in contest, parasitism can be measured more easily because hosts can be collected and parasitoids reared out). Once measured, predation rates, as with parasitism, required further analyses to estimate the likely population-level consequences [28, 29].

Examples

Three examples of direct non-target impact have been widely discussed in the literature, and the scientific documentation is best for these three parasitoids: *C. concinnata* (Diptera: Tachinidae), *Microctonus aethiopoides* Loan (Hymenoptera: Braconidae) and *Trichopoda pilipes* Fabricius (Diptera: Tachinidae). Details of these cases are discussed below individually because impacts vary spatially, temporally, or among non-target species.

(a) *Compsilura concinnata*. This parasitoid, released in North America in 1905, was one species among a large group of parasitoids and predators introduced against the gypsy moth (*Lymantria dispar* [L]) (Lepidoptera: Erebidae), a defoliating forest pest [30]. *C. concinnata* is a highly polyphagous tachinid fly, and at the time of its release was recognized as parasitizing >50 insect species [31, 32], a number now significantly increased to several hundred [20]. The highest rates of parasitism by this fly have been recorded on larvae of native saturniid moths – including silk moths [20], buck moths [33] and the luna moth, *Actias luna* (L) [34]. Rates of parasitism vary within group and by region, and this fly has likely affected some non-target species’ populations in some areas, but not others [10].

For the buck moth *Hemileuca lucina* H. Edwards, Stamp and Bowers [33] found attack rates in Massachusetts (USA) of 26–53%, which likely would reduce populations if sustained for several consecutive years. For *Hemileuca maia* (Drury), also in pitch pine habitats in Massachusetts, Selfridge et al. [35] found low and inconsequential levels of parasitism by *C. concinnata*. In contrast, Boettner et al. [20] found 36% parasitism by *C. concinnata* of this species in the same habitat, also in Massachusetts.

For luna moth (*A. luna*), experimental deployment of larvae at sites in Virginia by Kellogg et al. [34] resulted in high levels of attack on some groups, particularly of older instars. Larvae were deployed on separate leaves, at four per small tree, and left in the field for one instar period only. Of all detected parasitism, 78% was caused by *C. concinnata*, and the level of parasitism suffered by deployed caterpillars varied from 0 to 62%, depending on instar and deployment date. The higher of these rates of attack, if sustained, might be sufficient to depress populations, and more information is needed to determine how attack rates vary over time.
habitat and location, and if densities of experimental cohorts affect outcomes.

For giant silkmoths, some of North America’s largest and most attractive moths, Boettner et al. [20] found high levels of attack on cohorts of larvae of both promethea (Callosamia promethea Drury) and cecropia (Hyalophora cecropia [L.]) moths. For cecropia larvae placed five per tree in the field and left for their lifetimes, none (of 500) survived beyond the fifth instar. When individual instars were deployed for one instar period, C. concinnata parasitized 81% of the larvae in each of the first three instars. When larvae of C. promethea were deployed in groups of different sizes for 6 or 8 days, 70 and 66% of larvae, respectively, were parasitized by C. concinnata. These rates, if representative of nature, suggest a high level of impact on populations of these silk moths. Lower levels of attack (25–30%) on these same species are reported from New York by Parry [10]. Goldstein et al. [36] report that the island of Martha’s Vineyard (Massachusetts) retains an intact macrolepidoptera fauna that includes the imperial moth (Eacles imperialis Drury), a species that has declined or disappeared throughout much of New England, and Goldstein et al. [36] related the persistence of this population to the absence of C. concinnata on Martha’s Vineyard, as evidenced by tachinid catures in traps.

These studies collectively support the view that C. concinnata has had population-level impacts on several species of macrolepidoptera in the northeastern USA. Further documentation of the variation of this impact in time and space would be useful, especially contrasting areas with and without outbreaks of gypsy moth. Population modelling may provide useful insights here.

(b) Microtus aethiopoides. Biotypes of this parasitoid were introduced into several countries for control of invasive pests of forage crops, including the alfalfa weevil, Hypera postica (Gyllenhal) (Coleoptera: Curculionidae), in the USA in 1958 [37, 38] and Sitona discoides Gyllenhal (Coleoptera: Curculionidae) in Australia (in 1977) [39, 40] and New Zealand (in 1982) [41, 42], successfully suppressing the target pests in all locations. Barratt et al. [43, 44] have extensively investigated the effects of this parasitoid on native weevils in New Zealand and, to a lesser degree, Australia. No non-target studies have been carried out with this species in the USA. In general, non-target attacks were found in New Zealand [43, 44] on several native weevils, while no significant effects were found in Australia [45, 46]. In New Zealand, laboratory tests found that a variety of native weevils (nine species) were attacked and yielded offspring (suggesting they were in the ‘physiological host range’), while field collections found 14 species of non-target weevils that were parasitized, showing use under natural conditions [43, 47]. Extensive surveys covering altitudinal gradients in three locations collected 12,000 weevils comprising some 36 species, and, of these, eight weevil species were parasitized by M. aethiopoides [44]. Overall, parasitism of non-target species was very low (~2%), but varied by region, collecting site, and season.

Of nine sites surveyed, for 6 years, a moderately high level (24%) of parasitism was found for only one species of native weevil (Nicaea fraudator Broun), at just one site. Irenimus egens (Broun), another species known to be susceptible to attack, was present at that site, at similar densities, but was attacked at a much lower rate.

Population models were developed for N. fraudator and used to estimate levels of population impact associated with particular levels of parasitism. The model indicated that field parasitism rates of 30% implied various levels of population impact depending on the weevil population’s reproductive rate, being a 30% population reduction when reproduction rates were low, but only an 8% reduction when reproduction rates were high [48]. Therefore, the level of parasitism sometimes seen in New Zealand on some species of weevils would likely have a greater impact on populations at higher altitudes, where rates of weevil reproduction (measured as intrinsic rate of increase) are lower.

In contrast to New Zealand, in southeastern Australia a survey by Barratt failed to find any evidence of significant impact on native weevils. Some 197 non-target weevils, comprising 29 species from 15 collection sites, produced just a single non-target weevil (Prosyleus sp.; Curculionidae: Entiminae: Leptopini) parasitized by M. aethiopoides [45]. A second, later survey in Australia [46] detected no further cases of non-target parasitism.

(c) Trichopoda pilipes. Howarth [1] correctly noted that the native Hawaiian ‘koa bug,’ Coleotichus blackburni White (Hemiptera: Scutelleridae), was a suitable host for the egg parasitoid Trissolcus basalis Wollaston (Hymenoptera: Scelionidae) [49, 50] and for the nymphal/adult parasitoid T. pilipes (F.) (Diptera: Tachinidae), two species that were introduced into Hawaii in 1962 against the invasive pest stink bug Nezara viridula (L.) (Hemiptera: Pentatomidae). From these relationships and circumstantial evidence of decline of koa bug on Oahu following the introduction of these parasitoids, Howarth [1] assigned blame for this decline to the biological control project, particularly to the tachinid T. pilipes. However, a field investigation by Johnson et al. [51] found only partial evidence in support of Howarth’s [1] assertion. Johnson et al. [51] measured parasitism of lifestages of koa bug in several habitats and found that egg parasitism due to T. basalis never exceeded 26% and was only detected at sites below 500 m and only on one host plant; in contrast, egg predation by a spider and several species of ants (accidental introductions) was as high as 87%. Parasitism of adult bugs by the tachinid T. pilipes was near zero at 21 of 24 sites, but did reach high levels (up to 70% of females and 100% of males) at three sites where bug density was high, suggesting that dense populations of koa bug may no longer be ecologically possible because of density-dependent attacks by T. pilipes on koa bug aggregations.

Putting direct attack by parasitoids in context

Whether the type of impact seen in the case of C. concinnata is rare or common is critical to determine if
impacts of insect biocontrol agents are likely to harm populations of non-target insects. While the above cases make it clear that non-target attacks are possible (by species first used as biocontrol agents in 1905, 1958 and 1962, respectively) and that at certain times and locations these attacks may be of sufficient magnitude to locally reduce population densities, they do not clarify if such impacts are likely for agents released since non-target effects of introduced arthropod agents for pest insect control became of concern (ca. 1995) and better regulated. Below, in ‘How Common Have Population-Level Nontarget Effects Been?’ we discuss a longer series of cases to put potential risk from introduced natural enemies to non-target species into perspective.

Mitigation of direct attacks
Since ca. 1995, requirements for determining the likely host ranges of insect biocontrol agents have increased in countries most commonly practicing insect biological control [52]. Our summary of host range information on parasitoids introduced from 1985 to 2015 (Table 1) suggests a reduction in the proportion of agents with family-level specificity and an increase in agents with genus or better level of specificity (Fig. 1). Few insect biological control agents, however, are monophagous, and most are likely to have host ranges that include some other species that are taxonomically related or ecologically similar, which may be attacked, but likely at lesser degrees than the target pest. The key to mitigating direct impacts of introduced parasitoids and predators is to correctly estimate likely host ranges relative to the non-target fauna (i.e., native species or valuable introduced species such as weed biocontrol agents) in the area of release. The goal is not to avoid all host use, but to avoid damaging population-level effects on non-target species.

Type 2. Negative food web effects
The concept
Introduced species can affect native species through food webs [53]. In some cases, they may directly attack native species, but the level of such attack may be significantly increased by the introduced species’ ability to maintain larger populations by attacking, but not suppressing, the target pest or other species, a situation termed ‘apparent competition.’ In other cases, the introduced species never (or rarely) attacks non-target species, but their populations are still reduced through competition with the introduced natural enemy for food or hosts, a situation often termed ‘displacement.’

Apparent competition grades into simple direct attack and may be difficult to recognize except by an enhanced level of impact when the natural enemy is in association with the other host. For example, C. concinnata directly parasitizes larvae of various native moths and butterflies, as discussed above, but C. concinnata numbers, and hence the numbers of such attacks, are likely to rise and fall with the local density of gypsy moth, the target host. Here we have arbitrarily considered this case as one of simple direct attack because the link to gypsy moth densities, while quite likely, is supported by only very limited data [54]. A better example of apparent competition by an introduced biological control agent is that of Cotesia glomerata (L.) (Hymenoptera: Braconidae), Pieris rapae (L) and Pieris oleracea Harris (both Lepidoptera: Pieridae) in southern New England, as discussed below [55–57].

Displacement of one species of parasitoid by another introduced later has been observed during biological control projects [58–60]. This has generally been viewed as a favourable process, as each more efficient parasitoid drives the invasive host insect to a lower level and excludes less efficient biocontrol agents. However, if the displaced species are native parasitoids exploiting an exotic pest, this could be viewed as an undesirable impact on a native species whose ‘commonness’ declines due to the introduced agent. However, such observations typically are made in the context of studies of mortality of the introduced pest insect, often in a crop. Decline of a native parasitoid (or predator) from former abundance on a non-native host on an introduced crop plant is not by itself evidence of significant ecological impact because both the host and its crop habitat are an artificial human construct. The important question is whether or not the superior introduced parasitoid displaces the native parasitoid from its native hosts in natural habitats. Unfortunately, because the focus of most studies is on pests on crops, observations of displaced native parasitoids on non-pest hosts in native habitats are rare and should receive more emphasis. Therefore, further study is needed to determine the status of affected native parasitoids in non-crop habitats.

(a) Apparent competition. This interaction is named apparent competition because superficially after a new herbivore arrives, a related local native herbivore begins to decline, making it appear as if the invasive species is competing with the native one for some resource, while in reality the negative population impacts on the native species are mediated through unequal effects of a shared natural enemy. Evidence for apparent competition has been sought in a variety of systems in which one member of a pair of herbivores is invasive and one native, and the parasitoid attacking them both is a local native species. Apparent competition has been found in some cases [61] but not others [62].

The link to biological control is the subset of apparent competition cases in which the parasitoid (or predator) mediating the interaction is a species introduced for classical biological control of the non-native member of the herbivore pair. Few such cases have been documented, possibly because of a lack of work in this area. Redman and Sibber [54] noted that if they artificially deployed larvae of the butterfly Papilio canadensis (Rothschild and Jordan) (Lepidoptera: Papilionidae), those larvae placed near gypsy moth populations suffered higher rates of parasitism (45%).
(most from *C. cinnucnata*) than larvae deployed in areas without gypsy moths (16%). This difference was statistically significant, although there was no significant effect on the percentage of larvae reaching the adult stage (3.8 versus 4.3%), suggesting the action of some compensatory mechanism later in the life cycle.

A well-documented instance of apparent competition due to a biological control agent is that of the parasitoid *C. glomerata* (*Hymenoptera: Braconidae*), introduced to control the invasive brassica pest *P. rapae*. This parasitoid appears to be cause of the decline of a related native white butterfly, *P. oleracea* (formerly *Pieris napi oleracea*) in Massachusetts, but not in northern Vermont (USA) due to differences in volinism [55–57]. Interestingly, this effect was later reversed by *P. oleracea*’s use of a non-native host plant [57] and the displacement of *C. glomerata* from its position as the dominant parasitoid of *P. rapae* in crops by the introduction of *Cotesia rubecula* (Marshall), another biological control agent that is a specialized parasitoid of *P. rapae* [60].

(b) Displacement via competition for prey. Perhaps the best-studied example of displacement of native species by introduced predators is the case of two introduced ladybird beetles, *H. axyridis* and *C. septempunctata*, in North America and Europe (*H. axyridis* only). In North America, these species replaced native ladybirds as the common species in a wide variety of crops, causing formerly common native ladybirds to become rare at the study locations [18, 63–66]. Among the most widely affected species were *Adalia bipunctata* (L.) and *Coccinella novemnotata* Herbst. More recently, the invasion in Europe of *H. axyridis* has also begun to affect native ladybirds there [67]. To explain why displacement of native ladybird species happened, several mechanisms have been proposed and to some extent tested, including direct predation effects on native ladybirds (‘intraguild predation’ or IGP), apparent competition mediated by pathogens, and displacement due to reduction of available prey in sampled habitats.

Asymmetrical IGP effects (ones that are more severe on the native species) have been demonstrated, showing that larger non-native species often have the advantage over smaller native ones [68, 69]. However, while asymmetrical IGP is well demonstrated [70], whether it has caused population declines of native species is not. Limited attempts to test IGP as the factor responsible for the decline in native ladybirds in crops have not supported the idea [71].

Another possible mechanism, apparent competition mediated by a pathogen, is a novel idea supported by one study [72]. In Europe, the microsporidian *Nosema thompsoni*, found in but harmless to *H. axyridis*, is lethal to *C. septempunctata*, a local native species. When *C. septempunctata* adults or larvae eat eggs or larvae of *H. axyridis*, they die. There is no evidence that pathogens associated with non-native ladybirds affect additional native species of North American or European ladybirds, but this possibility merits investigation.

The third possible mechanism postulated for disappearance of native species following the appearance of non-native ladybirds is that these competing species drive densities of shared prey to levels too low to support the native species. For example, Mizell [73] states that *H. axyridis*’ presence on crape myrtle (*Lagerstroemia indica L.*) in northern Florida in 1999–2001, which had been associated with declines of the native aphid, was later reversed by *P. oleracea*’s use of a non-native host plant [57] and the displacement of *C. glomerata* from its position as the dominant parasitoid of *P. rapae* in crops by *Cotesia rubecula* (Marshall), another biological control agent that is a specialized parasitoid of *P. rapae* [60].

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species and the absence of the highly competitive non-native ladybirds. Hasler and Kieckhefer [80] surveyed putative native ladybird habitats (fields and areas of woody vegetation) in South Dakota but found that the targeted native ladybirds were rare in the habitats surveyed and that both H. axyridis and C. septempunctata were present in many of the putative refuge habitats. In contrast, Bahlai et al. [81], analysing a 24-year data set from Michigan (with larger acreage of semi-natural forest habitats than South Dakota), found that only two species of ladybirds showed statistically significant declines (A. bipunctata and C. maculata) after the establishment of these two exotic ladybird species. They also found that in semi-natural forested habitats ladybird assemblages were unique in both composition and variability from those in crop fields and concluded that such forested areas acted as refuges for native coccinellids.

The sum of evidence suggests that these two non-native coccinellids, H. axyridis and C. septempunctata, have greatly lowered the abundance of several native ladybirds in agricultural fields. While the same native coccinellids also seem rare in natural habitats, earlier estimates of their abundance there are lacking, and we cannot, therefore, know if significant changes have occurred in those habitats. Finally, a question not yet raised by researchers on this topic is whether the density of these native coccinellids in crops, where they previously exploited high-density prey species that were often themselves exploited by parasitoids, is the right benchmark against which to measure impacts, or if a more appropriate standard might not be densities of native coccinellids in non-crop habitats (e.g., native forests or grasslands) where these native ladybirds presumably exploited native prey.

(c) Displacement via competition among parasitoids for hosts. Introduced parasitoids may displace local species (either native or previously introduced species) if they are more efficient at exploiting hosts (see subsection ‘Group 3. Displacement or Other Indirect Impacts’). There is, however, no clear well-documented example in which an introduced parasitoid has had dramatic population-level impacts on a native parasitoid where it is acting on its native hosts in their native habitat. Rather, displacement has only been documented in crops, where an introduced parasitoid displaces native parasitoids that are usually exploiting invasive hosts. However, one case exists where such displacement of native parasitoids from native hosts is likely to have occurred, but has not yet been adequately documented: the release of the American braconid Lysiphlebus testaceipes (Cresson) in Europe. This aphid parasitoid (attacking mostly species in the Tribe Aphidini [personal communication, Starý]) was introduced (1973–1974) from Cuba to France for control of pest aphids in citrus [82]. In addition to providing control of the target pests, L. testaceipes spread into non-crop habitats and became the dominant parasitoid on a number of native aphids inhabiting various types of vegetation, including forests [82]. The list of aphid species parasitized by L. testaceipes increased as it spread, reaching at least 32 by 1986 [83], and continued to increase as the parasitoid’s range expanded into the Iberian Peninsula [84]. In southeastern Europe, a total of ten host species were recorded (among 115 aphid species sampled from 422 plant species), and this parasitoid was not only found principally on species of Aphis (A. craccivora Koch, A. fabae Scopoli, A. nerii Boyer de Fonscolombe, A. ruborum [Börner], A. urticae Gmelin, A. gossypii Glover, Aphis sp.), but also occurred on species in Rhopalosiphum and Toxoptera [85]. It is possible that L. testaceipes, which attacks many native European aphids in various habitats, may suppress some species of native parasitoids exploiting native hosts in native habitats. However, this has not yet been documented, in part because the collection of information on the native aphid parasitoids—their presence, abundance, phenology and host ranges—requires an extremely high level of taxonomic training and ecological knowledge and several years work to understand temporal and spatial effects.

(d) Parasitoid host shifts caused by competitive displacement. Diachasmimorpha tryoni (Cameron) (Hymenoptera: Braconidae), a parasitoid of frugivorous tephritids, attacked lantana gall fly (Eutreptia xanthochaeta Aldrich [Diptera: Tephritidae]) in the laboratory but did not do so in the field in Hawaii after its release until a superior competitor, Fopius arisanus (Sonan) (Hymenoptera: Braconidae), was introduced. After that release, competition apparently caused D. tryoni to shift onto lantana gall fly, which was a more available host in the presence of F. arisanus [86].

Putting risk in context

Polyphagous and oligophagous parasitoids likely pose risk to native parasitoids. Documenting such events, however, is difficult because of the high level of taxonomic skill needed to separate parasitoid species and make sense of the survey results. Projects assessing these types of non-target effects, especially population-level consequences, require work spanning several consecutive years with study sites that are representative of the various habitats within which the agents of interest are operating.

Mitigation

Looking forward, regardless of what past introductions may have done, the solution to minimize unwanted non-target effects is to introduce parasitoids with narrow host ranges, as estimated by adequate pre-release testing in quarantine and, if reliable data are available, host use in the natural enemy’s area of origin.

Type 3. Beneficial food web effects

Beneficial indirect effects on native species can also follow biological control of invasive pest insects. Schreiner and Nafus [87] observed population increases of native moths following biological control of Penicillaria jacosatrix Guenée.
(Lepidoptera: Noctuidae) on mango in Guam by the tachinid Blepharella lateralis Macquart. Pest suppression led to a large increase in flowering by mango that caused several native moths to increase in abundance because this resource had improved.

In Queensland, Australia, biological control of invasive crop-pest scales (Ceroplastes destructor Newstead and Ceroplastes rubens Maskell [both Hemiptera: Coccidae]) provided benefits in forest ecosystems by reducing densities on native forest plants of invasive ants that were attracted to honey dew produced by invasive scales. Uncontrolled scale populations tended by invasive ants reduced vigour of forest plants and decreased use of plants by larvae of native lycaenid butterflies, such as Hypochrysops miskini (Waterhouse) and Pseudodipsas cephenea Hewitson. These native butterflies must be tended by native ants, and invasive ants disrupt this important mutualism (as described by Sands in Van Driesche et al. [88], with further details in Waterhouse and Sands [40]).

In Tahiti, invasion (due to movement of infested plants) of the glassy-winged sharpshooter, Homalodisca vitripennis (Germar) (Hemiptera: Cicadellidae), posed a significant risk for native spiders, for whom this hyperabundant sharpshooter proved to be a poisonous prey [89]. Biological control of the invader by release of the mymarid egg parasitoid Gonatocerus ashmeadi Girault greatly reduced the pest’s densities [90], which subsequently lowered this threat to native spiders.

Type 4. Hybridization with native congeners

The concept
Natural enemies may sometimes be introduced into areas that contain closely related species that may have different host or prey ranges. If these species have been geographically separated, they may lack the premating barriers needed to sustain their separate species identities, and inter-species matings may occur, leading to hybridization and genetic introgression [91]. Hybridization is common in some groups in nature. For example, the eastern and Canadian tiger swallowtails (Papilio glaucus L. and P. canadensis Rothschild & Jordan), whose distributions are generally distinct, have a hybrid zone along their common border [92].

When individuals of distinct species mate, several outcomes are possible: (1) mating may occur but be infrequent due to differences in habitat or host plant affiliations, allowing separation of the species even in partial sympatry. In this case a stable, low rate of hybridization may occur due to overlap, accidents, or chance where the species’ distributions overlap. This outcome is probably of little or no ecological consequence; (2) in other cases, there may be substantial contact between the species due to similarity in habitat, and mating may be relatively frequent. If offspring are infertile, there may be selection on mating behaviours to reduce the rate of hybridization over time; and (3) if overlap is substantial, selection for premating segregation is ineffective due to lack of variation in mating behaviours, and offspring are fertile, species may fully introgress with each other and one or both species may cease to exist in their previous taxonomic status, leading to a reduction in biodiversity.

Examples
Several examples of hybridization are discussed in the literature for insect biocontrol agents and they are discussed here.

(a) Chrysoperla lacewings. Green lacewings are widely mass produced and sold to home gardeners and commercial growers by insectaries. The most commonly sold forms are European or Asian populations of Chrysoperla carnea (Stephens), which are part of a species complex. Such sales have potential to bring commercialized forms into contact with similar, but locally distinct lacewings. In such cases, there is an opportunity for hybridization. For example, in laboratory studies, the Japanese endemic species Chrysoperla nipponensis (Okamoto) (a member of the C. carnea complex) readily hybridized with the commercially marketed exotic form of C. carnea [93, 94]. For this reason, regions with rare or endemic green lacewings may want to prohibit importing closely related, exotic green lacewings from commercial sources [95].

(b) Chestnut gall wasp parasitoids. The Chinese gall wasp Dryocosmus kuriphilus Yasumatsu (Hymenoptera: Cynipidae) is a pest of chestnuts (Castanea spp.) that has invaded Japan and other areas. The Chinese parasitoid Torymus sinensis Kamijo (Hymenoptera: Torymidae) was introduced into Japan to suppress D. kuriphilus, where it came into contact with a closely related native Japanese species, Torymus beneficus Yasumatsu & Kamijo (Hymenoptera: Torymidae), of which two biotypes have been recognized. The introduced parasitoid subsequently hybridized with both of the two native biotypes at rates of about 1% (for the early-spring biotype) and 20% (for the later-spring biotype) [96]. However, despite this difference in hybridization rates, both biotypes of T. beneficus were eliminated in Japanese chestnut orchards [97, 98], suggesting that the mechanism of elimination was not solely hybridization but more likely due to displacement through competition for hosts.

(c) Laricobius adelgid predators. The predatory beetle Laricobius nigrinus Fender (Coleoptera: Derodontidae) has been moved from its native range in western North America (where it is a specialized predator of hemlock woolly adelgid, Adelges tsugae Annand) to the eastern USA for biological control of an invasive population of an invasive population of the same adelgid. Following relocation, L. nigrinus has hybridized to a degree with its native congener Laricobius rubidus LeConte, which mainly attacks adelgids on white pine (Pinus strobus L.). Hybridization occurs at a stable rate of 10–15% [99, 100]; hybridization occurs more often on hemlock (Tsuga canadensis [L.] Carrière) than on white pine, where L. rubidus dominates
Resource partitioning appears to be happening, with *L. nigrinus* increasingly becoming the dominant predator on hemlock, while *L. rubidus* remains dominant on white pine [100].

**Putting risk in context**

Hybridization between an introduced species and a local native congener, as described above, is not uniquely associated with biological control agents. Rather, many species moved by people for recreational or sport purposes have hybridized with closely related species when the two are moved into sympatry, in some cases endangering the native form. Well-known examples include the movement of rainbow trout (*Oncorhynchus mykiss* [Walbaum]) throughout the western USA into rivers and lakes where it endangers local trout species through a mix of predation, competition, and hybridization [101]. Similar outcomes have occurred in *Anas* ducks, where the introduction of the common mallard (*Anas platyrhynchos* L.) has led to extensive hybridization with closely related species, such as the grey duck (*Anas superciliosa* Gmelin) in New Zealand [102].

**Mitigation**

Tests to detect hybridization potential between species proposed for introduction and congeners living where releases are planned can be run in quarantine. For example, the proposed introduction of *Laricobius osakensis* Montgomery and Shiyake from Japan into the eastern USA was preceded by tests to measure the potential to hybridize with the previously introduced *L. nigrinus*. In this instance, successful interspecific mating was not detected [103]. In contrast, Naka et al. [93, 94] found high potential for hybridization between native Japanese green lacewings (*C. nipponensis*) and commercial *C. carnea* and warned against introduction of the commercially available populations.

**Type 5. Attack on weed biocontrol agents**

**The concept**

Some insect biological control agents can, depending on their ecology and host ranges, attack weed biological control agents that are similar, taxonomically or ecologically, to the targeted herbivorous pest.

**Examples**

Three examples of this are discussed below; others almost certainly exist. Examples discussed include an oligophagous weevil parasitoid (*M. aethiopoides*); braconid parasitoids of tephritid flies, a family that includes both fruit-infesting pests and gall-making weed control agents; and a predaceous mite that attacks spider mites, which mostly are crops pests, but have also been used as weed biological control agents.

(a) The oligophagous weevil parasitoid, *M. aethiopoides*. This parasitoid has been used successfully to control several pest weevils of forage crops [38, 42] and is known to attack some native weevils in New Zealand [44] (see earlier discussion of this case). Among the non-target weevils attacked is the introduced weed biocontrol agent *R. conicus*, which has controlled nodding thistle (*Carduus nutans* L.) in parts of the USA and New Zealand [104, 105]. In New Zealand, this weevil has been found to be parasitized by *M. aethiopoides* at rates up to 17% [106].

(b) Parasitoids of frugivorous tephritid flies. Several species of parasitoids, including *Diachasmimorpha longicaudata* (Ashmead), *D. tryoni* and *Psyttalia fletcheri* (Silvestri) (all Hymenoptera: Braconidae), have been introduced to Hawaii to attack invasive frugivorous tephritid flies. Investigations were later undertaken to determine if these species attacked the gall fly *E. xanthochaeta*, introduced to suppress invasive lantana. In the laboratory, the level of attack on *E. xanthochaeta* larvae by *D. longicaudata* or *P. fletcheri* was reduced but not eliminated if gall fly larvae were presented naturally inside their galls. If attack did occur, *D. longicaudata* developed successfully but *P. fletcheri* did not [107]. In contrast, both *D. tryoni* [108] and *Diachasmimorpha kroussii* (Fullaway) [109] (Hymenoptera: Braconidae) did attack some lantana gall fly larvae in laboratory trials. In the field, however, <1% of lantana gall flies were parasitized by *D. longicaudata* at sites where 37% of this parasitoid’s normal host (*Bactrocera dorsalis* [Hendel] [Diptera: Tephritidae]) were attacked [110]. Field attack rates, however, are not reported for the other parasitoids.

(c) Predatory phytoseiids attacking spider mites. The gorse spider mite, *Tetranychus latus* (Dufour) (Acar: Tetranychidae), has been released in New Zealand and the USA for control of gorse (*Ulex europaeus* L.). This spider mite, however, has failed to have any persistent, significant effect on gorse. Field studies in Oregon (USA) showed this was likely due to feeding on the spider mite by predatory phytoseiid mites, including *Phytoseiulus persimilis* Athias–Henriot, a non-native phytoseiid that established in Oregon after being released for control of pest spider mites in agricultural fields [111].

**Putting risk in context**

Attacks on weed biocontrol agents by locally existing parasitoids, while potentially damaging from a practical point of view, is a common phenomenon, occurring, for example, in about 40% of all weed biological control agents established in South Africa [112]. Such use of introduced herbivores by native parasitoids may or may not affect their population levels. Attack by *Mesopolobus* sp. (Hymenoptera: Pteromalidae) on rush skeletonweed gall midge (*Cystiphora schmidti*) (Diptera: Cecidomyiidae), for example, in Washington state (USA) did not prevent development of damaging levels of galls on the target weed [113], and rates of parasitism by native parasitoids on a biological control agent may vary greatly among locations.
or plant species [114]. Similarly, native predators may
attack herbivores introduced for weed biological control
[115, 116], reducing their efficacy in some cases [116].

Mitigation
Safety of new insect biocontrol agents to previously
released weed biocontrol agents can be determined during
host range testing for the new agent. What cannot be
avoided is potential future conflict with unspecified
weed biocontrol agents whose release might latter be
desired, unless their possible use is foreseen at the time of
the insect biocontrol agent’s proposed introduction. For
everyone, Nadel et al. [117], when estimating the host range
of Bracon celer Szépligeti (Hymenoptera: Braconidae) for
potential introduction to California against olive fruit fly,
Bactrocera oleae (Rossi) (Diptera: Tephritidae: Dacinae),
found the parasitoid could attack and successfully develop
in Parastreptus regalis Munro (Tephritidae: Tephritinae),
a gall making fly of interest as a potential weed control agent
for Cape ivy, Delairea odorata Lem. Consequently, B. celer
was rejected for introduction into California, at least until
it is clarified if P. regalis is going to be introduced.

How Common Have Population-level Non-target
Effects Been?

Deciding how best to assess the risk of biological control
introductions has become an important focus of classical
biological control of arthropods. New knowledge gained
from in-depth studies of particular cases over the last 30
years has improved our ability to assess risk and determine
how it can be lowered [118]. However, a comprehensive
review of results of all parasitoid and predator releases for
insect biological control has not been done and is not likely
to be done because of the constraints of resources and
scientific expertise. Consequently, any attempt to deter-
mine the frequency of such impacts devolves into collecting
all the cases for which an attempt to obtain such informa-
tion has been made (on the basis that cases with no data
do not tell us there are no impacts, but only that the case
has not been evaluated). It is less likely than for weed
biocontrol agents that the impacts of insect biocontrol
agents would be observed outside of deliberate scientific
studies.

Cases where data exist, however, are not a random
sample of all introductions, but rather seem to fall into
three groups, each with strong, but different biases. One
group consists of cases in which preliminary knowledge
suggested that non-target effects had or were likely to have
occurred and the researcher was interested in finding such
cases because they could produce positive, publishable
results that fit into a trending area of emphasis in the
science. A second group of studies consists of work by
biological control scientists who investigated historical
cases where non-target impacts were asserted, but data
were lacking. Such studies were often carried out either
because the scientist was located in the affected region or
had a personal interest in the system. The third group
of cases consists of more recent projects carried out by
biological control scientists who developed extensive
pre-release information (subject to stricter regulations for
new projects) or investigated consequences of previous
projects. The purpose of this work was to test hypotheses
developed during host specificity testing in quarantine after
agents were established in the field (i.e., were agents as host
specific as predicted).

This scarcity of well-developed studies on insect
biocontrol agents contrasts with weed biocontrol whose
herbivorous agents are generally large, visible and reason-
ably easy to collect and identify. As a consequence, the
number of recorded cases of non-target impacts by weed
biocontrol agents actually reflects the real number of cases,
and in this instance, it is probably reasonable to infer that no
information of non-target impacts means that no impacts
occurred. This strong difference between non-target impact
assessments for insect and weed biocontrol agents is not
likely to change because it is caused, in part, by the small
size and taxonomic complexity of insect biocontrol agents
and the often poorly understood native insect fauna in the
receiving environment.

Therefore our ability to assess the level of non-target
impacts for insect biocontrol agents (parasitoids and
predators) will be imperfect and will consist of collecting
and analysing published peer-reviewed information. We
should expect knowledge to increase as more effort in
this research area is made. However, these types of field
studies, reviews, or metastudies may be subjected to the
biases because of the research motivations listed above.
Here we discuss the literature as of 2016 to the best of our
knowledge, grouping studies as: (1) no impact on non-
target species, (2) population-level impacts through attack
or (3) indirect population-level impacts through mechan-
isms such as apparent competition or displacement through
competition for hosts or prey. For the third case, we
exclude displacement from an anthropogenic system (such
as a native parasitoid being displaced from attacking an
invasive pest on a crop); such evidence by itself does not
mean significant ecological impact because the native
natural enemy must have had a native host and its displace-
ment in this native habitat by an introduced agent(s) is,
in our opinion, the critical issue of most concern. At this
point, displacement of native natural enemies in native
habitat as opposed to agroecosystems has been inade-
quately addressed in previous studies assessing non-target
impacts of introduced biological control agents.

Below we discuss 22 past cases, selected by us for
purposes of this discussion, in which efforts were made
to detect non-target impacts. We grouped 12 of these as
showing no convincing evidence of significant impact, four
showing direct impact, and six showing alleged indirect
impacts via displacement, of which in four cases we argue
that displacement of native parasitoids from their native
hosts has not been shown.

http://www.cabi.org/cabreviews
Group 1. No impact

Bessa remota and leuvana moth

The introduction to Fiji of the tachinid fly B. remota (originally given as Psychomyia remota) successfully controlled a devastating pest of coconut, the defoliating moth Leuvara indescens Beth.-Bak. (Lepidoptera: Zygaenidae) [119–121]. This case is portrayed by Howarth [1] as the cause of extinction for two moths, the target L. indescens (asserted by Howarth to be native to Fiji) and another, certainly native, zygaenid called Heteropan dolens Druce. If both statements were well substantiated, this would be a case of great importance. However, neither assertion is supported by adequate evidence [22]. The parasitoid is native to the East Indies region [122] and is clearly polyphagous. Host range testing done 50 years later, when its introduction to India was being considered, found parasitism rates in the laboratory of 4–20% in larvae of eight Lepidoptera in various families [123]. However, the target pest on Fiji was considered invasive at the time of the original work [120, 122] and in later analyses [21, 22]. As for H. dolens, there are no records of this moth being attacked by B. remota, and this species may continue to exist on Fiji [22]. Consequently, there are no data to support claims that B. remota has caused the extinction of either L. indescens or H. dolens.

Australian mealybug parasitoids in New Zealand

A post-release monitoring program in New Zealand found that four species of Australian parasitoids (Tetracnemoidae sydneiensis [Timberlake], Anagrus fusciventris [Girault], Gyransoided advena Beardsley and Parectromoides varipes [Girault]) (all Hymenoptera: Encyrtidae) of longtailed mealybug (Pseudococcus longispinus [TargioniTozzetti]) that were accidentally introduced by commerce do not affect native mealybugs in New Zealand, which occur in native forest. Longtailed mealybugs placed in native forest on potted citrus were always unparasitized, in contrast to similarly deployed longtailed mealybugs placed in orchards, which were consistently parasitized. The native mealybugs Paracoccus glaucus (Maskell) and Paracoccus zealandicus (Ezzat & McConnell) placed in orchards on potted pigeonwood plants, Hedyccarya arborea J. R. Forst. et G. Forst., a native plant host of these mealybugs, were unparasitized by the exotic parasitoids. Collections of native mealybugs from native forest were parasitized by only native parasitoids. Collectively, these experiments and surveys show high specificity of these exotic parasitoids, probably because of an aversion to forage in forest habitats, and no change in the host ranges of any of the introduced parasitoids 14–47 years after their self-introduction [124].

Citrus blackfly parasitoids on the island of Dominica

A survey of 51 sites in the Carribean Island of Dominica by Lopez et al. [125] found a high degree of suppression of the target citrus blackfly, Aleurocanthus woglumi Ashby (Hemiptera: Aleyrodidae), and no instances of parasitism on other whiteflies (six species, a mix of native and introduced) by either of the two released parasitoids, Amitus hesperidum Silvestri (Hymenoptera: Platygasteridae) and Encarsia perplexa Huang and Polaszek (Hymenoptera: Aphelinidae).

Neotropical phytoseiid, Typhlodromalus aripo, in Africa

In Malawi and Mozambique, native mite communities on the introduced crop cassava (Manihot esculenta Crantz) were monitored for 2 years following the introduction of the phytoseiid predatory mite Typhlodromalus aripo De Leon for control of cassava green mite, Mononychellus tanojoa (Bondar) [126]. In Mozambique, densities of all the common phytoseiids on cassava – Euseius boetae (Meyer & Rodrigues), Euseius bwende (Pritchard & Baker) and Uckermannesius saltus (Denmark & Matthyssse) – remained stable during the study, despite establishment of T. aripo and its suppression of the target pest mite. In Malawi, two of the most common native cassava phytoseiids – Euseius fustis (Pritchard and Baker) and Iphiseius degenerans (Berlese) – increased in abundance, while that of the third species, U. saltus, was not affected.

Parasitoids attacking the endemic Hawaiian moth Udea stellata

Udea stellata (Butler) (Lepidoptera: Crambidae) is a common, non-threatened, endemic Hawaiian moth. Kaufman [127] examined sources of mortality affecting life stages of this moth and found seven polyphagous endoparasitoids attacking it: (a) three species likely moved accidentally in commerce: Casinaria infesta (Cresson), Trypta flavoorbitalis (Cameron) and Trichistus nr. aitkeni (all Hymenoptera: Ichneumonidae); (b) two species introduced for biological control: Meteorus lapthymgaea (Viereck) and Cotesia marginiventris (Cresson) (both Hymenoptera: Braconidae); and (c) two likely endemic species: Diadegma blackburni (Cameron) and Pristomerus hawaiiensis (Perkins) (both Hymenoptera: Ichneumonidae). The two biocontrol agents were introduced to Hawaii in 1942 to control the sugarcane pest Spodoptera exempta (Walker) (Lepidoptera: Noctuidae). Highest rates of apparent field parasitism were from the accidentally moved species T. flavoorbitalis and occurred mainly below 850 m elevation. The parasitoids introduced as biocontrol agents were detected in the target moth only above this elevation [127, 128]. Kaufman and Wright [129] explored these relationships more thoroughly, using demographic techniques such as life tables and marginal rate analyses. They found that the impact of parasitoids on U. stellata larvae was much lower than apparent parasitism had suggested, only about a 5% population reduction. The large difference between this finding and their earlier study was caused by a high rate of predation on larvae, which had not been accounted for previously. Furthermore, Kaufman and Wright [129] found that it was the accidentally introduced parasitoid T. nr. aitkeni that dominated the parasitoid guild (48.5% of all parasitoids reared in this study), not the accidentally introduced species T. flavoorbitalis, as reported earlier.
This study clearly illustrates the ease with which field data drawn from simple samples, unaided by a demographic analysis framework, can be misleading. It also suggests that accidentally introduced parasitoids (never subjected to selection criteria) can be more damaging to local native species than biological control agents. We suggest that these two types of invasions, deliberate (i.e., intentional release of biological control agents) and accidental (i.e., self-introduction or via the live plant trade), should be distinguished during assessments of impact on non-target species.

Peristenus digoneutis (Hymenoptera: Braconidae)
This European parasitoid of certain species of Lygus mirid bugs was introduced into eastern North America to suppress a native species, *Lygus lineolaris* (Palisot de Beauvois). Before this introduction, the target pest was parasitized by a presumed native euphorine braconid, *Peristenus pallipes* (Curtis) at a low level (9%) [130]. However, it may be that *P. pallipes* is itself invasive, as it parasitizes at a high rate only two invasive European mirids [131].

After its introduction into the eastern USA, *P. digoneutis*’ effects on mirids and their parasitoids were assessed over a 19-year period by Day [130], who found that parasitism of *L. lineolaris*, the target pest of the biocontrol program, increased to 64% and its density dropped by two-thirds. The parasitoid *P. pallipes* remained present in the system throughout the study. Some individuals of the mirid *Adelphocoris lineolatus* (Goeze) were parasitized, but its density was not reduced. *Leptoperna dolabrata* (L.), a European grass-feeding species, was not attacked by *P. digoneutis*. These observations suggest that the introduced parasitoid reduced the target host’s density without damaging populations of either its native parasitoid or those of other mirids found in the same habitat.

For this same system, Haye et al. [132] assessed the value of laboratory host range test results as a predictor of field events. They did this by first assessing rates of *P. digoneutis* parasitism in the laboratory for a range of European mirids and then measuring parasitism of the same species collected from their native habitats in Europe. They reared *P. digoneutis* from ten field-collected hosts — three species of *Lygus* and seven non-lygus species in the subfamily Mirinae. These findings were consistent with laboratory testing, showing that all seven non-target species that were parasitized in the laboratory were also attacked and successfully parasitized in the field. However, rates of parasitism observed in the field were low (<1% for 8 of 10 species), in contrast to laboratory parasitism (11–100%, by species). Haye et al. [132] suggested that such native range host surveys can help interpret quarantine data on parasitism, given that in small cages there is no need to find host habitats or hosts, as would be necessary in the field. So, while negative data in small cage laboratory studies probably indicate a high degree of safety to rejected species, the meaning of acceptance of species for parasitism under confined laboratory conditions is more ambiguous.

In summary, the introduction of *P. digoneutis* into the eastern USA for lygus bug control appears to have achieved its goals without population-level non-target impacts. *Peristenus digoneutis*, however, has also been released (since 1998) into the western USA [133], where there is a larger set of potential non-target mirids. Mason et al. [134], considering the possible effects of *P. digoneutis*, concluded from laboratory testing that native *Lygus* spp. in the region were at risk of being parasitized, but other regional non-target mirids were not. Information on actual field outcomes in western North America is not yet available and is needed.

Torymus sinensis Kamijo (Hymenoptera: Torymidae) in Italy
This parasitoid of the chestnut gall wasp, *D. kuriphilus*, has been released in several countries invaded by *D. kuriphilus*. Following its release in Italy, instances of non-target attack were sought by collection of a total of 1371 non-target galls (nine species of gall makers) in north-central Italy over a 2-year period from four species of oak and one of wild rose [135]. Five native Torymid parasitoids were reared from the collected galls but *T. sinensis* was recorded from only one non-target galls wasp, *Biothiza pollida* Galle (Hymenoptera: Cynipidae), from which two males of *T. sinensis* were reared. These field records are consistent with the fact that in the laboratory all the non-target galls tested were unsuitable for *T. sinensis* oviposition, except for the cynipid *Andricus curvar* Milan Zubrik.

Rodolia cardinalis in the Galápagos
Seven years after this lady beetle’s release in the Galápagos, Hoddle et al. [136] evaluated the effects of *R. cardinalis* (Mulsant) (Coleoptera: Coccinellidae), released for control of the cottony cushion scale, *Icerya purchasi* Maskell (Hemiptera: Monophlebidae), on native insects on the islands to compare observed outcomes with quarantine predictions. Before release, up to 60 native or endemic species of plants on the islands were affected by the scale, causing population declines of some critically endangered plants and associated specialized insects [137, 138]. The assessment (2009–2011) found the project to have been safe and effective [136]. On evaluated plant species, scale densities were reduced by ~60–98% compared with pre-release surveys. Most native plants surveyed were no longer heavily infested by the scale, with the exception of the dune-inhabiting *Scaevola plumieri* (L.) Vahl., which still supported substantial, but fluctuating scale populations. Also, in urban areas, scale-tending by invasive ants kept scale populations high. During 22 h of field-cage observations, *R. cardinalis* adults were offered five non-target arthropod species. A total of 351 predator/prey encounters were observed, 166 with *I. purchasi* and 185 with non-target prey. Encounters with cottony cushion scale resulted in 53 attacks (32% rate) but none of the 185 encounters with non-target species resulted in attacks [136]. Collectively these studies demonstrated that this introduced natural
enemy was beneficial to the biota of the Galápagos Islands and was without observable negative consequences.

Pteromalus puparum on Bassaris butterflies in New Zealand
The butterfly known as the yellow admiral, Vanessa (Bassaris) itea (F.) (Lepidoptera: Nymphalidae), is listed by Lynch and Thomas [24] as having been significantly affected by the pupal parasitoid P. puparum (L.) (Hymenoptera: Pteromalidae) (introduced against P. rapae [L.]). On the strength of a personal communication by George Gibbs, field studies assessing the impact of P. puparum on V. itea showed that in natural habitats parasitism rates by this species were low (~7%), but they increased to ~73% if study populations were in close proximity to P. rapae populations [139]. Despite this, Hicks [139] concluded that the most important factor depressing populations of V. itea was loss of its larval food plant, a stinging nettle (Urtica sp.), and Patrick and Dugdale [140] do not list V. itea in their summary of threatened New Zealand Lepidoptera.

Impacts of this same parasitoid on another New Zealand butterfly, the red admiral (Bassaris goneilla [F.]) (Lepidoptera: Nymphalidae) is not mentioned by Lynch and Thomas [24], but an impact was similarly presumed to have been caused by P. puparum [141]. Further analysis, however, using field data and a population growth model [142] found that P. puparum’s impact (5%) was minor compared with another generalist pupal parasitoid, Echthromorpha intricatoria (F.) (Hymenoptera: Ichneumonidae), an accidentally introduced species. This ichneumonid parasitoid reduced the butterfly’s density in the same modelling analysis by an estimated 30%.

Trigonospila brevificies in New Zealand
The tachinid T. brevificies (Hardy) was introduced into New Zealand against light-brown apple moth, Epiphyas postvittana (Walker) (Lepidoptera: Tortricidae). It was later found attacking several native tortricids (144). Of all parasitoids individuals reared from the sampled tortricids, T. brevificies comprised 15.6–79.5% of the total. However, rates of parasitism by T. brevificies on individual host species were not given, but rather it was stated that the whole parasitoid guild caused 13–26.5% parasitism (by host species) [143]. Without rates of attack by T. brevificies on individual host species and without a lifecycle-based understanding of their meaning, we conclude that there is as yet no evidence of population-level impacts by this parasitoid on non-target tortricids in New Zealand.

Trichopoda giacomellii (Diptera: Tachinidae)
The tachinid T. giacomellii (Blanchard) was introduced into Australia in 1996 for control of the stink bug N. viridula (L) following study of its likely host range [144]. Initial laboratory studies found that three non-target bugs were attacked and supported tachinid development: Plautia affinis Dallas, Alciphron glaucus (F.) and Glaucias amyoti (White) (all Hemiptera: Pentatomidae).

After establishment of the tachinid, field studies in New South Wales were conducted in 1999–2000 to measure its relationships with non-target pentatomids and scutellerids. Information was collected from 11 plant species, which collectively supported nine pentatomid species and two scutellerids. Some 1686 host individuals, summed over all species, were examined. Of the 11 bug species collected, nine were not attacked at all, one species had one parasitized individual out of 369 (0.03%), and one, P. affinis, had an overall parasitism rate of 4.8% (21/441), although at individual collecting sites, rates of parasitism ranged from 0.5 to 50%, effects likely mediated by attraction to the host plant. Only attack on P. affinis might rise to the level of population-level impacts, but only in selected locations [145].

Parasitoids of frugivorous and native gall-making tephritids in Hawaii
Efforts to control pest tephritids in Hawaii that attack fruits or coffee berries have included screening for attack by parasitoids of these pests on native tephritid gall makers. This was done either during consideration of new parasitoids for release or, for species released in the past, as later follow-up studies. The effort examined the propensity of seven parasitoids to probe or attack larvae of Trupanea dubautiae (Bryan), a native gall-making tephritid that infests flowerheads of the native composite shrubs Dubautia rairadioides Hillebrand. Studies included laboratory studies and, for previously released species, field surveys. Duan and Messing [146] found that neither D. lonicaudata nor P. fletcheri attacked T. dubautiae larvae in intact galls in laboratory tests. In a further study, Duan and Messing [147] found no attack on this same gall maker by another parasitoid, Tetrastrichus giffardianus Silvestri (Hymenoptera: Eulophidae), under laboratory conditions, nor in a field survey on Kauai. A fourth parasitoid, D. kraussi, also did not attack this gall maker in laboratory tests [110]. Wang et al. [148] also found no attack in laboratory tests of this same gall maker by any of three additional parasitoids: Fopius caudatus (Szépligeti), Fopius ceratitivorus Wharton and F. arisanus (all Hymenoptera: Braconidae). Collectively, these studies indicate no risk to this native gall maker from any of these seven introduced parasitoids.

Group 2. Direct trophic impact

Tamarinia (formerly Tetrastichus) dryi in La Réunion
On the island of La Réunion in the Indian Ocean, the parasitoid T. dryi (Waterston) (Hymenoptera: Eulophidae) was introduced during a successful programme to control two introduced psyllids that vector bacteria causing citrus greening disease. This case is listed in a review of non-target impacts by van Lenteren et al. [25] as causing ‘reductions in population levels’ of a local psyllid whose name was given as Trioza eastopi Orian [149], but which is a junior synonym of Trioza litseaef Bordage. This psyllid is known only from two

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islands: La Réunion, where it is pest of vanilla cultivation [150] and Mauritius (Diana Percy, personal communication). On La Réunion, populations were high on a widely planted, introduced shrub, *Litsea chinensis* Jacq., which is a traditional medicinal plant from the Andhra Pradesh region of India. While this psyllid may be native and endemic to La Réunion, it is possible that it may not be, and it could have arrived on *L. chinensis* from India. Uncertainty over the area of origin for *T. litsea*, and its abundance on La Réunion, need to be clarified. Until *T. litsea* is confirmed to be a native species and to be endangered by *T. dryi*, the ecological importance of its reduction in density remains unclear and somewhat doubtful.

**Brachymeria lasus and two native butterflies on Guam**

In Guam, native butterflies have experienced considerable decline. To understand if this was linked to species introduced for biological control, Nafus [151] measured apparent mortality rates for life stages of two native nymphalid butterflies on Guam: *Hypolimnas anomala* (Wallace) and *Hypolimnas bolina* (L.). For the egg stage, native ants were the dominant source of mortality for both species. In neither case did an introduced biological control agent cause important levels of egg parasitism. For larvae, a pathogen was an important source of mortality and larval parasitoids were not found. Only in the pupal stage did a biological control agent, *B. lasus* (Walker) (Hymenoptera: Chalcididae), cause significant levels of mortality, but only for *H. bolina* (25%). These findings demonstrate use of this species as a host in the field by *B. lasus*. However, since data were not placed in a lifetable context so that marginal attack rates could be calculated from apparent mortality rates, the actual population-level significance of this parasitism estimate and the subsequent importance of parasitism by *B. lasus* are unclear.

**C. glomerata in the Canary Islands**

Lozan et al. [152] detected *C. glomerata* on the island of La Palma in the western Canary Islands, where it was found parasitizing an island endemic pierid butterfly, *Pieris cheiranthis* (Hübner). While rates of attack are not documented, it appears that the butterfly, a forest species, is principally in contact with the parasitoid at forest edges and not inside intact forests. This observation is consistent with evaluations in Massachusetts, which found that *Pieris virginia* (Edwards), also a forest species, was not attacked by *C. glomerata* inside forests [153]. In the Canary Islands, *C. glomerata* was not introduced as a biocontrol agent, having likely hitchhiked on traded goods.

**Peristenus relictus Loan (≡ *P. stygicus*) and the western tarnished plant bug**

This parasitoid, introduced into the western USA against the native western tarnished plant bug (*Lygus hesperus* Knight) (Hemiptera: Miridae), is an oligophagous parasitoid of mirid bugs, including *L. hesperus*, *L. lineolaris*, *Polymerus basalis* (Reut.), *Labopidicola geminatus* (Johnston) and *Psallus seriatus* (Reut.) (= *Pseudatomoscelis seriatus*) [154]. In northern Germany, part of the native range of *P. relictus*, the ecological host range of this parasitoid includes at least 16 mirids in the subfamilies Mirinae, Phylinae or Bryocorinae [155]. These data suggest that *P. relictus* is a generalist mirid parasitoid. However, it was not the primary source of parasitism of most of its hosts [155] and appears to have only minor population-level effects on those it attacks. In laboratory tests in western North America, *P. relicitus* was found to attack and develop in a number of non-*Lygus* mirids, including *Amblytulys nasutus* (Kirsch.), *L. dolabrata* (L.) and *Melanotrichus coagulatus* (Uhler) [134]. Post-release field studies are needed to determine if *P. relicitus* has population-level effects on native non-target mirids in its introduced North American range [134].

**Group 3. Displacement or other indirect impacts**

Several cases of displacement of native parasitoids by introduced parasitoids are listed by Bennett [59], Lynch and Thomas [24], and van Lenteren et al. [25]. But a close examination suggests some of these reports may not be ecologically important. Of the 17 cases listed in Table 2 of Lynch and Thomas [24] as having significant effects on non-target species, four (*C. concinnata, M. aethiopoides, T. pallipes, C. septempunctata*) seem likely or possible cases of important impact on non-target native species, and these have been discussed in earlier sections. Another six cases of presumed displacement (two for *C. flavipes*, two for *A. holoxanthus, C. noacki* and *T. brevifacies*) seem to be cases with no ecological importance for native species (for several differing reasons, as discussed below), and their inclusion in Table 2 of Lynch and Thomas [24] may be misleading. The problem here lies with labeling a case as one of impact or displacement as it invites further repetitive citation without consideration of relevant underlying details. One further case in Lynch and Thomas [24], that of *P. puparum* and the yellow admiral in New Zealand, has been discussed above under no impacts, as the impacts of this introduced parasitoid were demonstrated to be unimportant at the population level [139]. Here below we provide details for additional cases where displacement is claimed by Bennett [59] or Lynch and Thomas [24].

**Cotesia flavipes Cameron in Trinidad and Brazil**

This Asian braconid parasitoid was introduced from India and Pakistan into the Caribbean and, later, throughout the sugarcane-producing regions of Latin America against the sugarcane borer *Diatraea saccharalis* (F.) (Lepidoptera: Crambidae). This and three other economically important species in the genus *Diatraea* are considered native to the Americas, and historically they supported several native parasitoids. One of these borers, *Diatraea lineolata* (Walker), is a maize stock borer attacked by the native braconid *Apanteles diatraeae* Muesebeck, typically at about the 10% level [156].
In Trinidad, after the build-up of *C. flavipes*, parasitism of *D. lineolata* by *A. diatraeae* was undetectable in a 1984–1985 survey [59], suggesting that this species had been displaced by the newly introduced parasitoid. However, the time period over which surveys were conducted was relatively short and Trinidad is only a small part of the range of this parasitoid, which also includes Mexico. In Mexico, Rodríguez-del-Bosque and Smith [157] detected *A. diatraeae* at a low level on another borer, *Diatreaea mullerella* Dyar & Heinrich, in Guerrero, Mexico, and noted that it was a common parasitoid of several species of *Diatreaea* throughout Mexico. Similarly, Tejada and Luna [158] found it to be the dominant parasitoid of *Diatreaea* spp. larvae in the state of Nuevo León in Northern Mexico. These records, while in need of greater amplification (and possibly molecular level work to confirm species identities), demonstrate that the introduction of *A. flavipes* has not caused widespread displacement of *A. diatraeae* in Latin America but rather may have displaced it only locally (in Trinidad) or from only one of its hosts. The current status of *A. diatraeae* in Trinidad should be reassessed.

Bennett [59] also reported effects of *C. flavipes* in Brazil (following its 1978 introduction from Pakistan) on the abundance of two native tachinids. These effects were characterized as ’the native tachinid parasitoids Metagonistylum minense and *P. claripalpis* have become scarce. While they are no longer represented in survey collections in many fields, they occur sporadically in collections from other fields.’ Also, Trejos et al. [159] recorded the presence of both of these tachinids in the Cauca Valley in Colombia. These survey results suggest that there are likely important temporal and spatial effects on the abundance of native and introduced parasitoids and the hosts that they share. Surveys should be of sufficient duration and across many study sites, so that robust conclusions can be drawn about population-level impacts from natural enemy introductions.

*Aphytis holoxanthus* DeBach (Hymenoptera: Aphelinidae) This parasitoid has controlled the armored scale *Chrysomphalus aonidum* (L.), which is native to Asia, but is widely invasive in several citrus-producing regions around the world. It is listed by Bennett [59] and Lynch and Thomas [24] as being responsible for displacing two native parasitoids: one in Florida, *Pseudohomalopoda prima* Girault (Hymenoptera: Encyrtidae), and one in Brazil, *Aphytis costalimai* (Gomes) (Hymenoptera: Aphelinidae). However, in both cases, the same sequence of events seemed to have happened. First, a native parasitoid moved from native hosts and habitats into citrus groves (an artificial habitat created by people with an introduced tree) where it attacked an introduced Asian scale (*C. aonidum*) and became common on that host. Later, because control by native parasitoids was insufficient, the specialized parasitoid *A. holoxanthus* was introduced (into Florida in 1960 and Brazil in 1962). *Aphytis holoxanthus* became the dominant parasitoid on *C. aonidum*, removing it as an available high-density resource for local native parasitoids that had been opportunistically exploiting the uncontrolled scale populations. This replacement does not mean, however, that native parasitoids suffered a negative ecological impact. Rather, they lost a previous gain due to the proliferation of a pest species in a man-made ecosystem, the citrus crop. In the case of *P. prima* in Florida, Bennett [59] recorded that this parasitoid remained the dominant parasitoid of the diaspidid scale *Acutaspis morrisonorum* Kosztarab on southern red cedar, *Juniperus virginiana* var. *silicola* (Small) Bailey. This same scale occurs on several native pines in the southern USA, including *Pinus taeda* L. and *Pinus echinata* Mill. [160]. More recently, Ceballos et al. [161] reported collection of *P. prima* from *Aspidiotus destructor* Signoret on coconut (*Cocos nucifera* L.) in Cuba. As for the parasitoid in Brazil, Terán et al. [162] reported *A. costalimai* from scales on citrus in northern Argentina 23 years after the introduction of *A. holoxanthus* to the region. These records suggest that both of these native parasitoids remain present on various native scales infesting non-crop plants and rarely being collected unless they attack a pest scale on an economically important crop.

*Aphytis holoxanthus* DeBach (Hymenoptera: Aphelinidae) This parasitoid was introduced into Europe to control the whitely *Aleurothrixus floccosus* Maskell, and Lynch and Thomas [24] list Vigiani [163] (also repeated by van Lenteren et al. [25]) as recording it as displacing *Encarsia margaritiventris* Mercet (Hymenoptera: Aphelinidae) from the viburnum whitefly, *Aleturothrixus jelinekii* (Frauenfeld), a native species in Europe. Little is known about the host range of *E. margaritiventris* as there are few published records, but it is likely not monospecific, as Malumphy et al. [164] recorded it as being reared from the whitefly *Aleurochiton aceris* (Modeer) in Lithuania. More data from field surveys are needed to evaluate this case.

*T. brevifacies* (Hardy) (Diptera: Tachinidae) This tachinid, introduced into New Zealand to control light-brown apple moth (*E. postvittana*), is recorded by Lynch and Thomas [24] through Roberts [165] as significantly harming the parasitoid *Xanthopimpla rhopaloceros* Kreiger (Hymenoptera: Ichneumonidae). This latter parasitoid, however, is not native to New Zealand, having been introduced from Australia as part of the same biocontrol program targeting *E. postvittana* [166]. Both parasitoids coexist sympatrically in New Zealand where they attack light-brown apple moth [143].

*Diadegma semicalamus* (Hymenoptera: Ichneumonidae) Two additional cases of apparent displacement not reported by Bennett [59] or Lynch and Thomas [24] were noted in this review of the literature: the ichneumonid *D. semicalamus* (Hellén) in Africa and various parasitoids introduced into the USA against the tobacco whitefly, *Bemisia tabaci* (Gennadius) strain B (also known as *B. argentifolii*) (discussed below).
D. semicalausum was released in Kenya in 2002 to control a cabbage pest, the diamondback moth, Plutella xylostella (L.) (Lepidoptera: Plutellidae). This release increased parasitism of diamondback moth larvae from 14 to 53% and consequently lowered crop damage. At the same time, rates of attack on the pest by several native parasitoids decreased. Attack rates on the pest by Diadegma molipla (Holmgren) (Hymenoptera: Ichneumonidae) and Oomyzus sokolowskii (Kurdjumov) (Hymenoptera: Eulophidae) on cabbage in Kenya decreased from 5.4 to 2.8% and 9.0 to 2.2%, respectively [167]. Is such an impact significant to populations of these native parasitoids? In addition to attacking diamondback moth in cabbage fields, these native parasitoids also attack it on a variety of wild crucifers (weeds or native plants), where they were found co-existing with the introduced parasitoid 3–4 years after its release [168]. Also, these native parasitoids remained present, in lower numbers, 3–4 years post release on diamondback moth in cabbage fields [169]. These native parasitoids are known to be widely distributed in southern Africa, having been recovered from diamondback moth, for example, in South Africa [170]. D. molipla has also been recorded in the literature from the potato tuberworm (Phthorimaea operculnea [Zeller]) (Lepidoptera: Gelechiidae) in Egypt [171]. Since this record is of a host in a different family, it should be confirmed.

While much is not known about the native hosts and habitats of these non-target parasitoids, these records from crop studies suggest both that their densities have been lowered in in cabbage fields in some areas, but also that they are widespread geographically, found on many host plants, and several insect species, which likely ensures their continued population-level well-being. Such instances of displacement, in the view of the authors, do not represent loss of biodiversity because of introduced natural enemies.

Parasitoids introduced into the USA against tobacco whitefly

In response to large financial losses in cotton, winter vegetables, melons and greenhouse crops from the invasion of the B strain of the tobacco whitefly (B. tabaci), some 20 parasitoid populations were collected from this species in many countries and introduced into the southwestern USA [172]. Prominent among 11 released parasitoid populations (species × country combinations) was Eretmocerus mundus (Mercet) (Hymenoptera: Aphelinidae) from Spain [173]. Of five species released in California, E. mundus later was found in a 10-year survey to have become the dominant parasitoid on B. tabaci on cotton in California and to have displaced the native Eretmocerus species formerly attacking B. tabaci on that crop [174]. However, when non-cotton host plants were surveyed for whiteflies and their parasitoids, it was found that E. mundus did not attack either of two likely native whiteflies – the banded-wing whitefly (Trialeurodes abutilonae [Haldeman]) on sunflower (Helianthus annuus L.) or mulberry whitefly (Tetraurodes mori Quaintance) on mulberry (Monus sp.) [174]. Furthermore, the native parasitoids formerly attacking B. tabaci on cotton, Eretmocerus eremicus Rose and Zolnerowitch and Eretmocerus jacobilli Rose and Zolnerowitch, were found attacking banded-wing whitefly and mulberry whitefly on their respective host plants, indicating that displacement of these parasitoids by E. mundus was primarily from the introduced B. tabaci on cotton and not from other whitefly hosts on different plants [174].

As part of the same program, introductions into Arizona of the same suite of parasitoids resulted in the displacement (from B. tabaci on cotton) of two native species – E. eremicus and Encarsia meritans (Gahan) – by the exotic parasitoids Eretmocerus sp. (Ethiopia) and Encarsia sophia (Gahan) in the early 2000s [175]. Information has not been published, however, concerning the status of these native parasitoids in Arizona on other species of whiteflies on other plants. It is quite possible that niche division, rather than general displacement, is also at work in Arizona, and this possibility needs to be resolved.

Looking Ahead: What Impacts Will New Parasitoid/Predator Introductions Have On Non-Target Species?

Forecasting likely host use

Avoidance of non-target effects from new introductions of parasitoids or predaceous arthropods is based on estimating fundamental host or prey ranges and releasing only species that are adequately specific for where they will be released, where they might naturally spread, and where they have a high risk of being accidentally transported [176, 177]. Part of the selection process is correct species-level recognition of the candidate natural enemy, as candidate natural enemies may be collected from a species complex whose aggregate host range is larger that that of some of its member species [178–180]. Estimating host ranges of parasitoids and predators was considered unimportant until about 1990 because non-target insects were considered to generally be of little economic importance and unimportant as species for conservation [181]. Methods for estimating parasitoid and predator host ranges were developed as extensions of methods used earlier for weed biocontrol agents and are reviewed by Van Driesche and Reardon [182] and discussed by van Lenteren et al. [183] and Babendreier et al. [176]. Here, framed around some key ideas, we discuss more recent contributions to methods for determination of host ranges.

Herbivore host range estimation, for weed biocontrol, seeks to understand the taxonomic limits of what an agent’s adults or offspring can eat, if given the opportunity. The assumption is strongly and correctly made that if plants are closer taxonomically to the target weed, they will be inherently at greater risk of being eaten by the agent [184].

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Tests assess what the adult and immature stages will eat, what host species the agents can feed on to maturity, and what plants adult agents will lay eggs on when given access to test species in small cages. Trials are either run one plant species at a time or in pairs (or larger groupings), where one species is the target pest. Alternatively, agents may be offered target and non-target plants in various sequences over time. Small-cage tests in quarantine cannot assess the ability of natural enemies to orient to a plant from a distance, distinguish it upon contact, and chose a preferred plant among locally available hosts.

Given this history, one should ask whether estimating parasitoid and predator host/prey ranges would be a simple extension of methods developed for herbivorous natural enemies. Consider the following. First, plants frequently defend themselves against herbivores with secondary plant compounds, which once developed by a lineage of plants tend to be conserved. These specialized compounds tend to deter generalist-feeding insects that lack an ability to survive possible intoxication following consumption, but these same compounds often are specific attractants for the specialists associated with the plant lineage [185]. Insects, in contrast, generally do not produce specialized chemical defences, although some species sequester toxins from their host plants, such as the cardiac glycosides obtained by larvae of monarchs, Danaus plexippus (L.) (Lepidoptera: Nymphalidae), from milkweeds (Asclepias spp.). The correlation between herbivore host ranges and plant taxonomy is a core theoretical tenant for screening weed control agents. This approach, however, is weak when applied as the basis to determine the host ranges of insect control agents [186]. Specialized secondary compounds do not have a large influence on parasitoid and predator host ranges, but host taxonomy still functions as a partial predictor of risk for non-target species based on: (1) the general morphology of potential host insects, (2) the manner in which non-target species feed on host plants and (3) where non-target species live in the physical environment. Beyond host taxonomic position, several other important factors must be considered when attempting to assess the host range of entomphagous natural enemies.

The first of these factors is that host odours and odours from the insect’s host plant are important attractants for many parasitoids and predators, which orient from a distance towards the plants on which their hosts or prey are feeding [187–189]. Such plant volatiles also play an important role in host finding by herbivorous insects, but this factor has not been widely used in estimating herbivore host ranges because it requires use of olfactometers or wind tunnels to assess long distance responses to odours from different plants, and the use of these devices in quarantine may be difficult due to space limitations. For herbivores, it has been possible to generally ignore the need to assess ‘attraction from a distance’ because the secondary plant compound signal is strong and its effects are easier to measure in the laboratory. But for parasitoids and predators, the absence of a strong signal analogous to that provided to herbivores by secondary plant compounds makes the ‘attraction from a distance’ factor more important to assess.

Second, plant tissues, unlike those of insects, usually do not have mechanisms (other than plant chemistry) that actively attempt to kill attackers. In contrast, insects have blood cell-based immune systems that attempt to defeat parasitism through mechanisms such as encapsulation, and if successful, such measures limit the host ranges of internal parasitoids [190]. In response, parasitoids have developed countermeasures to defeat encapsulation, such as the use of polydnaviruses by braconids and ichneumonids [191].

Third, parasitoid and predator host/prey ranges are typically less specialized than those of specialized herbivores used as weed biocontrol agents. The challenge, then, is how are we to estimate the range of hosts whose populations are likely to be reduced by a parasitoid (population-level impact) based on results from laboratory testing, given that this strongly affected group will be some subset of all hosts that the parasitoid can attack. Minor levels of attack on some non-target species by parasitoids is likely to occur during quarantine testing. However, such attacks may not translate into significant population-level impacts in the field, and this possibility needs consideration when data from quarantine tests are being analysed and interpreted.

This distinction between host use under quarantine conditions and population-level impacts in the field was clearly stated by Blossey [192]. How, therefore are predictions about population-level impact to be made from laboratory data designed to measure host use? Several alternative methods of investigation have potential to do so, including literature surveys [193], field surveys in the agents’ native range [194], post-release monitoring in the area of release [193] and population modelling [142]. What is currently needed is to expand the inventory of well-studied cases that allow us to examine the strength of such methods for assessing risks of significant non-target impacts and identify reasons for exceptions to anticipated outcomes.

**Moving from host use to population-level effects**

Post-release, estimates of population-level effects on non-target species caused by deliberately introduced biological control agents can be made with life table studies [129], studies of impact using cohorts deployed on host plants [51] or over physical gradients [51], or through the application of population models that use field-collected demographic data [11, 142].

Pre-release prediction of likely non-target impact, however, cannot use the above methods because the agent is not yet present in the field in the country of intended release. Paynter et al. [195] propose a pre-release
method for predicting host use by weed biocontrol agents on non-target plants based on the ratio, in quarantine tests, of attacks on the non-target versus target plants. This method, however, only predicts host use, not population-level impact and the approach may not be applicable to entomophagous natural enemies. Wright et al. [196] presented a method for assessing risk of use of a non-target species from augmentative release of an egg parasitoid (Trichogramma ostrinae Pang and Chen, Hymenoptera: Trichogrammatidae) that assessed risk as the cumulative probability over a series of links in a decision tree (e.g., probability of dispersing to non-target species × probability of attacking non-target species, etc.). In principle, this system could be applied to classical biological control agents provided necessary information on habitat use, density in habitat, and attack rate on non-target species could be developed. Risk of introductions, particularly for augmentative biocontrol agents, has also been discussed in terms of the product of risk of establishment × dispersal × host range, making it possible to estimate effects on native species. This was done, for example, for eight species of predatory mites introduced into Japan [197]. Such schemes, however, while using knowledge and judgment, are fundamentally forecasts, not facts.

What is needed to improve understanding of the potential impacts of insect biocontrol agents is to conduct replicated (across sites and over time) longitudinal studies that are designed to assess, population-level impacts in the field and compare those estimates with rates of attack on non-target species in pre-release tests. For new agents undergoing release consideration, this would mean conducting impact studies on non-target species of interest. For agents released without host range testing, such studies would require carrying out both field impact studies and after-the-fact laboratory host-specificity tests.

From host impact to determination of a project’s risks and benefits

Ideally regulatory agencies should compare potential benefits to program costs (known or anticipated), including monetary costs and ecological damage to non-target species. Predictions of non-target impacts made before releases are largely educated guesses, based on some sense of the likely host range of the agent as determined from quarantine studies or literature reviews, how attack might translate into population-level impacts, and the value of the non-target species likely to be affected. These negative effects then have to be compared with the ecological damage or economic costs that might reasonably be expected if the pest is not controlled, together with an estimate of the chances of successfully controlling the pest. While most of the above quantities are rarely precisely known, the benefits and losses of such actions are easiest to compare if they are in the same currency (either both ecological damage or both economic losses). When targets are agricultural pests that do not affect natural areas, benefits to nature are indirect in the form of reduced use of pesticides and lower levels of environmental contamination. Direct economic benefits to farmers [198–200] are part of the ‘benefits ledger’ and are not required as part of these analyses, but can be very important for justifying programmes. In countries with acts intended to protect endangered species, risk to those species may override the larger picture of risks and benefits, as such laws may, as in the USA, be inflexible with regard risk to the protected species, placing their interests above all else.

Risk analysis is complicated, and factors that will need to be taken into account will vary by project. Hoelmer and Kirk [201] discuss how several lines of information can be combined to improve selection of biological control agents. Some risk modelling suggests that even non-target species that are low on the agent’s preference scale may be harmed if the agent builds quickly to very high densities when the host is still abundant and if during this period the agent spills over to a small non-target population (i.e., apparent competition [202]). This possibility is not yet part of main stream risk analysis and would likely be very difficult to estimate accurately, but it should receive further investigation, especially if the suspected impact is not going to be transitory. In the context of weed biocontrol agents, it has been suggested that such effects can be minimized by avoiding the release of agents that fail to control the host (and thus remain abundant). The application of this idea to parasitoids is unknown. C. concinnata, for example, was highly effective in control of one of its target hosts (brown-tail moth, Euproctis chrysorrhoea [L.]) and yet is a wide generalist with important non-target effects as discussed above.

Summing up the risks for an agent’s introduction was attempted by Wyckhuys et al. [203] for Binodoxys communis (Gahan) (Hymenoptera: Braconidae) released against soybean aphid, Aphis glycines Matsumura, in North America. This summation was based on information about host suitability (as seen in laboratory tests), seasonal overlap of the parasitoid with susceptible hosts, and protection of native aphids by physical refuges or ant-tending. Ultimately such risk assessments describe probable risks, and it remains the job of regulators to decide on behalf of society if the risks to non-target species posed by introductions of natural enemies are warranted.

Another factor bearing on accuracy of risk prediction is selecting appropriate native species for non-target testing in quarantine. Barratt et al. [204] describe a new tool (PRONITI) that is intended to strengthen this process. As a test case, they applied the tool, as an after-the-fact exercise to M. aethiopoides’ 1982 introduction to New Zealand, since a great deal is known about its subsequent relationships with native species of non-target weeds. The exercise concluded that if PRONITI had been used, many of

http://www.cabi.org/cabreviews
the species subsequently attacked would have been chosen for host-range testing, and thus use of this system would have provided a much clearer assessment of the agent’s likely host use. Population-level impacts (as opposed to predicting possible non-target use), however, are not predictable using PRONTI

Recent practice (1985–2016) as predictor of future non-target impacts

A question of interest for this article, in addition to compiling and analysing records of past impacts, is whether safety practices used by biological control practitioners are improving and reducing risk to non-target species. Improving practice is based on better science (e.g., technical aspects of host range and risk estimation) and policy (e.g., societal goals and institutions that determine what risks are acceptable or even recognized). To address this issue of improved practice, we compiled information on cases of parasitoid or predator introductions over the last 30 years (1985–2015) that spans the historical period when the goal of estimating host ranges for insect control agents was adopted and gradually implemented in the USA. In Table 1, we list 158 parasitoid species introduced during this period (94 in the first decade, 41 in the second and 23 in the third; also included in Table 1 are seven species that were studied but not released). While not a complete list of parasitoids introduced over this 30-year period, Table 1’s entries are, we believe, unbiased with regard to the level of host specificity exhibited by the natural enemies of interest. Entries were drawn from the senior author’s personal files (assembled over the period 1976–2015), reading of additional articles on species mentioned tangentially in the first group of articles, and from the BioCat database records for North America (Canada, Mexico and the USA, including its overseas territories). In Table 2, we list 23 species of predacious arthropods introduced over the same period (1985–2015), compiled in a similar manner as species in Table 1.

For each record, the senior author read the primary literature on the biocontrol agent to understand its likely host range and then used that information to choose a taxonomic rank (order, family, subfamily, tribe, genus or species) most likely to encompass all of the agent’s known hosts or prey. This does not imply that all the members of that taxonomic unit are actual hosts, but only that no smaller unit contains all the known hosts. This classification system should be treated as an index of the host range (based on hosts known from the literature and other available data), rather than a true estimate of the fundamental host range based on quarantine studies, which often were not done.

In most cases in Table 1, the parasitoid’s host range was not known and was not estimated by the researchers before the agent’s introduction. In the first and second decades (with some exceptions in the second decade), agents were introduced if, based on available information, they were considered to be primary parasitoids of the target pest and likely to be efficacious. Host range was generally not estimated, although some information was usually available in the form of records of attacks on other hosts, or from studies designed to investigate if species related to the target pest could be used as alternative hosts. This approach was largely replaced in the third decade by checking the host status of a list of more or less taxonomically (or ecologically) related non-target species, limited to those species that could be obtained for testing. The transition to formal estimation of the fundamental host range [205] of entomophagous natural enemies based on experimentation, as is done for weed biocontrol agents, is gaining momentum in several countries (e.g., New Zealand and the USA). While continued momentum towards an increased requirement for host range and host specificity data are expected, strong differences exist in the biological factors structuring host ranges of parasitoids/predators versus herbivorous insects that may limit progress towards this goal.

In general, analysis of trends found in Table 1 (see Fig. 1) showed a shift in the third decade (2005–2015) towards a preponderance of agents showing an index of genus- (60%) or species-level (8%) specificity (with only 12% being assigned a family-level or above index of specificity) compared with the first and second decades, when 50 and 40% of introductions had family level or above categorizations of specificity and only 21–27 (1985–1994 and 1995–2004, respectively) with genus, or 1–11% (1985–1994 and 1995–2004, respectively) with species-level specificity. In all three decades, 11–12% of introductions could not be classified in this manner due to lack of information.

Concluding Remarks

From our consideration of the above-cited literature, we drew the following points for our final emphasis and recommend they be incorporated in future practice.

Caveats and clarifications

• Evaluate original evidence; do not just repeat past claims. Our views of biocontrol’s non-target impacts such derive from facts not assertions. It is important that new works, either in their introductions to research articles or summaries in review articles should do more than just repeat conclusions of earlier studies. Rather, it is required that authors consider the evidence in past studies and make their own critical judgments on the strength of what is being claimed. Otherwise, errors of either pessimism or optimism cannot be expunged and replaced with data-supported conclusions. A good example of the need for this process is that of *B. remota*

http://www.cabi.org/cabreviews
and whether or not it caused the extinction of either its host (L. iridescens) or a second species (H. dolens) on Fiji, as claimed by Howarth [1], disputed by Kuris [21] and Hoddle [22], but not supported by adequate evidence.

- **Distinguish biological control agents from adventive parasitoids and predators.** Some studies considered in this review did not properly distinguish adventive (self-introduced) parasitoids from ones deliberately introduced as biocontrol agents when commenting on non-target impacts of biocontrol agents. Invasions by polyphagous natural enemies occur naturally or because of commercial movement of plants and other goods. The impacts of accidentally introduced species, even if they are classified as biological control agents elsewhere, are not an indictment of sound biological control practice.

- **Critical need for good taxonomy BEFORE releasing new species.** Another theme found in past projects reviewed here was that many biological control agents turn out to be new species or members of a cryptic complex, hidden by an overly broad name. New species are often described as part of the process of exploration related to actual or potential biological control projects [206]. Such descriptions and clarifications of any cryptic species (that might mask the true identity of potential agents) should be completed before agents are released. See Paterson et al. [207] for an example of the use of molecular methods to identify cryptic species, as further supported by mating studies. Correct appreciation of the taxonomic status of new agents before their introduction should be a clear goal for future work.

- **Prioritize agents and begin by releasing the likely best species first.** In view of past projects acting to the contrary, it is important to emphasize that in future, thoughtful programs should not operate on the principle that release of all species (sometime referred to as the lottery or shotgun approach) found to be primary parasitoids of the target pest is justified (e.g., as against Russian wheat aphid [208–210]) and sweetpotato whitefly strain B [172]). Rapid release of many species with little time to evaluate impacts of any may be a sign of poor conduct, likely to draw criticism [211].

- **Displacement of native species from a non-native host on a crop is not evidence of actual ecological harm.** An important misconception that we noted when reviewing sources on past projects is that past views of displacement of native species by biological control agents need revision. Denying a native species the opportunity to use an exotic pest on an exotic plant (e.g., agricultural crops) by introducing an effective biological control agent of the pest should not be considered as a non-target impact. Rather displacement is an impact only when the native species is displaced from its native host in its usual ecological niche.

### Guides for future thinking

Thinking about non-target impacts associated with past work, there are several points that should be embraced in future work.

- **Distinguish host use from population-level impact**
  Feeding on or parasitizing a native species at levels that do not significantly lower its long-term population density should be considered ‘use’ (A eats B), but not ‘impact’ (A makes B scarcer) and should not be considered as ecologically damaging. While exact numbers are not known, we suggest that, in the absence of data, we should assume that rates of attack <10% are likely to have little impact, while rates of >50%, if widespread and sustained over time, may reduce populations of affected non-target species.

- **Develop country-level online summaries of relevant information**
  Given the difficulty we experienced in assembling information on past releases in the USA, apart from access to BioCat, we recommend that in the future, all countries making releases of biological control agents should develop and maintain web-accessible databases where lists for newly released agents are given, noting year, location and target pest, as well as references or links to sources of information about the estimated host ranges of the agents. Such information, publically available, would provide a firm basis for future analyses of the impacts of biocontrol projects.

- **Plan biological control projects with conservation partners**
  Given the obvious past record of conservation biologists and biological control scientists working separately, at least in the USA, as reflected in the literature we reviewed, it is recommended that in the future, strong efforts be made to reduce that separation. Collaborative studies with conservation biologists are an effective way to build bridges and maximize use of existing expertise. See Van Driesche et al. [212] for discussion of potential joint efforts.

- **Conduct post-release comparisons of actual impact to predicted risk based on quarantine studies**
  Many past projects considered in this review, seemed to end prematurely, perhaps for lack of funding after pest densities declined. This has left a considerable gap in our knowledge. Post-release activities are central to improving our understanding of the level of accuracy of quarantine predictions in forecasting field non-target impacts. Post-release studies, either retroactively for past projects or as part of current programmes, should improve understanding of the ecological consequences of natural enemy releases and model insect life systems to link observable levels of attack to population-level impacts (which is what matters).
### Table 1. Parasitoids introduced between 1985 and 2015, with notes on the level of specificity of each, drawn from literature records (worldwide) and BioCat (for North America, including Canada, Mexico, the USA and US overseas territories, 1985–2004 [end of available records]). Records for North America are relatively complete, but for other regions are partial but not selective. Introductions were only excluded if: (1) species were unidentified (e.g., Aphytis sp.), (2) had also been released in the country before 1985), (3) were duplicative (i.e., release of the same agent in more than one country was not generally tracked unless there were important differences the current authors wished to capture, which was done only in a few cases) or (4) no published information could be located on the release of the species.

<table>
<thead>
<tr>
<th>Year released</th>
<th>Agent</th>
<th>Target</th>
<th>No. of test spp.</th>
<th>Test outcomes</th>
<th>Smallest taxon including all likely hosts</th>
<th>Released Established¹ (year if given equals offirst release)</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>1985–1994</td>
<td></td>
<td></td>
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<tr>
<td>1 1985 USA (Hawaii) from Pakistan</td>
<td><em>Diadegma semiclausum</em> Hellén (Ichneumonidae)</td>
<td><em>Plutella xylostella</em> (L.) <em>Xylostelididae</em></td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Species? No other hosts are reported in the literature and the species show a strong response to cabbage odour, particularly when infested by <em>P. xylostella</em> [214]</td>
<td>R+/E? 1985</td>
<td>[213, 214]</td>
</tr>
<tr>
<td>2 1985 USA</td>
<td><em>Doryctobracon</em> (formerly <em>Opius</em>) <em>trinidadensis</em> Gahan (Braconidae)</td>
<td><em>Anastrepha suspensa</em> (Loew) <em>Tephritidae</em></td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Genus? Known only from original description, which lists two hosts <em>Anastrepha serpentina</em> (Wiedemann) and <em>Anastrepha striata</em> Schiner</td>
<td>R+/E? 1985</td>
<td>[215]</td>
</tr>
<tr>
<td>3 1985 USA (Guam)</td>
<td><em>Ganaspidium utilis</em> Beardsley (now <em>Banacuniculus utilis</em>) [216] (Eucoilidae)</td>
<td><em>Liriomyza trifoli (Burgess)</em> <em>Agromyzidae</em></td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family? All species in <em>Ganaspidium</em> are parasitoids of <em>Agromyzidae</em> (Diptera) [216]</td>
<td>R+/E+ 1985</td>
<td>[216–219]</td>
</tr>
<tr>
<td>4 1985 USA (Texas) from Mexico</td>
<td><em>Mallochia pyralidis</em> Wharton (Ichneumonidae)</td>
<td><em>Eoreuma loftini</em> (Dyar) <em>Crambidae</em></td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Unknown</td>
<td>R+/E? 1985</td>
<td>[220]</td>
</tr>
<tr>
<td>5 1985 USA (Northern Mariana Is.)</td>
<td><em>Pedobiulus foveolatus</em> (Crawford) (Eulophidae)</td>
<td><em>Epilachna philippinensis</em> Dieke (Coccinellidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Subfamily? (Epilachninae) Literature hosts include principally various epilachnine coccinellids, e.g., <em>Epilachna varivestis</em> Mulsant [221] and <em>Hemosepilachna vigintioctopunctata</em> (F.) [222]; unusual records that need confirmation include the skipper <em>Borbo cinnara</em> (Wallace) [223] and a braconid wasp (<em>Cotesia</em> sp.) [224]</td>
<td>R+/E+ 1985</td>
<td>[222–225]</td>
</tr>
<tr>
<td>6 1985–91 Canada from Europe</td>
<td><em>Phygadeuon wiesmanni</em> Sachtleben (Ichneumonidae)</td>
<td><em>Rhagoletis pomonella</em> (Walsh) <em>Tephritidae</em></td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Genus? Literature records include other species of <em>Rhagoletis</em>, such as <em>R. cerasi</em> L. [226], <em>R. alternata</em> Fallén [227]</td>
<td>R+/E– 1985</td>
<td>[226–228]</td>
</tr>
<tr>
<td>7 1985–91 Canada</td>
<td><em>Psyttalia carinata</em> (Thompson) (Opus) <em>rhagoleticola</em> (Sachtleben) (Braconidae)</td>
<td><em>Rhagoletis pomonella</em> (Walsh) and/or <em>Rhagoletis cerasi</em> (Tephritidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Unknown Other species in the genus are parasitoids of tephritids</td>
<td>R+/E– 1985</td>
<td>[228]</td>
</tr>
<tr>
<td>Year released</td>
<td>Agent</td>
<td>Target</td>
<td>No. of test spp.</td>
<td>Test outcomes</td>
<td>Smallest taxon including all likely hosts</td>
<td>Released Established(^1) (year if given equals first release)</td>
<td>References</td>
</tr>
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<tr>
<td>1985–87 USA from Mexico</td>
<td>Trichogramma atopovorilia Oatman &amp; Platner (Trichogrammatidae)</td>
<td>Diatraea grandiosella Dyar (Crambidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Two Families (Noctuidae and Crambidae) Literature records include eggs of noctuids (e.g., Helicoverpa zea [Boddie] [229] and Anticarsia gemmatalis Hübner [230]) and crambids (e.g., Diatraea grandiosella Dyar [231])</td>
<td>R+/E− 1985 [229, 231, 232]</td>
<td></td>
</tr>
<tr>
<td>1985 Israel from USA</td>
<td>Trichogramma platneri Nagarkatti (Trichogrammatidae)</td>
<td>Ascotis selenaria Denis &amp; Schiffmüller (= Boarmia selenaria) (Geometridae) Cryptoblabes gnidiella (Milliere) Pyralidae</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Order or multiple families At the time of introduction, it was known to attack eggs of moths in Tortricidae (Cydia pomonella [L.]; Amorbia cuneana [Wasm.]), Geometridae (Sabulodes aegrotata [Gn.]; Boarmia selenaria Schiff.), and Pyralidae (Cryptoblabes gnidiella Milliere)</td>
<td>R+/E− (Blumberg, per. comm.) Ca 1985 [233–236]</td>
<td></td>
</tr>
<tr>
<td>1985 New Zealand</td>
<td>Aphidius rhopalosiphi De Stephani-Perez (Braconidae)</td>
<td>Metopolophium dirhodum (Walker) Aphidae</td>
<td>4–6?</td>
<td>Several (4–6?) exotic aphids were examined as hosts, but no native species were tested at the time. In addition host records from the literature were considered.</td>
<td>Tribes Aphiidini and Macroshiophini This information was determined ~30 years after the introduction by Cameron et al., [239]</td>
<td>R+/E+ 1985 [237–239]; David Teulon, per. comm.</td>
<td></td>
</tr>
<tr>
<td>1985–1987 USA (Northern TX, from Mexico)</td>
<td>Trichogramma atopovorilia Oatman and Platner (Trichogrammatidae)</td>
<td>Diatraea grandiosella Dyar (Crambidae)</td>
<td>0</td>
<td>Laboratory host testing was done to determine if certain pest borers were susceptible, including various crambids (formerly part of Pyralidae): Diatraea considerata Heinrich, D. saccharalis (F.), D. grandiosella Dyar and Eoreuma loftini (Dyar)</td>
<td>Unknown</td>
<td>R+/E− [232, 245]</td>
<td></td>
</tr>
<tr>
<td>1985–1987 USA (northern Texas, from Mexico)</td>
<td>Allorhogas pyralophagus Marsh (Braconidae)</td>
<td>Diatraea grandiosella Dyar (Crambidae)</td>
<td>0</td>
<td>Laboratory host testing was done to determine if certain pest borers were susceptible</td>
<td>Two Families Grass stem borers in Crambidae and Noctuidae; known hosts include Diatraea saccharalis, Emmalocera depressella (Swinhoe), several species of Chilo, and the noctuid Acigona steniellus (Hampson), among others</td>
<td>R+/E− 1985–1987 [232, 246–250]</td>
<td></td>
</tr>
<tr>
<td>No.</td>
<td>Year</td>
<td>Location</td>
<td>Host</td>
<td>Family</td>
<td>Known Hosts</td>
<td>Notes</td>
<td></td>
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</tr>
<tr>
<td>14</td>
<td>1985–1987 USA (northern Texas, from Mexico)</td>
<td><em>Macrocentrus prolificus</em> Wharton (Braconidae)</td>
<td><em>Diatraea grandidiella</em> Dyar (Crambidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Known hosts include several species of <em>Diatraea</em> (D. considerata Heinrich, D. grandidiella Dyar, D. saccharalis [F.])</td>
<td></td>
</tr>
<tr>
<td>15</td>
<td>1985–1987 USA (Northern Texas, from Mexico)</td>
<td><em>Digonogastra kimballi</em> Kirkland (Braconidae)</td>
<td><em>Diatraea grandidiella</em> Dyar (Crambidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Two Families Known hosts are <em>Eoreuma loftini</em> and 5 species of <em>Diatraea</em> (all Crambidae, Pyralidae)</td>
<td></td>
</tr>
<tr>
<td>16</td>
<td>1985–1987 USA (Northern Texas, from Mexico)</td>
<td><em>Pedobius furvus</em> Gahan (Eulophidae)</td>
<td><em>Diatraea grandidiella</em> Dyar (Crambidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Two Families Grass stem borers in Pyralidae and Noctuidae</td>
<td></td>
</tr>
<tr>
<td>17</td>
<td>1986–89 USA (Texas and Florida)</td>
<td><em>Aphytis yanonensis</em> DeBach &amp; Rosen (Aphelinidae)</td>
<td><em>Parlatoria pergandi</em> Comstock (Texas) <em>Unaspis citri</em> (Florida) Diaspididae</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Literature records include diaspidid scales such as <em>Unaspis yanonensis</em> Kuwana [256]</td>
<td></td>
</tr>
<tr>
<td>18</td>
<td>1986–87 Canada</td>
<td><em>Euthrix</em> (formerly <em>Ernestia</em>) consobrina (Meigen) Tachinidae</td>
<td><em>Mamestra configurata</em> Walker (Noctuidae)</td>
<td>5</td>
<td>Of five NT noctuid species tested by placing a fly maggot on the test larva, four supported maggot development to pupation [259]</td>
<td>Subfamily (Hadenine)</td>
<td></td>
</tr>
<tr>
<td>19</td>
<td>1986–90 USA</td>
<td><em>Peristenus conradi</em> Marsh (Braconidae)</td>
<td><em>Adelphocoris lineatus</em> (Goeze) Miridae</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Species? Post-release surveys in two US states found parasitism in only the target mirid, of 7 species surveyed</td>
<td></td>
</tr>
<tr>
<td>20</td>
<td>1986 USA (Hawaii)</td>
<td><em>Tetrastichus brontispae</em> Ferrière (Eulophidae)</td>
<td><em>Brontispa chalybeipennis</em> (Zacher) Chrysomelidae</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family? (coconut-feeding chrysomelid beetles) Field host records include <em>Brontispa froggatti</em> [Brontispa longissima] var. selebensis Gestro [262], <em>Brontispa mariana</em> Spaeth [263], <em>Gastronella centralinea</em> (Fairm.) and <em>G. lugubris</em> (Fairm.) [264], <em>Octodonta nipae</em> (Maulik) (Chrysomelidae) [265]</td>
<td></td>
</tr>
<tr>
<td>21</td>
<td>1986–89 USA</td>
<td><em>Thripobius javae</em> (Girault) (= <em>T. semiluteus</em> Boucek) Eulophidae</td>
<td><em>Heliothrips haemorrhoidalis</em> (Bouché) Thripidae</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Subfamily (Panchaetothripinae) This estimation was made by Froud et al., [266] based on literature records</td>
<td></td>
</tr>
<tr>
<td>22</td>
<td>1986 USA (Guam)</td>
<td><em>Trichogramma platneri</em> Nagarkatti (Trichogrammatidae)</td>
<td><em>Penicillaria jocosatrix</em> Guenée (Noctuidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Order or multiple families At the time of introduction, it was know to attack eggs of moths in Tortricidae (<em>Cydia pomonella</em> [L.]; <em>Amorbia cuneana</em> [Wism.]), Geometridae (<em>Sabulodes aegrotata</em> [Gn.; <em>Boarmia selenaria</em> Schiff.] [233], and Pyralidae (<em>Cryptoblabes gnidiella</em>) [270]</td>
<td></td>
</tr>
<tr>
<td>23</td>
<td>1986–1987 Guam (from India)</td>
<td><em>Aleioides nr. circumspectus</em> (Nees) Braconidae</td>
<td><em>Penicillaria jocosatrix</em> Guenée (Noctuidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Unknown</td>
<td></td>
</tr>
</tbody>
</table>

Source: [http://www.cABI.org/cabreviews](http://www.cABI.org/cabreviews)
Table 1. (Continued)

<table>
<thead>
<tr>
<th>Year released</th>
<th>Agent</th>
<th>Target</th>
<th>No. of test spp.</th>
<th>Test outcomes</th>
<th>Smallest taxon including all likely hosts</th>
<th>Released Established¹ (year if given equals first release)</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>25 1986–1987 Guam (from India)</td>
<td>Euplectrus nr. parulus Ferriere (Eulophidae)</td>
<td>Penicillaria jocosatrix Guenée (Noctuidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Unknown</td>
<td>R+/E+ 1986–1987 [270]</td>
<td></td>
</tr>
<tr>
<td>26 1986 Turkey (from California)</td>
<td>Eretrnocerus debachi Rose and Rosen (Apheilidae)</td>
<td>Parabemisia myricae (Kuwana) Aleyrodidae</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Species? Known only from target host</td>
<td>R+/E+ 1986 [273]</td>
<td></td>
</tr>
<tr>
<td>27 1987 Cyprus (from France)</td>
<td>Metaphycus bartletti (Annecke and Myrhardt) Encyrtidae</td>
<td>Saissetia oleae (Olivier) Coccidae</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Genus (Saissetia and closely related Coccidae)</td>
<td>R+/E+ 1988–89 [274–276]</td>
<td></td>
</tr>
<tr>
<td>28 1987 Togo and later other parts of West Africa (from India by CABI)</td>
<td>Gyransoida tebygii Noyes (Encyrtidae)</td>
<td>Rastroccoccus invadens Williams (Pseudococcidae)</td>
<td>3</td>
<td>One species each of Pseudococcus, Planococcus, and Phenacoccus were tested but none were parasitized</td>
<td>Species Did not parasitize either R. iceryoides (Green) or R. mangiferae (Green) in laboratory tests [277]</td>
<td>R+/E+ 1988 [277–279]</td>
<td></td>
</tr>
<tr>
<td>29 1991 West Africa (from India by CABI)</td>
<td>Anagyrus mangicola Noyes (Encyrtidae)</td>
<td>Rastroccoccus invadens Williams (Pseudococcidae)</td>
<td>3</td>
<td>One species each of Pseudococcus, Planococcus, and Phenacoccus were tested but none were parasitized</td>
<td>Species Did not parasitize either R. iceryoides (Green) or R. mangiferae (Green) in laboratory tests [277]</td>
<td>R+/E+ 1991 [277, 279]</td>
<td></td>
</tr>
<tr>
<td>30 1987 USA (Hawaii)</td>
<td>Psyllaephagus yaseeni Noyes (Encyrtidae)</td>
<td>Heteropsylla cubana Crawford (Psyllidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family (Psyllidae) No other hosts are recorded but there are few studies on this species</td>
<td>R+/E+ [280, 281]</td>
<td></td>
</tr>
<tr>
<td>31 1987–89 USA</td>
<td>Trissolcus basalis (Wollaston) (formerly Microphanurus basalis and Asolcus basalis) Scelionidae</td>
<td>Nezara viridula L. (Pentatomidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family (Pentatomidae) Hosts recorded in the literature are various pentatomids: Euschistus servus Say and Eubrychus floridus Say among others</td>
<td>R+/E+ [282–285]</td>
<td></td>
</tr>
<tr>
<td>32 1987 Hawaii (from Tobago)</td>
<td>Psyllaephagus rotundifolius (Howard) (first identified as Psyllaephagus sp. near rotundifolius) (Howard) Encyrtidae</td>
<td>Heteropsylla cubana Crawford (Psyllidae)</td>
<td>2</td>
<td>Two other NT species in the same genus were accepted as hosts [286]</td>
<td>Genus? Recorded hosts include target plus 2 other species in same genus: H. huasachae Caldwell and H. fuscus Crawford) [286]</td>
<td>R+/E? 1988 [286, 287]</td>
<td></td>
</tr>
<tr>
<td>33 1988–89 USA (Guam)</td>
<td>Adelencyrtus oceanicus Doubt (Encyrtidae)</td>
<td>Funchapsis oceanica (Lindinger) Diaspididae</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Unknown No other host records in literature</td>
<td>R+/E+ [288, 289]</td>
<td></td>
</tr>
</tbody>
</table>
34 1988 USA  
Ageniaspis (formerly Holcothorax) testaceipes (Ratzeburg) Encyrtidae  
Phyllonorycter (formerly Lithocolletis) craetaegella (Clemens) Gracillariidae  
No laboratory host range estimation done  
Genus? Limited to ecological niche of leafminers on deciduous trees? Most literature records are gracillariid (Lep.) leafminers in the genus Phyllonorycter including P. blancardella (F.) [290], P. ringoniella (Matsumura) [291], and Phyllonorycter pyriferella (Gerasimov) [292]. However one record is of the gelichiid (Lep.) leafminer Recurvaria syrictis Meyrick [293] (needs verification)  
R+/E+ (in Connecticut [294]) [290–294]

35 1988–89 USA  
Aphidius rhopalosiphi De Stefani-Perez (Braconidae)  
Diuraphis noxia (Kurdjumov) Aphidae  
No laboratory host range estimation done  
Tribes Aphidini and Macrophialini  
Biotypes of this species may exist that have different host ranges [295]  
R+/E– [208, 239, 295]

36 1988 USA Florida, from Bolivia  
Larra godmani Cameron (Sphecidae)  
Scapteriscus spp. (now Neoscapteteriscus), This was an accidental contaminant in a shipment of Larra bicolor from Bolivia [296] (Gryllotalpidae)  
No laboratory host range estimation done  
Genus (Neoscapteteriscus)  
+/–? 1988 [296, 297]

37 1988 USA Florida, from Bolivia  
Larra bicolor F. (Sphecidae)  
Scapteriscus vicinus Scudder (now Neoscapteteriscus) Gryllotalpidae  
No successful attack observed on one native NT species, Neocurtilla hexadactyla (Perty), the only native cricket in the family in the region [296]  
Genus (Neoscapteteriscus)  
This parasitoid is functionally specific to the genus level in the US because it does not successfully attack the only native mole cricket in the region, Neocurtilla hexadactyla (Perty) [296]  
+/+ 1988 [296, 298]

38 1988–89 USA  
Telenomus remus Nixon (Scelionidae)  
Spodoptera frugiperda (J. E. Smith) Noctuidae  
Of 39 NT spp tested, of those not in the Noctuidae, there were seven Arctiidae, five Geometridae, one Mimallonidae, two Notodontidae, and two Pyralidae. Of these only one pyralid was parasitized. Of the 21 noctuids tested, 11 spp. in 11 genera were parasitized  
Family (Noctuidae)  
From Spodoptera mauritia Boisd., in Malaya [299], Prodenia (Spodoptera?) litura F. [300], and 11 other noctuids and 1 pyralid [301]  
R+/E– [299–301]

39 1988 Israel (from USA-CA; from Australia originally)  
Cryptochaetum iceryae (Williston) Cryptochaetidae  
Icerya purchasi Maskell (Monophlebidae)  
No laboratory host range estimation done  
Family (Monophlebidae)  
(or perhaps just the genus Icerya or even only the target pest)  
R+/E+ 1988 [302]

40 Ca. 1986  
Roptrocerus xylophagorum (Ratzeburg) Pteromalidae  
Ips grandicollis (Eichh.) Curculionidae, Scolytinae  
No laboratory host range estimation done  
Subfamily (Scolytinae)  
Attacks species of Ips [303], Dendroctonus [304], Hylurgops [305], Pityogenes [306]  
R+/E+ ca 1986 [303–308]
<table>
<thead>
<tr>
<th>Year released</th>
<th>Agent (Continued)</th>
<th>Target</th>
<th>No. of test spp.</th>
<th>Test outcomes</th>
<th>Smallest taxon including all likely hosts</th>
<th>Released Established¹ (year if given equals first release)</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>41</td>
<td>1988 USA Florida</td>
<td>Ormia (Euphasiopteryx) depleta (Wiedemann)</td>
<td>Scapteriscus vicinus (Scudder) (Neoscaptheviriscus) Grylotalpidae</td>
<td>5</td>
<td>Ormia depleta was attracted to three of five Scapteriscus species tested, two of which were programme targets and one a non-target invasive [309]</td>
<td>Genus That North American Anurogrillus species would not be attacked was determined [310] based on song characteristics, which determine parasitoid attraction</td>
<td>R+/E+ 1988 [298, 309–313]</td>
</tr>
<tr>
<td>42</td>
<td>1988–91 USA (from both France and Korea)</td>
<td>Ageniaspis fuscololis (Dalman) Encyrtidae</td>
<td>Yponomeuta malinellus (Zeller) Yponomeutidae</td>
<td>0</td>
<td>No laboratory-based host range testing prior to release</td>
<td>Genus? At least five species in the target species’ genus known to be parasitized in the field [314]</td>
<td>R+/E+ 1988 [314–316]</td>
</tr>
<tr>
<td>43</td>
<td>1989–90 USA</td>
<td>Binodoxys (formerly in Trioxys) brevicornis (Haliday)</td>
<td>Brachycorynella asparagi (Mordv.)</td>
<td>0</td>
<td>No laboratory-based host range testing prior to release</td>
<td>Family? A polyphagus aphid parasitoid known from at least three genera of aphids in addition to that of the target (Binodoxys): namely Myzus cerasi (F.) [317], Cavariella sp. [318], and Hyadaphis coriandri (Das) [319]</td>
<td>R+/E+ [317, 318, 320, 321]</td>
</tr>
<tr>
<td>44</td>
<td>1989–91 USA</td>
<td>Eurytheae scutellaris (Robineau-Desvoidy) Braconidae, Aphidiinae</td>
<td>Yponomeuta malinellus (Zeller) Yponomeutidae</td>
<td>0</td>
<td>No laboratory-based host range testing prior to release</td>
<td>Family? Hosts recorded in the literature include species in three families: the pyralid Acrobasis consociella (Hubner) [322], various yponomeutids including Yponomeuta padellus (L.) [323], and the geometrid Abraxas pantaria (L.) [324]</td>
<td>R+/E− [316, 322, 324]</td>
</tr>
<tr>
<td>45</td>
<td>1989 New Zealand (from Argentina)</td>
<td>Microctonus hyperoeae Loan Braconidae, Euphorinae Sphecophaga vespurum (Curtis) Ichneumonidae</td>
<td>Listronotus bonariensis (Kuschel) Curculionidae Vestpula germanica (F.) and Vespula vulgaris (L.) Vespidae Yponomeuta malinellus (Zeller) Yponomeutidae</td>
<td>24</td>
<td>Of 24 NT weevils tested, one NT supported complete development</td>
<td>Tribe? or Subfamily? The one NT that supported development was in new genus</td>
<td>R+/E+ 1992 [325, 326]</td>
</tr>
<tr>
<td>46</td>
<td>1989 Australia (from Europe via New Zealand)</td>
<td>Diadegma (formerly Aphytis) armillata (also armillatum) (Gravenhorst) (perhaps now in Angelita) Ichneumonidae</td>
<td>Listronotus bonariensis (Kuschel) Curculionidae Vestpula germanica (F.) and Vespula vulgaris (L.) Vespidae Yponomeuta malinellus (Zeller) Yponomeutidae</td>
<td>8</td>
<td>No NT species tested were regularly attacked (three instances observed)</td>
<td>Subfamily Hosts appears to be restricted to the Vespinae</td>
<td>R+/E− 1989 [327]</td>
</tr>
<tr>
<td>47</td>
<td>1989–1991 USA (from both France and Korea)</td>
<td>Herpestomus brunnicornis (Gravenhorst) Ichneumonidae Telenomus luculus (Nixon) Scelionidae</td>
<td>Yponomeuta malinellus (Zeller) Yponomeutidae</td>
<td>3</td>
<td>Three NT hosts in the same genus in the native range were suitable hosts</td>
<td>Tribe? or Subfamily? Known from species in two genera of Yponomeutidae, including Yponomeuta rorubius (Hb.) [328], Y. evonymella L. [329], and Prays oleae (Bernard) [330]</td>
<td>R+/E− [315, 316, 326, 329, 331] (see also Wagener et al., 2006 [332] for notes on phylogeny of parasitoid genus); Agro et al., 2009 [330]</td>
</tr>
<tr>
<td>48</td>
<td>1989–1996 Samoa, Tonga, Fiji, and the Cook Islands</td>
<td>Telenomus luculus (Nixon) Scelionidae</td>
<td>Yponomeuta malinellus (Zeller) Yponomeutidae</td>
<td>11</td>
<td>All three NT in same genus were attacked; zero of eight NT noctuids in other genera were attacked</td>
<td>Genus? Considered adequate for island fauna.</td>
<td>R+/E+ 1989–1996 [334]</td>
</tr>
<tr>
<td>Year</td>
<td>Origin</td>
<td>Parasitoid</td>
<td>Host</td>
<td>Host Range</td>
<td>Notes</td>
<td></td>
<td></td>
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</tr>
<tr>
<td>1989–1990 USA</td>
<td>Encarsia inaron (Walker)</td>
<td>Ash whitefly, Siphoninus phillyreae (Haliday)</td>
<td>0</td>
<td>No formal host range testing was done before release. The parasitoid is known to attack species of whiteflies in several genera, including Siphoninus, Bemisia, Trialeurodes and Pealius</td>
<td>Family Medium? Not reported from non-pest, native U.S. whiteflies.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1990 Canada</td>
<td>Apanteles murinanae</td>
<td>Choristoneura fumiferana (Clemens)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family? The only other known field host is Eucosma nigricans (H.-S.), another tortricid of similar biology as C. murinana, with which it shares a common host and habitat [338]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1990–94 USA, Florida, from Hong Kong</td>
<td>Eretmocerus rul</td>
<td>Bemisia tabaci (Gennadius) strain B</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family? No other host records from field or laboratory studies were located</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1990–1995 USA from China</td>
<td>Cocobius nr. fulvus</td>
<td>Unaspis euonymi (Comstock)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Genus? C. fulvus is known only from two species Unaspis scales [341, 342], assuming that C. fulvus and C. nr fulvus are the same, which was never determined</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1990–1995 USA from China</td>
<td>Encarsia (formerly Phloea) nr. diaspidicola Silvestri</td>
<td>Unaspis euonymi (Comstock)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family? Assuming this is the same as E. diaspidicola and that it is not a species complex, then several diaspidid scales (including Pseudaulacaspis pentagona [Targioni-Tozzetti] and Quadraspidiotus perniciosus [Comstock]) are known hosts, but others are not hosts [345]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Year released</td>
<td>Agent</td>
<td>Target</td>
<td>No. of test spp.</td>
<td>Test outcomes</td>
<td>Smallest taxon including all likely hosts</td>
<td>Released Established (^1) (year if given equals offirst release)</td>
<td>References</td>
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<tr>
<td>58 1990–1995 (USA from China)</td>
<td><em>Aphytis proclia</em> (Walker) Aphelinidae</td>
<td><em>Unaspis eucnemis</em> (Comstock) Diaspididae</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family? Several diaspidid scales (including <em>Pseudaulacaspis pentagona</em> [Targioni-Tozzetti] and <em>Quadraspidiotus perniciosus</em> [Comstock] among others) are known hosts. Note, however, that <em>A. proclia</em> as a name may refer to more than one species</td>
<td>R+/E- 1990–1995 [342, 343, 347–349]</td>
<td></td>
</tr>
<tr>
<td>59 1992 USA Colorado, Washington and others, from Morroco and the Middle East</td>
<td><em>Ephedrus plagiator</em> (Nees) Braconidae, Aphidinae</td>
<td><em>Diuraphis noxia</em> (Kurdjumov) Aphidae</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family? Several aphids are recorded as hosts, including <em>Aphis pomi</em> de Geer [350], <em>Schizaphis graminum</em> (Rondani), <em>Rhopalosiphum padi</em> (L.), <em>Macrosiphum</em> (Sitobion) <em>avenae</em> (F.) [351], <em>Aulacorthum solani</em> (Kaltenbach) [352], among others</td>
<td>R+/E- [208, 351–353]</td>
<td></td>
</tr>
<tr>
<td>60 1992 USA</td>
<td><em>Aphelinus atriplicis</em> Kurdjumov (Aphelinidae)</td>
<td><em>Diuraphis noxia</em> (Kurdjumov) Aphidae</td>
<td>0</td>
<td>Limited to Aphididae; species identity misunderstood initially but sorted out after introduction</td>
<td>Family Safe to non-aphids; parasitizes many NT aphids; but population impacts are unknown</td>
<td>R+/E+ 1992 [208–210, 354–357]</td>
<td></td>
</tr>
<tr>
<td>61 About 1990 Mexico (from Africa)</td>
<td><em>Prorops nasuta</em> Waterson (Bethylidae)</td>
<td><em>Hypothenemus hampei</em> (Ferrari) Curculionidae, Scolytinae</td>
<td>2</td>
<td>Both NT species were used successfully as hosts</td>
<td>Family? Several genera of weevils are known to support oviposition and development. Boundaries of host range are not known</td>
<td>R+/E+ 1988 [258, 359]</td>
<td></td>
</tr>
<tr>
<td>62 1988 Mexico (from Africa)</td>
<td><em>Cephalonomia stephanoderis</em> Betrem (Bethylidae)</td>
<td><em>Hypothenemus hampei</em> (Ferrari) Curculionidae, Scolytinae</td>
<td>2</td>
<td>Both NT species were used successfully as hosts</td>
<td>Family? Several genera of weevils are known to support oviposition and development. Boundaries of host range are not known</td>
<td>R+/E+ 1989 [358, 359]</td>
<td></td>
</tr>
<tr>
<td>63 Ca. 1990 Guatemala (from Africa)</td>
<td><em>Phymachthus coffea</em> (LaSaile) Eulophidae</td>
<td><em>Hypothenemus hampei</em> (Ferrari) Curculionidae, Scolytinae</td>
<td>5</td>
<td>Two NT <em>Hypothenemus</em> sp. were attacked; one <em>Hypothenemus</em> sp. was not and two species in other bark beetle genera were not</td>
<td>Genus?</td>
<td>R+/E? 1990? [360]</td>
<td></td>
</tr>
<tr>
<td>64 1991 Canada (from Europe)</td>
<td><em>Aphantorhaphopsis</em> (Ceranthia) <em>samarensis</em> (Vileneuve) Tachinidae</td>
<td><em>Lymantria dispar</em> (L.) Erebidae, Lymantriinae</td>
<td>11</td>
<td>Of the 11 North America species (in five families) only one species, <em>Orgyia leucostigma</em> (J. E. Smith) (Lymantriinae) was a suitable host</td>
<td>Subfamily (Lymantriinae)</td>
<td>R+/E? 1991 [30, 361, 362]</td>
<td></td>
</tr>
<tr>
<td>65 1992 USA</td>
<td><em>Aphelinus nr asychis</em> (Aphelinidae)</td>
<td><em>Diuraphis noxia</em> (Kurdjumov) Aphidae</td>
<td>0</td>
<td>Limited to Aphididae; species identity misunderstood initially but sorted out after introduction</td>
<td>Family Safe to non-aphids; certainty of 'use' of many NT aphids; uncertainty about population impacts</td>
<td>R+/E+ 1992 [208–210, 354–357]</td>
<td></td>
</tr>
<tr>
<td>Year</td>
<td>Location</td>
<td>Genus/Moniker</td>
<td>Family</td>
<td>Notes</td>
<td></td>
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</tr>
<tr>
<td>1992</td>
<td>Australia</td>
<td>Citrostichus phyllocnistoides (Naryanin)</td>
<td>Eulophidae</td>
<td>Genus? relative to Australia biota: R+/E− 1992</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1992</td>
<td>Australia</td>
<td>Ageniaspis citricola (Longvinovskaya)</td>
<td>Encyrtidae</td>
<td>Genus? relative to Australia biota: R+/E+ 1992</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1992</td>
<td>Australia</td>
<td>Cirrospilus ingenuus (=quadristriatus)</td>
<td>Eulophidae</td>
<td>Genus? relative to Australia biota: R+/E+ 1992</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1992</td>
<td>USA</td>
<td>Neodryinus typhlocybae (Ashmead)</td>
<td>Dryinidae</td>
<td>Family? (Dryinidae): R+/E+ [364, 365]; Two Families: Attacks various fruit-boring and cocoon-forming tortricids such as Cydia molesta (Busck) and Grapholitha funebrana (Treitschke) (Mills, pers. comm.)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1993</td>
<td>USA, California and Washington (from Kazakhstan)</td>
<td>Lytopylus rufipes (Nees von Eisenbeck)</td>
<td>Phryganaeidae</td>
<td>No formal host range testing done</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1993</td>
<td>USA (CA) from Australia</td>
<td>Avetianella longoi Siscaro</td>
<td>Mymaridae</td>
<td>Unknown: Likely restricted to hosts on eucalpts due to attraction to host plant odours</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1993</td>
<td>USA (CA) from Australia</td>
<td>Jarra phoracantha Austin, Quicke, and Marsh</td>
<td>Braconidae</td>
<td>Unknown: Likely restricted to hosts on eucalpts due to attraction to host plant odours</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1993</td>
<td>Kenya (from Pakistan)</td>
<td>Cotesia flavipes Cameron</td>
<td>Braconidae</td>
<td>Unknown: Likely restricted to hosts on eucalpts due to attraction to host plant odours</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Notes:
- **R+/E−** indicates a species that is known to attack hosts in the same genus but not in the same family.
- **R+/E+** indicates a species that is known to attack hosts in the same genus and family.
- **R+/E?** indicates a species that has been tested in the field but not in the laboratory.

<table>
<thead>
<tr>
<th>No.</th>
<th>Year released</th>
<th>Agent</th>
<th>Target</th>
<th>No. of test spp.</th>
<th>Test outcomes</th>
<th>Smallest taxon including all likely hosts</th>
<th>Released Established^1 (year if given equals of first release)</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>77</td>
<td>1993 California (from Australia); 1994 Britain; 1997 France and Ireland; 2001 Chile</td>
<td>Psyllaephagus pilosus Noyes (Encyrtidae)</td>
<td>Ctenarytaina eucalypti (Maskell) Psyllidae</td>
<td>0</td>
<td>No formal host range testing done but this is likely a eucalyptus specialist [376], conferring high host specificity in the invaded range via the influence of the host plant</td>
<td>Family High specificity in invaded ranges due to attraction to Eucalyptus</td>
<td>R+/E+ 1993–1997</td>
<td>[376–379]</td>
</tr>
<tr>
<td>78</td>
<td>1993–1999, USA (from United Arab Emirates)</td>
<td>Eretmocerus emiratus Zolnerowich &amp; Rose</td>
<td>Bemisia tabaci (Gennadius) strain B (Aleyrodidae)</td>
<td>0</td>
<td>No laboratory host range estimation done Plans to test Trialeurodes vaporariorum (Westwood), T. abutilonia (Haldeman), and Bemisia berbericola (Cockerell) were made but not carried out</td>
<td>Family (Aleyrodidae) No field hosts recorded other than Bemisia</td>
<td>R+/E+ (in southern CA) &lt;2000</td>
<td>[172, 380–383]</td>
</tr>
<tr>
<td>80</td>
<td>1993–1999, USA (from Pakistan)</td>
<td>Eretmocerus hayati Zolnerowich &amp; Rose (Aphelinidae)</td>
<td>Bemisia tabaci (Gennadius) strain B (Aleyrodidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family (Aleyrodidae)</td>
<td>R+/E+ (in Texas and Mexico) &lt;2000</td>
<td>[172, 380, 382, 383]</td>
</tr>
<tr>
<td>81</td>
<td>1993–1999, USA (from Thailand, Taiwan)</td>
<td>Eretmocerus melanoscutus Zolnerowich &amp; Rose (Aphelinidae)</td>
<td>Bemisia tabaci (Gennadius) strain B (Aleyrodidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family (Aleyrodidae) No field hosts recorded other than Bemisia</td>
<td>R+/E+ (in Florida) &lt;2000</td>
<td>[172, 380, 382, 383]</td>
</tr>
<tr>
<td>82</td>
<td>1993–1999, USA (from Taiwan)</td>
<td>Eretmocerus nr. furuhashii Rose &amp; Zolnerowich (Aphelinidae)</td>
<td>Bemisia tabaci (Gennadius) strain B (Aleyrodidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family (Aleyrodidae) No field hosts recorded other than Bemisia</td>
<td>R+/E− &lt;2000</td>
<td>[172, 382, 383]</td>
</tr>
<tr>
<td>83</td>
<td>1993–1999, USA (from Spain, Israel)</td>
<td>Eretmocerus mundus Mercet (Aphelinidae)</td>
<td>Bemisia tabaci (Gennadius) strain B (Aleyrodidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family (Aleyrodidae) Field surveys in CA found E. mundus only from B. tabaci, not non-target whiteflies [174]; In the laboratory two species of Trialeurodes were attacked (B. abutilonia [Haldeman] and T. vaporariorum [Westwood]) [385]</td>
<td>R+/E+ (in California) &lt;2000</td>
<td>[170, 172, 381–383, 385]</td>
</tr>
<tr>
<td>84</td>
<td>1993–1999, USA (from Puerto Rico)</td>
<td>Amitus bennetti Viggiani &amp; Evans (Platygasteridae)</td>
<td>Bemisia tabaci (Gennadius) strain B (Aleyrodidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family (Aleyrodidae) No field hosts recorded other than Bemisia</td>
<td>R+/E? &lt;2000</td>
<td>[382]</td>
</tr>
<tr>
<td>85</td>
<td>1993–1999, USA (from Israel)</td>
<td>Encarsia lutea (Masi) Aphelinidae</td>
<td>Bemisia tabaci (Gennadius) strain B (Aleyrodidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family (Aleyrodidae) Males: Lepidoptera Known to attack several genera of whiteflies, including Trialeurodes abutiloneæa (Hald.) and T. vaporariorum (West.) and for males to emerge from eggs of noctuid moths [386], and Acaudaleyrodes citri (Priesn. &amp; Hosni) [387], Parabemisia myricae (Kuwana) [388], Aleurotobus spp. [389]</td>
<td>R+/E− &lt;2000</td>
<td>[382, 383, 386–389]</td>
</tr>
<tr>
<td>Year</td>
<td>Country</td>
<td>Species</td>
<td>Host</td>
<td>Status</td>
<td>Notes</td>
<td></td>
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<tr>
<td>1993–1999, USA</td>
<td>Encarsia sophia (=E. transvena) (Girault &amp; Dodd)</td>
<td>Bemisia tabaci (Gennadius) strain B (Aleyrodidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family (Aleyrodidae) Known to attack T. vaporariorum (Westw.) [390] and Bemisia tabaci B (Gennadius) Bondar [391]</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1993–1999, USA</td>
<td>Encarsia nr. pergandiella Howard (Aphelinidae)</td>
<td>Bemisia tabaci (Gennadius) strain B (Aleyrodidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family (Aleyrodidae) Females: Aleyrodidae or below Males: Aphelinid parasitoids Males develop as hyperparasitoids on whitefly parasitoids, including E. mundus [394]</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1993–1999, USA</td>
<td>Encarsia nr. hisipida De Santis (Aphelinidae)</td>
<td>Bemisia tabaci (Gennadius) strain B (Aleyrodidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family (Aleyrodidae) Known to attack T. vaporariorum (Westw.), Trialeurodes variabilis (Quaintance) [396], Paraleurodes minei Iaccarino [397], and Aleurotrachelus socialis Bondar [391]</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1993–2000 USA</td>
<td>Mastrus ridens Horstmann (formerly M. ridibundus) Ichneumonidae</td>
<td>Cydia pomonella L. (Tortricidae)</td>
<td>0</td>
<td>No laboratory host range estimation done at the time of the introduction to USA. Later, in New Zealand [398] some post-facto host range testing showed that of five species tested, 1 NT Cydia species and four others in the target’s family (Tortricidae) were attacked, but offspring were small and mostly male; known in native range only from target, but little sampling other than of the target Genus? Other tortricids could be killed by this parasitoid but seem unlikely themselves to support M. ridens populations due to a too rapid death from the paralysing venom of the parasitoid</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1994 USA California</td>
<td>Anaphes nitens (Girault) (other generic placements include Patasson and Anaphoidea) Mymaridae</td>
<td>Gonipterus scutellatus Gyllenhal See Mapondera et al. [399] for notes on cryptic species in genus. Curculionidae Gonipterus platensis (Marelli) (see Mapondera et al., [399] for notes on cryptic species in genus). (Curculionidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Genus? Field records include Gonipterus gibberus Boisdusval [400]</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1994 Spain (from South Africa??)</td>
<td>Anaphes nitens (Girault) (other generic placements include Patasson and Anaphoidea) (Mymaridae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Genus? Field records include Gonipterus gibberus Boisdusval [400]</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1994 Brazil (Bahia) (from Colombia or Venezuela)</td>
<td>Apoanagyrus diversicornis (Howard) Encyrtidae</td>
<td>Phenacoccus herrenii Cox and Williams (Pseudococcidae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Genus? Known only from species of Phenacoccus Species? No attack in laboratory tests and no other known field hosts</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
| 1994 Brazil (Bahia) (from Colombia or Venezuela) | Aenasius vexans (Kerrich) Encyrtidae | Phenacoccus herrenii Cox and Williams (Pseudococcidae) | 7 (6 years post-release) | No laboratory host range estimation done before release in Brazil, but subsequent studies of sene mealybug species found that A. vexans did not attack any of the non-target mealybugs tested | R+E+ 1994–1995 [403–405]
<table>
<thead>
<tr>
<th>Year released</th>
<th>Agent</th>
<th>Target</th>
<th>No. of test spp.</th>
<th>Test outcomes</th>
<th>Smallest taxon including all likely hosts</th>
<th>Released Established¹ (year if given equals first release)</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>1994 Brazil (Bahia) (from Colombia or Venezuela)</td>
<td>Acerophagus coccois Smith (Encyrtidae)</td>
<td>Phenacoccus herreni Cox and Williams (Pseudococcidae)</td>
<td>7 (six years post-release)</td>
<td>No laboratory host range estimation done before release in Brazil, but subsequent studies of seven mealybug species found that A. coccois attacked three (two species of Phenacoccus and Ferrisia virgata [Cockerell])</td>
<td>Family (Pseudococcidae) Known from Oncella acute (Lodbell) [407, 408]; Ferrisia virgata (Cockerell) [406] and several species of Phenacoccus</td>
<td>R+/E+ 1994–1995 [403–406, 408]</td>
<td></td>
</tr>
<tr>
<td>1994 USA/Florida (via Australia from Thailand)</td>
<td>Ageniaspis citricola Longinovskaya (Encyrtidae)</td>
<td>Phyllocnistis citrella Stanton (Gracillariidae)</td>
<td>0</td>
<td>No laboratory host range estimation done relative to US Phyllocnistis species; specificity assumed based on testing in Australia</td>
<td>Unknown relative to North American fauna</td>
<td>R+E+ 1994 [409–411]</td>
<td></td>
</tr>
<tr>
<td>Family (Gracillariidae)</td>
<td>Includes target, acceptable hosts were found in four genera o pentatomids</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1995 USA/Florida (via Australia from Thailand)</td>
<td>Cirrospilus ingenuus (quadristriatus) Gahan (Eulophidae)</td>
<td>Phyllocnistis citrella Stanton (Gracillariidae)</td>
<td>0</td>
<td>No laboratory host range estimation done relative to US Phyllocnistis species; specificity assumed based on testing in Australia</td>
<td>Unknown relative to North American fauna</td>
<td>R+E–? 1994 [409, 412]</td>
<td></td>
</tr>
<tr>
<td>1995–2004</td>
<td>1 1995 USA, California</td>
<td>Aphelinoides anatolica Nowicki (Mymaridae)</td>
<td>Circulifer tenellus (Baker) (sometimes given as Neocallimus tenellus) Cicadellidae</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family (Cicadellidae)</td>
<td>R+E+ [413–415]</td>
</tr>
<tr>
<td>2 1995–1999 Canada from Europe</td>
<td>Lathrolestes ensator (Brauns) Ichneumonidae</td>
<td>Hoplocampa testudinea (Kug) Tenethridae</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Species? This species is known only from one host, but this does not exclude it existence on hosts not sampled</td>
<td>R+E+ [416–418]</td>
<td></td>
</tr>
<tr>
<td>3 1995 (Texas), 1997 (Florida) USA, from South America</td>
<td>Pseudacteon tricuspid Borgmeier (Phoridae)</td>
<td>Solenopsis invicta, S. richteri and hybrids. Formicidae</td>
<td>13 + 1</td>
<td>Zero NT attack on 13 NT ants not in Solenopsis; zero NT attack on one NT in Solenopsis; Post-release, zero NT attacks on 15 NT ants, including one native NT congener</td>
<td>Species Group level within Genus</td>
<td>R+E+ 1997 [419–423]</td>
<td></td>
</tr>
<tr>
<td>4 1995 (Texas), 1997 Florida USA, from South America</td>
<td>Pseudacteon litoralis Borgmeier (Phoridae)</td>
<td>Solenopsis invicta, S. richteri, and hybrids (Formicidae)</td>
<td>27 + 1</td>
<td>Zero NT attack on 27 NT ants not in Solenopsis; one NT Solenopis (S. geminata) attacked (at 9% of target rate) but no successful development</td>
<td>Species Group level within Genus</td>
<td>R+E+ 2003 [419–421, 424, 425]</td>
<td></td>
</tr>
<tr>
<td>5 USA Not petitioned for release</td>
<td>Pseudacteon wasmanni Schmitz (Phoridae)</td>
<td>Solenopsis invicta, S. richteri, and hybrids Formicidae</td>
<td>27 + 2</td>
<td>Zero NT attack on 27 NT ants not in Solenopsis; two NT Solenopis attacked (S. geminata, S. saevissima, S. geminata at 11% target rate) of ten NT pentatomids, six attracted oviposition, but only three supported complete development, two at levels equal to target and one at one-third level of target. Species in other families were rejected</td>
<td>Genus level or better</td>
<td>R-N/A² [219, 421, 424]</td>
<td></td>
</tr>
<tr>
<td>6 1996 Australia (from Argentina)</td>
<td>Trichopoda giacometelli (Blanchard) Tachinidae</td>
<td>Nezara viridula (L.) Pentatomidae</td>
<td>14</td>
<td>Family</td>
<td>Including target, acceptable hosts were found in four genera o pentatomids</td>
<td>R+E+ Ca 1997 [144, 426]</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Year Range</td>
<td>Location</td>
<td>Host(s)</td>
<td>Parasitoid(s)</td>
<td>Host Range</td>
<td>Comments</td>
<td></td>
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<tr>
<td>7</td>
<td>1996–1999 Spain</td>
<td>Citrostichus phyllocnistoides (Naryanin) Eulophidae</td>
<td>Phyllocnistis citrella Stainton (note about eight other parasitoids were introduced into Spain for this pest that are not listed here) Gracillariidae</td>
<td>Unknown</td>
<td>No testing done relative to European leafminers. Post-release attack on an unidentified Nepticulidae on Pistacia lentiscus L. and Stigmella sp. on Rubus ulmifolius Schott in Sicily and Jordan respectively [427], and in Sicily this parasitoid Cosmopterix pulcherimella, Chambers (Cosmopterigidae), on Parietaria diffusa Mert. &amp; W. D. J. Koch and Liriomyza sp. (Agromyzidae) on Mercurialis annua L. [428], See Karamaouna et al. [429] for details on displaced parasitoids.</td>
<td>Unknown relative to European fauna</td>
<td>R+/E+1996–1999 [427–432]</td>
</tr>
<tr>
<td>11</td>
<td>1997–1998 USA</td>
<td>Coccobius fulvus (Compere and Annecke) (also given as Physcus fulvus) Aphelinidae</td>
<td>Aulacaspis yasumatsui Takagi (Diaspididae)</td>
<td>Unknown</td>
<td>No laboratory host range estimation done</td>
<td>Unknown</td>
<td>Family? Records exist of use of several mealybugs, including Phenacoccus herreni Cox &amp; Williams [434], Ph. gossypii Ben-Dov [403], P. madeirensis, and Ferrisia virgata (Cockerell) [406]</td>
</tr>
<tr>
<td>12</td>
<td>1997–2000 USA, California from Mexico</td>
<td>Encarsia noyesi Hayat (Aphelinidae)</td>
<td>Aleurodicus dugesii Cockerel (Aleyrodidae)</td>
<td>Unknown</td>
<td>No laboratory host range estimation done</td>
<td>Unknown</td>
<td>Genus? The only other recorded host is Aleurodicus dispersus Russell [440]</td>
</tr>
<tr>
<td>13</td>
<td>1997 US Virgin Islands, from Egypt or Pakistan; also, 1998 Puerto Rico; 1999–2000 continental USA; and 2004 Mexico</td>
<td>Gyranusoides indica Shafee, Alam &amp; Agarwal (Encyrtidae)</td>
<td>Maconellicoccus hirsutus (Green) Pseudococcidae</td>
<td>Unknown</td>
<td>No laboratory host range estimation done</td>
<td>Unknown</td>
<td>Family? No other information on other hosts was found. Post-hoc sampling of other mealybugs in California following release of the parasitoid and its establishment there on the target did not detect any parasitism of either Phenacoccus solenopsis Tinsley or Ferrisia species [442]</td>
</tr>
<tr>
<td>Year released</td>
<td>Agent</td>
<td>Target</td>
<td>No. of test spp</td>
<td>Test outcomes</td>
<td>Smallest taxon including all likely hosts</td>
<td>Released Established^{1} (year if given equals first release)</td>
<td>References</td>
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<tr>
<td>1997 USA Florida</td>
<td>Ceratogramma etiennei Delvare (Trichogrammatidae)</td>
<td>Diaprepes abbreviatus (L.) Curculionidae</td>
<td>8</td>
<td>No parasitism of seven NT species of Lepidopteran eggs or those of one NT species of weevil</td>
<td>Family? Attacks eggs of weevils concealed in plant tissues</td>
<td>R+/E− 1994</td>
<td>[443–446]</td>
</tr>
<tr>
<td>1998–1999, Guam from India</td>
<td>Euplectrus maternus Bhatnagar (Eulophidae)</td>
<td>Eudocima (Othreis) fullonia (Clerck) Noctuidae</td>
<td>0</td>
<td>No host range testing reported</td>
<td>Genus? Two additional species of underwing moths in the genus Eudocima are known to be parasitized: E. materna L. and E. homaena (Hübner)</td>
<td>R+/E−</td>
<td>Bhumannavar and Viraktamath [452]; Muniappan et al. [453]</td>
</tr>
<tr>
<td>1998–2000 USA from Central America</td>
<td>Idioporus affinis La Safle et Polaszek (Pteromalidae)</td>
<td>Aleurodiscus dugesi Cockerell (Aleyrodidae)</td>
<td>0</td>
<td>No host range testing reported</td>
<td>Family? No other hosts reported but little literature of any kind</td>
<td>R+/E+</td>
<td>[441]</td>
</tr>
<tr>
<td>1998 USA from Florida</td>
<td>Quadrastichus haitiensis (Gahan) Eulophidae</td>
<td>Diaprepes abbreviatus (L.) Curculionidae</td>
<td>0</td>
<td>No host range testing reported</td>
<td>Family? Attacks eggs of weevils concealed in plant tissues Genus.</td>
<td>R+/E+ 1998</td>
<td>[445, 446]</td>
</tr>
<tr>
<td>1998 USA</td>
<td>Tetrastichus sertifer Thomson (Eulophidae)</td>
<td>Lilioceris illi Scopoli (Chrysomelidae)</td>
<td>10</td>
<td>Two NT European species of Lilioceris tested and both were attacked; eight species of North American non-Lilioceris (six same family; two other families) and none were attacked</td>
<td>Genus</td>
<td>R+/E+ 1999</td>
<td>[451, 454–456]</td>
</tr>
<tr>
<td>1999 USA</td>
<td>Pseudacteon curvatus Borgmeier (Phoridae)</td>
<td>Solenopsis invicta Buren, S. richteri Forel and hybrids (Formicidae)</td>
<td>19 + 2^4</td>
<td>Zero NT – attack of 19 NT ants not in Solenopsis; two NT – attack of two NT Solenopsis, but at lower rates than on target; two positive NT but no significant attack in field in post-release evaluation</td>
<td>Genus</td>
<td>R+/E+ 1999/2000</td>
<td>[419, 423, 457–459]</td>
</tr>
<tr>
<td>2000 USA, Florida (from Japan?)</td>
<td>Lysiphlebia japonica Ashmead (Braconidae)</td>
<td>Toxoptera citricida (Kirkaldy) Aphididae</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Unknown</td>
<td>R+/E−</td>
<td>[460, 461]</td>
</tr>
<tr>
<td>2000–2003 Mexico; and Guam 2002</td>
<td>Acerophagus papayae Noves and Schauff (Encyrtidae)</td>
<td>Paracoccus marginatus Williams and Granara De Willink (Pseudococcidae) Toxoptera citricida Kirkaldy (Aphididae)</td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family?</td>
<td>R+/E+</td>
<td>[462]</td>
</tr>
<tr>
<td>2000 USA, Florida (from Guam)</td>
<td>Lipolexis oregmae Gahan (introduced as L. scutellaris Mackauer) Braconidae, Aphidinidae</td>
<td></td>
<td>0</td>
<td>No laboratory host range estimation done</td>
<td>Family? Post-release field studies found parasitism of two black citrus aphid species, Toxoptera citricida (Kirkaldy) and T. auranti (Boyer de Fonscolome), and cowpea aphid (Aphis craccivora Koch), spirea aphid (Aphis spireae scolis Patch) and melon aphid (Aphis gossypii Glover)</td>
<td>R+/E+ 2001–2002</td>
<td>[463]</td>
</tr>
<tr>
<td>Year</td>
<td>Location</td>
<td>Host Species</td>
<td>Parasitoid Species</td>
<td>Notes</td>
<td></td>
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</tr>
<tr>
<td>2000</td>
<td>Caribbean and Florida (USA) from China</td>
<td>Anagyrus kamali Moursi (Encyrtidae)</td>
<td>Maconellicoccus hirsutus Green (Pseudococcidae)</td>
<td>Two NT species of Planococcus were attacked, but failed to support complete development.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2000</td>
<td>USA, California, from Australia</td>
<td>Psyllaephagus bitleus Riek (Encyrtidae)</td>
<td>Glycaspis brimblecombei Moore (Psyllidae)</td>
<td>None of the three NT eucalyptus-feeding psyllids tested (Trioza eugeniise Froggatt, Ctenarytaina eucalypti [Maskell], Boreoglycaspis melaleucae Moore) were attacked.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2000</td>
<td>Trinidad</td>
<td>Anitus hesperidum Silvestri (Platygasteridae)</td>
<td>Aleurocanthus wogltimi Ashby (Aleyrodidae)</td>
<td>No host range testing reported.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2000 USA Florida</td>
<td>Aprostocetus vaquitarum Wolcott (Eulophidae)</td>
<td>Diaprepes abbreviatus (L.) Curculionidae</td>
<td>No host range testing reported.</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2001 Mexico</td>
<td>Phymastichus coffea (LaSalie)</td>
<td>Hypothenemus hampei (Ferrari)</td>
<td>Of three NT species in same genus as target, two were successfully parasitized and one was not. Two other species in other weevil genera were not parasitized. Attack on two Hypothenemus species were at levels of 14 and 6%</td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td>2001 Kenya (via South Africa, via Mauritius, but originally from Sri Lanka)</td>
<td>Xanthopimpla stemmator Thunberg</td>
<td>Chilo partellus (Swinhoe) Busseola fusca Fuller</td>
<td>One NT was as suitable as target hosts; one NT was inferior with a low proportion of accepted hosts producing parasitoid progeny.</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2001 New Zealand</td>
<td>Thripobius javae (Girault) (= T. semiluteus Boucek)</td>
<td>Heliothrips haemorrhoidalis (Bouché)</td>
<td>Two NT Panchaetothripinae thrips were exposed – the native Sigmothrips adearouana (Ward) and the African thrips Hercinothrips bicinctus Bagnall (adventive in NZ). Both were successfully parasitized.</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002 USA (Guam)</td>
<td>Anagyrus loecki Noyes (Encyrtidae)</td>
<td>Paracoccus marginatus Williams and Granara De Willink (Pseudococcidae)</td>
<td>No host range testing reported.</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002 USA (Guam)</td>
<td>Pseudoleptomastix mexicana Noyes and Schauf (Encyrtidae)</td>
<td>Paracoccus marginatus Williams and Granara De Willink (Pseudococcidae)</td>
<td>No host range testing reported.</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Year released</td>
<td>Agent</td>
<td>Target</td>
<td>No. of test spp.</td>
<td>Test outcomes</td>
<td>Smallest taxon including all likely hosts</td>
<td>Released Established</td>
<td>References</td>
</tr>
<tr>
<td>---------------</td>
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</tr>
<tr>
<td>32 2002 USA</td>
<td>Diaparsis jucunda (Holmgren)</td>
<td>Lilioceris lilii Scopoli</td>
<td>10</td>
<td>Two NT European species of Lilioceris tested and both were attacked; eight species of North American non-Lilioceris (six same family; two other families) and none were attacked</td>
<td>Genus: There are no native congeneres in North America</td>
<td>R+/E+ 2003</td>
<td>[451, 454, 456]</td>
</tr>
<tr>
<td>33 2002 USA</td>
<td>Lemophagus errabundus Gravenhorst</td>
<td>Lilioceris lilii Scopoli</td>
<td>10</td>
<td>Two NT European species of Lilioceris tested and both were attacked; eight species of North American non-Lilioceris (six same family; two other families) and none were attacked</td>
<td>Genus: There are no native congeneres in North America</td>
<td>R+/E+ 2003</td>
<td>[456]</td>
</tr>
<tr>
<td>34 ~2002 USA Never petitioned</td>
<td>Lemophagus pulcher Szépligeti</td>
<td>Lilioceris lilii Scopoli</td>
<td>10</td>
<td>Two NT European species of Lilioceris tested and both were attacked; eight species of North American non-Lilioceris (six same family; two other families) and were two were attacked</td>
<td>Subfamily level: ^4^Criocerinae at best</td>
<td>Not released</td>
<td>[451, 454]</td>
</tr>
<tr>
<td>35 2002 USA, California (from Egypt)</td>
<td>Allotropa nr. mecria (Walker)</td>
<td>Maconellicoccus hirsutus (Green)</td>
<td>4</td>
<td>Four NT-no parasitism in four mealybugs from three other genera (Pseudococcus, Paracoccus, and Phenacoccus) one NT species affected by host feeding</td>
<td>Genus?</td>
<td>R+/E- Released in 2003–2004</td>
<td>[442, 469]</td>
</tr>
<tr>
<td>36 ~2004 USA Not petitioned for release</td>
<td>Bracon celer Szépligeti</td>
<td>Bactrocera oleae Gmelin</td>
<td>3</td>
<td>No attack on one NT gall making tephritid, but successful parasitism on another, Parafreutreta regalis Munro, and host deaths without successful parasitism on another</td>
<td>Family?</td>
<td>R- N/A-</td>
<td>[117]</td>
</tr>
<tr>
<td>37 2005 Tahiti (USA)</td>
<td>Gonatocerus ashmeadi Girault</td>
<td>Homalodisca vitripennis (Germar)</td>
<td>3</td>
<td>One of three NT species was an acceptable host, another Homalodisca sp.</td>
<td>Tribe: Hosts are in the Proconiini, especially species in the same genus as target and with similar egg size and deposition pattern</td>
<td>R+/E+ 2005</td>
<td>[91, 470, 471]</td>
</tr>
<tr>
<td>38 2002 California (from Southeastern USA)</td>
<td>Gonatocerus fasciatus Girault</td>
<td>Homalodisca vitripennis (Germar)</td>
<td>3</td>
<td>One of three NT species was an acceptable host, another Homalodisca sp.</td>
<td>Tribe: Hosts are in the Proconiini, especially species in the same genus as target and with similar egg size and deposition pattern</td>
<td>R+/E+ 2002</td>
<td>[471, 472]</td>
</tr>
<tr>
<td>39 2005 New Zealand (from Ireland)</td>
<td>Microtonus aethiopoides Loan (all female strain)</td>
<td>Sitona Lepidus Gyllenhali</td>
<td>9</td>
<td>Five NT native species were parasitized at rates from 2–28%; risk perceived to be lower than the already established Moroccan strain of this species</td>
<td>Family?</td>
<td>R+/E+ 2006</td>
<td>[473, 474]</td>
</tr>
<tr>
<td>No.</td>
<td>Year</td>
<td>Location</td>
<td>Species family</td>
<td>Species</td>
<td>Host</td>
<td>Test Result</td>
<td>Comments</td>
</tr>
<tr>
<td>-----</td>
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</tr>
<tr>
<td>40</td>
<td>2005 USA</td>
<td>Haeckeliaia sperata Pinto</td>
<td>Diaprepes abbreviatus (L.)</td>
<td>No attack on eggs of two NT Lepidoptera or two NT Coleoptera (one Coccinellidae and one a non-Diaprepes Curculionidae)</td>
<td></td>
<td>R+/E+ 2006</td>
<td>[446]</td>
</tr>
<tr>
<td>41</td>
<td>2005 USA</td>
<td>Fidiobia dominica Evans and Peña</td>
<td>Diaprepes abbreviatus (L.)</td>
<td>No host range testing reported</td>
<td></td>
<td>R+/E+ 2006</td>
<td>[475]</td>
</tr>
<tr>
<td>42</td>
<td>2005 USA</td>
<td>Psyttalia boumbouri Sylvestri</td>
<td>Bactrocera oleae Gmelin</td>
<td>No attack on three NT lepithilds tested</td>
<td></td>
<td>R+/E+ 2005</td>
<td>[476]</td>
</tr>
<tr>
<td>43</td>
<td>2005 (Texas), 2007 (Florida) USA</td>
<td>Pseudacteon obtusus Borgmeier (Phoridae)</td>
<td>Solenopsis invicta, S. richteri and hybrids (Formicidae)</td>
<td>One NT species in Solenopsis tested (S. geminata) and found not to be attacked</td>
<td></td>
<td>R+/E+ 2006/2008</td>
<td>[477–479]</td>
</tr>
<tr>
<td>44</td>
<td>2005 USA</td>
<td>Pseudacteon nocens Borgmeier (Phoridae)</td>
<td>Solenopsis invicta, S. richteri and hybrids (Formicidae)</td>
<td>Low attack rates with one NT native Solenopsis</td>
<td></td>
<td>R+/E+ 2006</td>
<td>[478]</td>
</tr>
<tr>
<td>45</td>
<td>2005-present (mid-2016)</td>
<td>Citrostichus phyllocnistoides (Naryanin) Eulophidae</td>
<td>Phyllocnistis citrella Stainton (Gracillariidae)</td>
<td>No testing done relative to North American leafminers</td>
<td></td>
<td>R+/E+ Ca 2006</td>
<td>P. Stansly (pers. comm.)</td>
</tr>
<tr>
<td>46</td>
<td>2006 USA Minnesota</td>
<td>Binodoxys communis (Gahan)</td>
<td>Aphis glycines Matsumura</td>
<td>Six of eight NT, Aphis species were highly suitable, while two were either not or only marginally so. For 11 NT non-Aphis species, one was suitable, while three were marginally so and seven were not</td>
<td></td>
<td>R+/E+ − 2007</td>
<td>[203, 480, 481]</td>
</tr>
<tr>
<td>47</td>
<td>2006 Israel (from Australia)</td>
<td>Closterocerus chamaeleon (Girault)</td>
<td>Ophelimumus masked (Ashmead)</td>
<td>No testing done relative to Mediterranean gall makers</td>
<td></td>
<td>R+/E+ 2006</td>
<td>[482, 483]</td>
</tr>
<tr>
<td>48</td>
<td>2007 Israel (from Australia)</td>
<td>Stethynium ophelini Huber</td>
<td>Ophelimumus masked (Ashmead)</td>
<td>No testing done relative to Mediterranean gall makers. It may develop on other unidentified Ophelimum species (Zvi Mendel, pers. com.)</td>
<td></td>
<td>R+/E+ 2007</td>
<td>[482, 484]</td>
</tr>
<tr>
<td>49</td>
<td>2007 Comoros Islands (Indian Ocean) (from La Réunion)</td>
<td>Eremocerus coxoi Delvare</td>
<td>Aleurotrachelus australis Hempel</td>
<td>Tested against native whiteflies of Comoros Islands and none were attacked</td>
<td></td>
<td>R+/E+ 2007</td>
<td>[385, 486]</td>
</tr>
<tr>
<td>50</td>
<td>2007 USA Hawaii</td>
<td>Binodoxys communis (Gahan)</td>
<td>Aphis gossypii Glover</td>
<td>Three NT, non-native Aphis species were highly suitable while three NT, non-native non-Aphis species were either not suitable (one sp.) or marginally so (two spp.)</td>
<td></td>
<td>R+/E+ 2010</td>
<td>[487]</td>
</tr>
<tr>
<td>Year released</td>
<td>Agent</td>
<td>Target</td>
<td>No. of test spp.</td>
<td>Test outcomes</td>
<td>Smallest taxon including all likely hosts</td>
<td>Released Established¹ (year if given equals first release)</td>
<td>References</td>
</tr>
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</tr>
<tr>
<td>~2006 USA</td>
<td>Not petitioned for release</td>
<td>Fopius arisanus (Sonan)</td>
<td>Bactrocera oleae Gmelin</td>
<td>2</td>
<td>Known from the literature to develop on many fruit-feeding tephritids, including over 20 Bactrocera species and various species of Anastrepha, Carpomyia, Ceratitis, Dacus and Euphranta (at least 30 hosts). Did not attack two species of weed biocontrol gall-making tephritids tested</td>
<td>Family level, for fruit-feeders Given extensive list of host genera attached, it was rejected by scientists in charge</td>
<td>R- N/A- [488]</td>
</tr>
<tr>
<td>~2006 La Réunion (from Hawaii)</td>
<td>Fopius arisanus (Sonan)</td>
<td>Various pest frugivorous tephritids (no single target)</td>
<td>8</td>
<td>All eight local tephritids tested were attacked. This was seen as desirable</td>
<td>Family Known to attack at least 20 species of tephritids</td>
<td>R+/E+ ~2006 [489, 490]</td>
<td></td>
</tr>
<tr>
<td>2006? Not petitioned for release</td>
<td>Trichomalus perfectus (Walker)</td>
<td>Ceutorhynchus obstrictus (Marsham)</td>
<td>16⁻</td>
<td>Four NT attacked equal to target five NT attacked &lt;target seven NT not attacked</td>
<td>Genus level Not pursued as species-level specificity would be required to protect weed biocontrol agents</td>
<td>R– [491–493]</td>
<td></td>
</tr>
<tr>
<td>2007 USA</td>
<td>Spathius agrilii Yang</td>
<td>Agrilus planipennis (Fairmaire)</td>
<td>17 (F) + 9 (L)</td>
<td>Of 17 NT wood-boring species collected in the field in China, zero were attacked. Of nine NT Agrilus species, three were attacked and six not attacked in laboratory tests</td>
<td>Genus</td>
<td>R+/E+ 2007 [494–496]</td>
<td></td>
</tr>
<tr>
<td>2007 USA</td>
<td>Oobius agrili Zhang and Huang</td>
<td>Agrilus planipennis (Fairmaire)</td>
<td>12</td>
<td>In the laboratory, of six NT Agrilus, three were attacked and of six NT, non-Agrilus, none were attacked</td>
<td>Genus</td>
<td>R+/E+ 2007 [494, 496]</td>
<td></td>
</tr>
<tr>
<td>2007 USA</td>
<td>Tetrastichus planipennis Yang</td>
<td>Agrilus planipennis (Fairmaire)</td>
<td>6 (F) + 11 (L)</td>
<td>Of six NT Agrilus species collected in the field in China, zero were attacked. Of five NT Agrilus and six other buprestids, zero were attacked in laboratory tests</td>
<td>Species?</td>
<td>R+/E+ 2007 [494, 496]</td>
<td></td>
</tr>
<tr>
<td>2007 USA</td>
<td>Lixadmontia franki Wood</td>
<td>Metamasius callizona (Chevreul)²</td>
<td>1</td>
<td>One NT native Floridian congenera, Metamasius mosieri Barber, was tested and found to be attacked at significant rates in choice and no-choice tests</td>
<td>Genus Limited host range testing done</td>
<td>R+/E– 2007 [494] Frank, unpub.</td>
<td></td>
</tr>
<tr>
<td>2007 USA/Hawaii</td>
<td>Eurytoma erythrinae Gates</td>
<td>Quadrastichus erythrinae Kim</td>
<td>7</td>
<td>None of the seven NT gall-makers tested (one native, four bicontrol agents, two adventive) were attacked Eight NT-no parasitism on any tested species</td>
<td>Genus</td>
<td>R+/E+ 2008 [497]</td>
<td></td>
</tr>
<tr>
<td>~2008 Australia</td>
<td>Diareetes essigeliæ Starý and Zuparko</td>
<td>Essigella californica (Essig)</td>
<td>8</td>
<td>None of the 25 NT gall-makers tested (= no family-level native species in HI; tested used 13 families, two or which were endemic, and 19 are immigrant pests) were attacked</td>
<td>Family level, same as species level In Hawaii, there are not con-familial natives</td>
<td>R+/E+ ~2009 [498]</td>
<td></td>
</tr>
<tr>
<td>2008 USA/Hawaii</td>
<td>Aroplectrus dimerus L.</td>
<td>Darna pallivitta (Moore)</td>
<td>25</td>
<td>None of the 25 NT gall-makers tested (= no family-level native species in HI; tested used 13 families, two or which were endemic, and 19 are immigrant pests) were attacked</td>
<td>Genus?</td>
<td>R+/E+ 2010 [499]</td>
<td></td>
</tr>
<tr>
<td>Year</td>
<td>Location</td>
<td>Predator</td>
<td>Target</td>
<td>Results</td>
<td>Notes</td>
<td></td>
<td></td>
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</tr>
<tr>
<td>2008 USA/California</td>
<td>Tamarixia radiata (Waterson)</td>
<td>Diaphorina citri Kuwayama</td>
<td>Six NT-no parasitism</td>
<td>Genus?</td>
<td>R+/E+ 2010 [500]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ca 2008 Canada (from Europe) but never released</td>
<td>Aleochara bipustulata L.</td>
<td>Delia radicum (L.)</td>
<td>11 NT species supported psyllid development; most frequently attacked species had small pupae or were in families related to target</td>
<td>Order level</td>
<td>Never released [502]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ca 2009 Switzerland (from Mexico) but never released</td>
<td>Celatoria compressa (Wulp)</td>
<td>Diabrotica virgifera virgifera Leconte</td>
<td>Of nine NT species tested, the agent developed, at low rates, in only one test species</td>
<td>Species</td>
<td>R+/E+ 2010 [504]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2010 USA</td>
<td>Pseudacteon cultellatus Borgmeier (Phoridae)</td>
<td>Solenopsis invicta, S. richteri and hybrids (Formicidae)</td>
<td>One NT species in Solenopsis tested (S. geminata) and found to be attacked in about one-third of cases</td>
<td>Genus</td>
<td>R+/E? 2013 [504, 505]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2010 USA</td>
<td>Aphelinus glycinis Wooley and Hopper</td>
<td>Aphis glycines Matsumura</td>
<td>No NT attacks on aphids outside of genus Aphis. Of seven NT Aphis species, four were suitable for parasitism, while three were not</td>
<td>Species</td>
<td>R+/E+ [506–509]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ca 2011 New Zealand (from Tasmania, Australia)</td>
<td>Cotesia urabae (Austin &amp; Allen)</td>
<td>Uraba lugens Walker</td>
<td>One NT – substitute not attacked in native range one NT attacked by no development</td>
<td>Species</td>
<td>R+/E+ [506–509]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2013 USA/California</td>
<td>Diaphorencyrtus aligarhensis (Shafee et al.)</td>
<td>Diaphorina citri Kuwayama</td>
<td>Six NT – no parasitism; one adventive pest psyllid attacked (at 14% rate)</td>
<td>Genus?</td>
<td>+/-? [510]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2013 USA</td>
<td>Spathius galinae Belokobylskij and Strazanac</td>
<td>Agrilus planipennis Fairmaire</td>
<td>14 NT – no attack one pest NT-attacked</td>
<td>Genus</td>
<td>+/- 2015 [511, 512]</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2013 USA/Hawaii</td>
<td>Encarsia diaspidicola (Silvestri)</td>
<td>Pseudaulacaspis pentagona (Targioni)</td>
<td>None of the seven NT were parasitized or killed</td>
<td>Family?</td>
<td>R+/E+ [345, 513]</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Several diaspidid scales (including Pseudaulacaspis pentagona [Targioni-Tozzetti] and Quadraspidiotus perniciosus [Comstock]) are known hosts, but others are not hosts [345].
<table>
<thead>
<tr>
<th>Year released</th>
<th>Agent</th>
<th>Target</th>
<th>No. of test spp.</th>
<th>Test outcomes</th>
<th>Smallest taxon including all likely hosts</th>
<th>Released Established</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>27</td>
<td>Not yet petitioned for release but under study for release in the USA</td>
<td>Apanteles opuntiarum Martinez &amp; Berta</td>
<td>Cactoblastis cactorum</td>
<td>6</td>
<td>Six NT – five spp. in native range were not attacked; one sp. attacked</td>
<td>N/A</td>
<td>[514, 515]</td>
</tr>
</tbody>
</table>

1Outcomes: − (not released), +/+ (released and established), +/− (released but not established).
2Species tested in Australia included a range of other leafminers (one in target genus, three others in target family, seven more in four other families), as well as four gall makers and two less related weed biocontrol agents.
3Research group concluded this was primarily a parasitoid of S. saevissima and did not petition for release.
4Nineteen ants in genera other than that of the target (Solenopsis) and two in Solenopsis; same format used in following additional species of Pseudacteon.
5Assessments of coffee berry borer parasitoid host range was done after release had already occurred.
6Attack in laboratory was found on Lema trilineata White (Ceriocerinae), a native North American insect [454]; also, for all parasitoids of lily leaf beetle, potential conflict exists with use of Lilioceris beetles as future weed biological control agents of various invasive plants.
7All non-target species were in the same genus as the target pest.
8In Florida, there are three Metamasius weevils, two of which (including the target pest) attack bromeliads and one not. The nontarget species in bromeliads was attacked by the tachinid, but the NT species not in a bromeliad host was not. No other weevils outside of Metamasius attack bromeliads in Florida.
9Attack in field on Agrilus auroguttatus Shaefler unlikely because of extremely thick bark of oak hosts, but would be beneficial if it occurred, as this is highly damaging and invasive in California.
10Laboratory testing of this newly recognized species has yet to be done.
Table 2. Predacious insects (28 species) introduced between 1985 and 2015, with notes on the level of specificity of each, drawn from literature records (worldwide) and BioCat (for North America, including Mexico and the US overseas territories, 1985–2004 [end of available records]).

<table>
<thead>
<tr>
<th>Year released</th>
<th>Agent</th>
<th>Target</th>
<th>No. of Test spp.</th>
<th>Test outcomes</th>
<th>Likely safety</th>
<th>Reg. dec.¹</th>
<th>Ref.</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 1984 USA (from Korea) and 1990–1995 (from China)</td>
<td><em>Chilocorus kuwanae</em> (Silvestri) Coccinellidae</td>
<td><em>Unaspis euonymi</em> (Comstock) Diaspididae</td>
<td>0</td>
<td>No laboratory prey range estimation done</td>
<td>Order (Multiple families in the Coccoidea) Known field prey of this species include various diaspidid scales, e.g., <em>Unaspis yanonensis</em> (Kuwana) [516], <em>Quadraspidiotus perniciosus</em> (Comstock) [517], <em>Quadraspidiotus macroporanus</em> Takagi [518]; more broadly it is known to feed on at least 28 scale species in five scale families [519], including the mealybug <em>Pseudococcus citriculus</em> Green [520], the coccid <em>Protopulvinaria mangiferae</em> (Green) [521], and the eriococcid <em>Eriococcus lagerstroemiae</em> Kuwanae [522]; see also Bull et al. [523]</td>
<td>R+/E+ 1984/1990–1995</td>
<td>[342, 343, 516–523]</td>
</tr>
<tr>
<td>2 1984 USA (from Korea) and 1990–1995 (from China)</td>
<td><em>Cybocophalus nr. nipponicus</em> Enrody-Younga (for this analysis we conflate <em>C. nipponicus</em> and <em>C. nr. nipponicus</em>) Nitidulidae</td>
<td><em>Unaspis euonymi</em> (Comstock) Diaspididae</td>
<td>0</td>
<td>No laboratory prey range estimation done No host range testing done before the introduction</td>
<td>Family (Diaspididae) Known field prey of this species include various diaspidid scales, e.g., <em>Quadraspidiotus macroporanus</em> Takagi [518], <em>Unaspis yanonensis</em> Kuwana [524], as well as (for adult feeding) eggs of the tetranychid <em>Panonychus citri</em> (McGregor) [525]; while adult feeding ranges include multiple families, oviposition and development only occurred in diaspidid scales, with reproduction on six of nine species tested [526], as reproduction is more similar to that of a parasitoid than predator</td>
<td>R+/E+ 1984/1990–1995</td>
<td>[342, 343, 518, 524–526]</td>
</tr>
<tr>
<td>Year released</td>
<td>Agent</td>
<td>Target</td>
<td>No. of Test spp.</td>
<td>Test outcomes</td>
<td>Likely safety</td>
<td>Reg. dec.</td>
<td>Ref.</td>
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<tr>
<td>3 1985 Oman (from India) Chilocorus nigritus (F.)</td>
<td>Coccinellidae</td>
<td>Aspidiotus destructor Sign. (Diaspididae)</td>
<td>0</td>
<td>No laboratory prey range estimation done</td>
<td>Order (Hemiptera)</td>
<td>R+/E+ 1985</td>
<td>[527–532]</td>
</tr>
<tr>
<td>4 1986, Guam and Mariana Islands from Hawaii Curinus coeruleus (Mulsant)</td>
<td>Coccinellidae</td>
<td>Heteropsylla cubana Crawford (Psyllidae)</td>
<td>0</td>
<td>No laboratory prey range estimation done</td>
<td>Order (Hemiptera)</td>
<td>R+/E+</td>
<td>[533–536]</td>
</tr>
<tr>
<td>6 1988, USA from Europe Rhizophagus grandis Gylenhal</td>
<td>Rhizophagidae</td>
<td>Dendroctonus terebrans (Olivier) Curculionidae: Scolytinae</td>
<td>0</td>
<td>No laboratory prey range estimation done</td>
<td>Genus</td>
<td>R+/E+</td>
<td>[541, 542]</td>
</tr>
<tr>
<td>7 1989, USA Hippodamia undecimnotata (Schneider) (other generic placements include Semiadalia and Adonia; also known as Hippodamia oculata) (Coccinellidae) Note: this species apparently invaded North America on its own about the same time it was being introduced into other parts of the continent [543]</td>
<td>Coccinellidae</td>
<td>Diuraphis noxia (Kurdjumov) Aphididae</td>
<td>0</td>
<td>No laboratory prey range estimation done</td>
<td>Family (Aphididae)</td>
<td>R+/E?</td>
<td>[543–547]</td>
</tr>
<tr>
<td>No.</td>
<td>Year</td>
<td>Location</td>
<td>Species</td>
<td>Prey</td>
<td>Notes</td>
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<tr>
<td>8</td>
<td>1989, USA</td>
<td><em>Propylea quatuordecimpunctata</em> (L.) (Coccinellidae)</td>
<td><em>Diuraphis noxia</em> (Kurdjumov)</td>
<td>Aphidae</td>
<td>No laboratory prey range estimation done</td>
<td>Family (Aphididae) Field prey include <em>Aphis fabae</em> Scopoli [548]; larvae can develop on <em>Acyrthosiphon pisum</em> Harris and <em>Rhopalosiphum maidis</em> (Fitch) [547] and <em>Schizaphis graminum</em> (Rondani) [550]</td>
<td>R+/E+ [543, 547–551]</td>
</tr>
<tr>
<td>9</td>
<td>1990 USA</td>
<td><em>Scymnus frontalis</em> (F.) Coccinellidae</td>
<td><em>Diuraphis noxia</em> (Kurdjumov)</td>
<td>Aphidae</td>
<td>No laboratory prey range estimation done</td>
<td>Family (Aphididae) Larvae develop well on several aphids, including <em>Schizaphis graminum</em> (Rondani), <em>Macrosiphum</em> (Sitobion) <em>avenae</em> (F.) and <em>Acyrthosiphon pisum</em> Harris [552]</td>
<td>R+/E? [547, 552]</td>
</tr>
<tr>
<td>10</td>
<td>1990 USA</td>
<td><em>Oenopia (Synharmonia) conglobata</em> (L.) Coccinellidae</td>
<td><em>Diuraphis noxia</em> (Kurdjumov)</td>
<td>Aphidae</td>
<td>No laboratory prey range estimation done</td>
<td>Two or most Orders Feeds on several families, including Aphididae, Monophlebidae, Coccidae, Chrysomelidae, and Aphidae) Field prey records include a monophlebid scale (<em>Matsucoccus josephi</em> Bodenheimer et Harpaz) [553], a coccid scale, <em>Ceroplastes rusci</em> (L.) [554], and various aphids, e.g., <em>Aphis craccivora</em> Koch [555] and <em>Hyadaphis tataricae</em> (Aizenberg) [556], and eggs of flea beetles (Chrysomelidae) [557], the psyllid <em>Euphyllura straminea</em> Loginova [558]</td>
<td>R+/E? [547, 553–558]</td>
</tr>
<tr>
<td>11</td>
<td>1991 Canada</td>
<td><em>Leucopis ninae</em> Tanasijtshuk (Chamaemyiidae)</td>
<td><em>Diuraphis noxia</em> (Kurdjumov)</td>
<td>Aphidae</td>
<td>No laboratory prey range estimation done</td>
<td>Family (Aphididae) Prey include aphids, i.e., <em>Aphis nerii</em> Boyer de Fonscolombe [559] and <em>Rhopalosiphum padi</em> (L.) [560]</td>
<td>R+/E– [559, 560]</td>
</tr>
<tr>
<td>12</td>
<td>1991 Canada</td>
<td><em>Leucopis atritaris</em> Tanasijtshuk (Chamaemyiidae)</td>
<td><em>Diuraphis noxia</em> (Kurdjumov)</td>
<td>Aphidae</td>
<td>No laboratory prey range estimation done</td>
<td>Family? or Unknown No other prey records were found</td>
<td>R+/E– [561]</td>
</tr>
<tr>
<td>Year released</td>
<td>Agent</td>
<td>Target</td>
<td>No. of Test spp.</td>
<td>Test outcomes</td>
<td>Likely safety</td>
<td>Reg. dec.¹</td>
<td>Ref.</td>
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<tr>
<td>14 1993–1999, USA, including Puerto Rico</td>
<td><em>Serangium parcesetosum</em> Sicard (formerly <em>Catana parcesetosa</em>) Coccinellidae</td>
<td><em>Bemisia tabaci</em> (Gennadius) strain B (Aleyrodidae)</td>
<td>0</td>
<td>No laboratory prey range estimation done</td>
<td>Family (Aleyrodidae)</td>
<td>R+/E?</td>
<td>[566–568]</td>
</tr>
<tr>
<td>15 1995 USA, Hawaii</td>
<td><em>Rodolia blackburni</em> Ukrainsky; formerly <em>Rodolia limbata</em> (Blackburn) Coccinellidae</td>
<td><em>Icerya aegyptiaca</em> (Douglas) Monophlebidae</td>
<td>0</td>
<td>No laboratory prey range estimation done</td>
<td>Two Families (Monophlebidae and Diaspididae)</td>
<td>R+/E+</td>
<td>[569–571]</td>
</tr>
<tr>
<td>16 1997–98, USA, Florida, from Thailand</td>
<td><em>Cybocephalus binotatus</em> Grouvelle (Nitidulidae)</td>
<td><em>Aulacaspis yasumatsui</em> Takagi (Diaspididae)</td>
<td>0</td>
<td>No laboratory prey range estimation done</td>
<td>Family? (Diaspididae)</td>
<td>R+/E+</td>
<td>[438, 572, 573]</td>
</tr>
<tr>
<td>17 1997 US Virgin Islands and 2004, Mexico</td>
<td><em>Cryptolaemus montrouzieri</em> (Mulsant) Coccinellidae</td>
<td>Mealybugs and other Hemiptera</td>
<td>0</td>
<td>No laboratory prey range estimation done</td>
<td>Order level &gt;8 families of Hemiptera known as prey</td>
<td>Past releases in many locations</td>
<td>[574]</td>
</tr>
<tr>
<td>Year</td>
<td>Location</td>
<td>Species</td>
<td>Prey</td>
<td>Results</td>
<td></td>
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<tr>
<td>18</td>
<td>Caribbean, from India</td>
<td>Scymnus coccivora</td>
<td>Maconellicoccus hirsutus</td>
<td>No laboratory prey range estimation done</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>19</td>
<td>2002 Ecuador</td>
<td>Rodolia cardinalis</td>
<td>Icerya purchasi</td>
<td>Of 16 species tested, larvae fed on only one NT species (same genus); none supported development. Adults did not feed on any of the eight NT species</td>
<td></td>
<td></td>
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</tr>
<tr>
<td>20</td>
<td>1994, USA from Japan</td>
<td>Sasajiscymnus tsugae</td>
<td>Adelges tsuga</td>
<td>Three NT adelgids and one NT aphid were fed on by adult beetles but at low rates compared with target; no non-adelgid prey was able to support development</td>
<td></td>
<td></td>
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</tr>
<tr>
<td>21</td>
<td>2003, USA, Virginia, from Washington</td>
<td>Laricobius nigrinus</td>
<td>Adelges tsuga</td>
<td>All three NT adelgids tested received eggs, but none supported full development</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>22</td>
<td>Ca. 2005, USA from China</td>
<td>Scymnus ningshanensis</td>
<td>Adelges tsuga</td>
<td>Two of the three NT adelgids and the NT aphid were fed on by adult beetles but at low rates compared with the target pest; development was assessed only on the aphid, which did not support development</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>23</td>
<td>2009</td>
<td>Laricobius osakensis</td>
<td>Adelges tsuga</td>
<td>No oviposition and no development on six NT species; Some feeding by adults on three adelgids and one aphid, but not two scales</td>
<td></td>
<td></td>
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</tbody>
</table>
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