

## Review Article

# Nontarget effects in the biocontrol of insects with insects, nematodes and microbial agents: the evidence

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### Abstract

A database of nontarget effects in classical and inundative biocontrol (of insects) was constructed based on various published and unpublished sources. Data were found relating to the nontarget effects in only 1.7% of all documented classical biocontrol introductions. Eighty-seven classical introductions have led to recorded nontarget effects, but most of these were minor. Seventeen of these introductions led to population reductions or effects of similar severity. Only one purported extinction was found, and even here the level of supporting evidence is low. While the direct evidence is thus almost non-existent, the impression from the available data is that there may have been many more nontarget effects, particularly in the very early history of classical biocontrol (especially in Hawaii), and that some of the polyphagous insect predators and parasitoids that are still now occasionally being used have significant community level impacts. In the latter cases, these introductions may usually have been justified in cost-benefit terms, or by socio-economic urgency, but this is not always clear.

Safety in inundative biocontrol is apparently justified not by the host range of the agents (often considerable), or even the population effects in the field on nontargets (which occur as many times as not), but by the transience of effects, lack of persistence, and the argument that agents, if not already present, are unlikely to establish. The ecological underpinning of these conclusions requires continuing investigation, so that we can be more sure about the safety of these agents under various use strategies.

### Introduction – the ERBIC Project

In 1998, a four-year research project on 'Evaluating Environmental Risks of Biological Control Introductions into Europe' (ERBIC) began, funded by the European Union. This project has two major aims. The first is to review and examine current and past arthropod biocontrol practices to assess their environmental impact. The second is to develop appropriate methodologies and guidelines to allow the assessment of biocontrol agents and practices of arthropod biocontrol in the future, potentially leading to legislation on the subject in the European Union.

The first step of the project (conceptually, if not chronologically), was a review of the known nontarget effects of biocontrol worldwide. This was intended to reveal the state of our knowledge

about the environmental effects of biocontrol, providing the foundations on which the empirical and modelling work of the project could build. The empirical work within the project examines four European biocontrol systems, in the context of different agroecosystems, which were taken as case studies, and subjected to various empirical investigations. Finally, ecological theory and modelling is being used to enhance the empirical and database work, as a conceptual basis for considering nontarget risk, and to attain a better understanding of the empirical systems. Here we discuss in detail the results of the review of worldwide knowledge, which we hope effectively summarizes all that is already known about nontarget effects in the biocontrol of insects using insect, nematode and pathogen agents.

Several reviews already address the issue of nontarget impacts in biocontrol (e.g. Howarth, 1983; Pimentel *et al.*, 1984; Howarth, 1990, 1991; Samways, 1994; Lockwood, 1996; Simberloff & Stiling, 1996a, b; Stiling & Simberloff, 2000). Earlier papers

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raised questions regarding specific biocontrol introductions, and brought up many conceptual issues regarding the potential complexity of such effects (e.g. Howarth, 1983, 1991). More recent papers, such as Stiling & Simberloff (2000), begin to tackle the problem in a much more quantitative manner. However, there is still a need for further quantitative analysis, and for the bringing together of a more exhaustive list of examples. Many of the examples given in the literature are to some extent circumstantial, in that the causality of various recorded population reductions and extinctions is unclear. Thus, some authors lay the blame on biocontrol in cases where the cause of extinction has been joint and unclear (Simberloff & Stiling, 1996a), while others argue that the lack or weakness of evidence exonerates biocontrol (Van Lenteren, 1997). Here we attempt to be as objective as possible, by building a database of effects which have some evidence, while extrapolating from these to the likely frequency of nontarget effects generally. While we do not dismiss entirely strong suspicions of nontarget effects that have little or no direct evidence, we do lend them less weight than those that have such evidence. We also attempt to put the consideration of the negative effects of biocontrol into the context of the prevailing attitudes at the time the introductions were made. We accept that the nontarget effects that have been recorded probably represent a minority of those that have actually occurred, but do not feel that a logical reaction to this is to decide that biocontrol as a whole is 'too risky'. Indeed, there is no 'biocontrol as a whole' – more accurately there is a complex history of changing attitudes and practices. The question is, are they changing quickly enough?

### Methods – Database Construction

Published and unpublished sources for this review were found by several means. These included conventional literature search, consulting staff at CABI Bioscience and those from the ERBIC project, and by direct correspondence with 30 other experts in biocontrol worldwide (see acknowledgements). By using such

methods, we should have uncovered all, or nearly all, observations of nontarget effects generated by introductions against insects that exist so far.

Data relating to introductions of both insect and microbial agents (including nematodes) against insect pests were collected. Both classical introductions and other styles of biocontrol (such as augmentation and inundation) were considered. The data were organized into records in which target/agent/nontarget combinations at a particular place (country or island) were defined. These records have a number of other fields, giving more details. Many of these correspond to fields in BIOCAT, the database of classical biocontrol introductions of insects (see Greathead & Greathead, 1992). Apart from such BIOCAT fields, which give general data on the introduction and its success, there are fields describing the name of the nontarget, its classification (order, family), and the literature or other reference from which the nontarget effect originated. Other fields are: biocontrol type (classical or inundation/augmentation); agent type (parasitoid, fungus, nematode, insect predator); and study type (RR = recorded rearing, M = study of mortality or parasitism, PR = study of population reduction, E = extinction). The nature of the nontarget effect itself is summarized in a few sentences in one field, and this is translated into a severity index (0-9) in another. This allows further quantitative analysis. The severity index is shown in Table 1. This index is somewhat arbitrary, but allows the various forms of nontarget impact (mortality, population reduction and extinction) to be assessed in a comparative manner. Only if mortality was above a certain level was this assumed to be equitable to or to have caused a population-level effect. Thus, a severity index of 3 can mean either a recorded mortality level greater than 40%, or evidence of a minor population-level impact. The empirical relationship between maximum percentage parasitism recorded and likelihood of biocontrol success (or partial success) supports the choice of the level of mortality we assume here to be sufficient to cause a population level impact (Hawkins *et al.*, 1993).

**Table 1.** Suggested severity indexes for nontarget effects of biological control.

Severity level	Type of impact
0	no records of consumption, infection, parasitism, population suppression, or extinction
1	<5% mortality induced by consumption/infection/parasitism or equivalent in sublethal effects on fecundity, with no recorded significant population consequences
2	5-40% mortality from consumption/infection/parasitism, with no recorded significant population consequences
3	>40% mortality from consumption/infection/parasitism (at one time on a local population) and/or significant (>10%) short-term depression of a local population
4	>40% short-term depression of a local population, or permanent significant (>10%) depression of a local population
5	>40% long-term suppression of a local population, or >10% long-term suppression of a population covering a large area (100 × 100 km or more)
6	>40% long-term suppression of a population covering a large area (100 × 100km or more)
7	apparent extinction of a population covering a small area, where recolonization is likely in the long term, such as where nearby areas or islands may harbour surviving populations
8	certified extinction over a small area, where recolonization is unlikely or impossible (due to an isolated island habitat and/or limited species range - so could imply extinction of the species)
9	certified extinction of a population over an area of 100 × 100 km or more

**Table 2.** Examples of classical biological control introductions of insects leading to a nontarget effects with a severity index (see Table 1) of 3 or greater.

Agent	Pest	Date	Country	Nontargets affected	Severity	Reference
<i>Bessa remota</i> (Aldrich)	<i>Levuana iridescens</i> Bethune-Baker	1925	Fiji	<i>Heteropan dolens</i> Druce	7	Robinson 1975
<i>Cotesia flavipes</i> Cameron	<i>Diatraea saccharalis</i> F.	1971-78	Brazil	<i>Paratheresia claripalpis</i> Wulp <i>Metagonistylum minense</i> Townsend (= <i>Lydella minense</i> (Townsend))	6	Bennett 1993
<i>Cotesia flavipes</i> Cameron	<i>Diatraea saccharalis</i> F.	1985	Mexico	<i>Apanteles diatraeae</i> Muesebeck	6	Bennett 1993
<i>Compsilura concinnata</i> Meigen	<i>Lymantria dispar</i> L.	1906-09, 1978-85	USA	<i>Cecropia</i> spp.	3	Stiling & Simberloff 2000
<i>Microctonus aethiopoies</i> Loan	<i>Sitona discoideus</i> Gyllenhal	1982	New Zealand	<i>Nonnotus albicans</i> (Broun), <i>Irenimus aequalis</i> (Broun), <i>Irenimus aemulator</i> (Broun)	3	Barratt <i>et al.</i> 1997
<i>Pteromalus puparum</i> (L.)	<i>Pieris rapae</i> (L.)	1932-33	New Zealand	<i>Bassaris itea</i> (F.) (= <i>Vanessa itea</i> (F.))	3	George Gibbs pers comm.
<i>Aphytis holoxanthus</i> DeBach	<i>Chrysomphalus ficus</i> Ashmead (= <i>C. aonidum</i> (L.))	1960	USA (Florida, Texas)	<i>Pseudhomalopoda prima</i> Girault	6	Bennett 1993; Selhime <i>et al.</i> 1969
<i>Aphytis holoxanthus</i> DeBach	<i>Chrysomphalus aonidum</i> (L.)	1962	Brazil	<i>Aphytis costalimai</i> (Gomes)	6	Bennett 1993
<i>Cales noaki</i> Howard	<i>Aleurothrixus floccosus</i> Maskell	1973	Italy	<i>Eretmocerus longicornis</i> Viggiani & Battaglia, <i>Encarsia aleurotubae</i> Viggiani, <i>Encarsia margaritiventris</i> (Mercet), <i>Amitus aleurotubae</i> Viggiani & Mazzone	4	Viggiani 1994
<i>Coccinella septempunctata</i> L.	<i>Diuraphis noxia</i> (Mordvilko)	1957-1978	USA	<i>Coccinella novemnotata</i> Herbst	5	Wheeler & Hoebeke 1995
<i>Coccinella septempunctata</i> L.	<i>Diuraphis noxia</i> (Mordvilko)	1956-1971	USA (South Dakota)	<i>Adalia bipunctata</i> (L.), <i>Coccinella transversoguttata richardsoni</i> Brown  <i>Hippodamia tredecimpunctata tibialis</i> (Say)	5  3	Elliott <i>et al.</i> 1996
<i>Copidosoma floridanum</i> (Ashmead)	<i>Chrysodeixis eriosoma</i> Doubleday	1969	New Zealand	'Trichogrammatoidea'	5	Roberts 1986
<i>Cryptolaemus montrouzieri</i> Mulsant	<i>Dysmicoccus brevipes</i> (Cockerell)	1938-39	Mauritius	<i>Dactylopius opuntiae</i> (Cockerell)	3	Goeden & Louda 1976
<i>Cryptolaemus montrouzieri</i> Mulsant	<i>Pseudococcus fragilis</i> Brain (= <i>P. calceolariae</i> (Maskell))	1929-30	South Africa	<i>Dactylopius opuntiae</i> (Cockerell)	3	Goeden & Louda 1976
<i>Tetrastichus dryi</i> Waterston (= <i>Tamarixia dryi</i> Waterston)	<i>Trioza erytraeae</i> Del Guercio	1974	La Réunion	<i>Trioza eastopi</i> Orian (= <i>T. litseae</i> Bordaga)	5	Aubert & Quilici 1983
<i>Trigonospila brevifacies</i> (Hardy)	<i>Epiphyas postvittana</i> (Walker)	1967-69	New Zealand	<i>Xanthopimpla rhopaloceros</i> Krieger, 'endemic moths'	5	Roberts 1986
<i>Trichopoda pilipes</i> (F.)	<i>Nezara viridula</i> (L.)	1962	Hawaii	<i>Coleotichus blackburniae</i> White	3	Follet <i>et al.</i> 2000
<i>Trissolcus basalus</i> Wollaston	<i>Nezara viridula</i> (L.)	1962	Hawaii	<i>Coleotichus blackburniae</i> White	3	Follet <i>et al.</i> 2000

The field of 'study type' deserves some elaboration. Some studies are simply records of hosts (collected in the field) from which a parasitoid has been reared, so are thus called recorded rearings (RR). Others record parasitism or other forms of mortality by percentage (mortality, M). Some studies actually measure the reductions of populations observed at some spatial scale over some time scale (population reduction, PR). A small number of reports consider suspected extinctions or extirpations (extinction, E). Each type of study has a different range of effects that it can detect. For example, studies that only examine mortality cannot determine population-level impacts. For some studies, the effect looked for may not have been found, but the presence of more minor effects is neither ruled-out nor proven. This would produce a 'zero' in terms of the severity index recorded, but such a zero does not itself imply no impact at all.

## Results

In total 512 records (i.e. unique nontarget/agent/location combinations) were gathered. Of these, 429 arose from classical biocontrol introductions, and 83 from inundative or augmentative uses of agents. The former group represented data pertaining to 92 classical programmes. All but one of these introductions were of insects, and were listed in the BIOCAT database. The exception was the introduction of the nematode *Steinernema scapterisci* Nguyen & Smart 1990 (Rhabditida, Steinernematidae) against pest mole crickets (*Scapteriscus* spp.) in Florida in 1985, where the possibility of infection of the nontarget northern mole cricket, *Neocurtilla hexadactyla* (Perty) (Orth., Gryllotalpidae), was investigated (but infection was not found) (Parkman & Smart, 1996). Thus, for only 1.7% of the 5279 recorded classical introductions of insects recorded in the BIOCAT database, could data be found relating to nontarget effects or potential nontarget effects in the field. That the majority of these introductions were of parasitoids (rather than insect predators) probably just reflects their popularity as agents.

Data indicating positive evidence of nontarget utilization or negative impact were found for 87 of these classical introductions and these yielded 402 records of such effects (i.e. 402 unique nontarget/agent/location records). However, only 17 introductions (of 14 agents) were found for which there was evidence of a serious nontarget effect, such as a mortality rate beyond 40% or direct evidence of population consequences, and these provided 27 records. These records are listed in Table 2. The number of such records is low because a relatively low proportion of studies directly assessed the possibility of serious effects in classical biocontrol. Most records (347 out of 429) simply recorded the presence of host or prey utilization ('RR' studies). There were 42 records of quantified parasitism or mortality rates ('M' studies), 35 of assessments of population reduction ('PR' studies), and five cases where authors suggested a specific biocontrol introduction may have had a role in an extinction ('E' studies). Not all of these investigations or suggestions had any supporting evidence, and there were many studies in which effects (such as mortality or population reductions) were looked for but not found, giving many 'severity = 0' records.

The remaining records of the database were devoted to inundative or augmentative uses of agents. Most agent types were represented, namely nematodes, bacteria, fungi and parasitoids. Studies of inundation or augmentation here usually involved simulated biocontrol application on plots or in microcosms in which nontargets were sampled, rather than studies during or after actual operational use. Often commercial field trials that combined efficacy and environmental studies provided the data. It is assumed that these studies gave accurate data about the nontarget effects of these biocontrol methods in practice. Forty-seven instances of such an application affecting a nontarget (at any

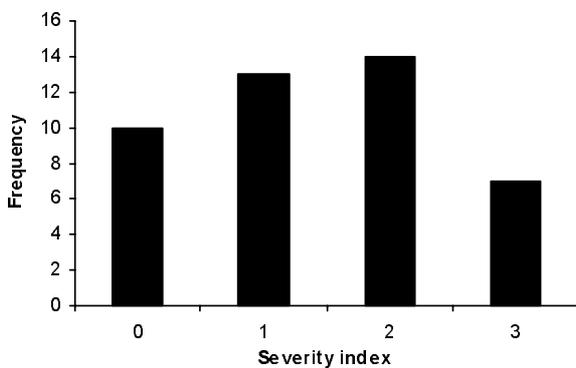
level) were recorded, out of 82 records. However, since organisms were often grouped into broad classes, such as orders or genera, in these studies, this figure is no doubt an underestimate of the total number of nontarget impacts *per se*. For pathogens and nematodes, it was often the case that the potential host range of the agents was known, from laboratory studies, to be very large, but dependent on dose and conditions. The field host range, however, was usually not determined directly. In most cases, only the population consequences were measured, as mortality or infection not leading to a population consequence was of less interest to the investigators. Factors such as persistence and dispersal processes, which might determine the environmental impact in practice, were given greater emphasis. There were only three examples regarding the use of parasitoids inundatively, and two of these consisted of a little information collected tangentially to a biocontrol programme, rather than an in-depth study of nontarget effects (Autrique *et al.*, 1989). While more data do exist on the nontarget effects of parasitoids used inundatively, such as *Trichogramma* spp., they were not readily available in a form that could be encapsulated in the database (but see Orr *et al.*, 2000). Further studies within the ERBIC project are illuminating the effects of *Trichogramma* and *Encarsia* parasitoids, as well as adding to the body of work on pathogens, nematodes, and inundative uses of predators. Finally, despite the intended exhaustiveness of this survey, it is felt that some examples of field nontarget tests of biopesticides and other inundative agents must remain to be included. In any case, this section of the database suffers the same problems of under-reporting as the section concerning classical introductions. All non-classical uses of agents now in the database together with brief details are given in Table 3.

### Classical biocontrol – 1. Studies of rearing records, parasitism and mortality rates

There were 308 simple recorded rearings on nontargets of parasitoids used in classical biocontrol and 39 simple records of nontarget utilization by other types of agent. These cases comprised the majority of all nontarget records. While such data may tell us something about the realised host ranges of agents (see Funasaki *et al.*, 1988; Hawkins & Marino, 1997; Stiling & Simberloff, 2000), they say little about the likelihood of more severe effects, and it is these serious effects with which we are chiefly concerned. To get an accurate impression of the relative frequency of minor effects, such as occasional low-percentage parasitism, and the more serious cases, we must restrict ourselves to quantitative studies of mortality (Figure 1). Where mortality has been properly studied, 64% of cases have a mortality rate less than 5%. In 90.5% of cases the degree of mortality has been less than 40%. These data suggest that many agents utilize nontarget hosts at a low level, and often do not generate sufficient mortality to imply some kind of population-level effect. Stiling & Simberloff (2000) used studies of the proportion of successful introductions in classical biocontrol to argue that 10-26% of cases where a nontarget is within the agent's fundamental host range result in population-level impacts. The current analysis suggests that this may be an overestimate, implying the true proportion is 9.5% or less. However, this conclusion is dependent on the choice of 40% mortality as the cut-off point for population level effects. If mortality rates less than 40% were likely to lead to significant population consequences, it could be concluded that the analysis presented here and that of Stiling & Simberloff (2000) are in broad agreement. However, both analyses potentially incorporate a bias in estimating the proportion of severe effects. In the case of Stiling & Simberloff (2000), this is due to their use of the proportion of successes in biocontrol to deduce proportions of severe effects for nontargets within the agents' host range. We would argue that the agents are chosen to have a maximum impact on the target, while the same is not true of nontargets, so that this extrapolation may not apply.

**Table 3.** Inundative biocontrol agents, and papers providing investigations into their nontarget effects.

Type of agent	Agent	Reference	Mean severity
Bacterium	<i>Bacillus thuringiensis</i> Berliner var. <i>kurstaki</i>	Miller 2000	4
Fungus	<i>Beauveria bassiana</i> (Bals.) Vuill. strain ARSEF 2883	James <i>et al.</i> 1995	5
Fungus	<i>Beauveria bassiana</i> (Bals.) Vuill. strain GHA	Jaronski <i>et al.</i> 1998	3
Fungus	<i>Metarhizium anisopliae</i> var. <i>acridum</i> Driver & Milner	Peveling <i>et al.</i> 1999	0
		Stolz 1999	0
Nematode	<i>Heterorhabditis bacteriophora</i> Poinar 1976	Georgis <i>et al.</i> 1991	0
Nematode	<i>Heterorhabditis megidis</i> Poinar, Jackson & Klein 1987	Bathon 1996	3
Nematode	<i>Heterorhabditis</i> sp.	Battisti 1994	0
Nematode	<i>Steinernema carpocapsae</i> (Weiser 1955) Curran 1989	Bathon 1996	1
		Battisti 1994	0
		Georgis <i>et al.</i> 1991	0
Nematode	<i>Steinernema feltiae</i> (Filipjev 1934) Wouts, Mracek, Gerdin & Bedding 1982	Bathon 1996	2.6
		Battisti 1994	2
		Georgis <i>et al.</i> 1991	0
Nematode	<i>Steinernema kraussei</i> (Steiner 1923) Travassos 1927	Battisti 1994	0
Parasitoid	<i>Diachasmimorpha longicaudata</i> Ashmead (= <i>Biosteres longicaudata</i> Ashmead)	Duan <i>et al.</i> 1997a	1
Parasitoid	<i>Lysiphlebus fabarum</i> Marshall	Autrique <i>et al.</i> 1989	3
Parasitoid	<i>Lysiphlebus testaceipes</i> (Cresson)	Autrique <i>et al.</i> 1989	4



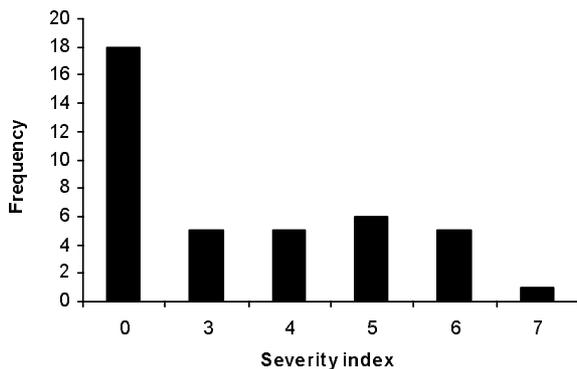
**Figure 1.** Frequencies of nontarget effects from studies records of quantified mortality (M studies) in classical biocontrol, as classified according to the severity index (Table 1): 0 = mortality or parasitism looked for, but none found; 1 = rearing record of unspecified proportions, or quantified mortality <5%; 2 = mortality of 5–40%; 3 = mortality >40%.

For the present study, the particular set of study systems represented in the database may influence the impression given of the relative frequencies of the different severity levels. Certain sets of studies, involving particular species complexes in particular parts of the world, and produced by particular research groups, dominate this section of the database, due to the relatively low number of studies conducted. If it were not for the relatively recent work on nontarget weevils in New Zealand by Barratt and co-workers (Barratt *et al.*, 1997), studies on tephritids and their parasitoids in Hawaii by Duan, Messing and co-workers (Duan *et al.*, 1996, 1997a; Duan & Messing, 1998), and a paper by Follett and co-workers (Follett *et al.*, 2000) on native pentatomids in Hawaii, the number of studies directly quantifying mortality in nontargets would be tiny. Many more studies of mortality in

independent systems, chosen randomly, are needed to get a truly unbiased impression. Here we briefly review the main studies.

Barratt and co-workers (e.g. Barratt *et al.*, 1997) focused on the introduction of the braconid parasitoids *Microctonus hyperodae* (Loan & Lloyd) (Hym., Braconidae) and *Microctonus aethioides* Loan (Hym., Braconidae) against the lucerne pests *Listronotus bonariensis* Kuschel (Col., Curculionidae) and *Sitona discoideus* Gyllenhal (Col., Curculionidae), respectively. *Microctonus aethioides* was the earlier introduction (made in 1982), and is the more polyphagous of the two agents. The recorded incidents of nontarget mortality were much more serious for this agent, with recorded parasitism rates greater than 40% (and often much higher) in three nontarget weevil species. *Microctonus hyperodae* was introduced in 1991 and, by contrast, is a specialist and almost completely benign. The differences could well be due to changing attitudes over the period between the introductions, although the two case studies serve more as an examination of the predictability of field host range from laboratory studies.

The Duan and Messing group examined fears about nontarget tephritids, including both natives and weed biocontrol agents, being affected by control agents of a number of tephritid pests, such as medfly (*Ceratitidis capitata* (Wiedemann); Dipt., Tephritidae), in Hawaii (Duan *et al.*, 1996, 1997a; Duan & Messing, 1998). There were a large number of species in their studies, and an even larger number of species combinations, but suffice to say that, overall, there was little evidence for substantial nontarget host utilization by the agents they examined. While none of their studies revealed field parasitism rates of a nontarget beyond 40%, there were examples of up to 28% parasitism by *Diachasmimorpha tryoni* Cameron (Hym., Braconidae) (introduced in 1913) against nontarget *Eutreta xanthochaeta* Aldrich (Dipt., Tephritidae), which is a biocontrol agent of the weed lantana (*Lantana camara* L.; Verbenaceae) (Duan & Messing, 1998). All of their other examples of field parasitism were below 5%. In fact, they examined in detail the host location behaviours of the parasitoids in question, which seem to limit the potential for nontarget effects (Duan & Messing, 1997a, b; Duan *et al.*, 1997b; Duan & Messing, 1998).



**Figure 2.** Frequencies of nontarget effects from studies of population reduction and extinction, classified according to the severity index (Table 1): 0 = no solid evidence of population level effects due to biocontrol; 3 = significant (>10%) local, short-term, population depression; 4 = >40% short-term depression, or >10% long-term depression of a local population; 5 = >40% long-term suppression of a local population or >10% suppression of a global population; 6 = >40% long-term suppression of a global population, 7 = apparent local extinction or near extinction.

Follett *et al.* (2000) looked at the consequences of two biocontrol introductions against the southern green stink bug *Nezara viridula* (L.) (Hem., Pentatomidae). They used a combination of examining museum specimens for parasitoid eggs, and of looking at parasitism in the field, to assess how serious the parasitism of various Hawaiian bugs has been. Cumulative maximum mortality from parasitism (calculated using different data sources on different stages) of *Coleotichus blackburniae* White (Hem., Scutelleridae) (the koa bug) by *Trissolcus basalis* Wollaston (Hym., Scelionidae) and *Trichopoda pilipes* (F.) (Dipt., Tachinidae) reached 50% and 57%, respectively. This great mortality is thought to have contributed to their changes in abundance (see next section – note also that these two records have been classified into both PR and M study types in calculating the relevant graphs). Three out of 58 (5.2%) museum specimens of alien pentatomid *Thyanta custator accerra* McAtee (Hem., Pentatomidae) were found with eggs of parasitoid *Trichopoda pilipes*, indicating that there had been an attempt at parasitism. Aside from these cases, all the examples of Follett *et al.* (2000) involved parasitism of less than 5%, and in most cases no parasitized specimens of the species investigated were found (although the samples sizes were sometimes very small). Howarth (1991) suggested that the observed drop off in abundance of the Hawaiian pentatomid genus *Oechalia* was due to these parasitoid introductions. So far only evidence to the contrary has been found from these more detailed direct studies of mortality (Follett *et al.*, 2000). It should also be noted that using the presence of eggs on museum specimens as an indicator of percentage parasitism will tend to, if anything, overestimate the actual parasitism rates, as their presence does not necessarily indicate successful parasitism or mortality of the host.

In addition to the three systems discussed above, there is a growing body of information on the negative effects of introductions into North America against the gypsy moth, *Lymantria dispar* L. (Lep., Lymantriidae), collected by various researchers (Schaefer *et al.*, 1989; Stamp & Bowers, 1998; Stiling & Simberloff, 2000). Much of this concerns the polyphagous tachinid agent *Compsilura concinnata* Meigen (Dipt., Tachinidae), first introduced in 1906 (Stamp & Bowers, 1998; Stiling & Simberloff, 2000). Unpublished data by Boettner

(referred to in Stiling & Simberloff, 2000) recorded an 81% parasitism rate on nontarget *Cecropia* spp., and *Hemileuca lucina* H. Edwards (Lep., Saturniidae) has been recorded with a 31.6% parasitism rate (Stamp & Bowers, 1998). It seems likely that other hosts of this polyphagous agent are utilized to a similar degree. Another gypsy moth agent, *Coccygomimus disparis* Viereck (= *Pimpla disparis* Viereck) (Hym., Ichneumonidae) has been reported parasitizing nontargets, in the case of *Malacosoma americanum* F. (Lep., Lasiocampidae) at rates of up to 33% (Schaefer *et al.*, 1989).

Inspection of the studies that dominated this section of the database demonstrates that, if anything, these cases were selected because nontarget effects were likely, and thus carry a bias toward more severe effects. In general these results should be treated prudently, because of this bias, because the total number of records is low, and since there are essentially only nine truly independent studies.

### Classical biocontrol – 2. Studies of population-level impacts and extinctions

Twenty-two records of population-level effects (from 'PR' and 'E' studies) stand up to scrutiny, with a further 18 identified where effects at a population level were looked for but not found (Figure 2). The choice of 'study systems' is more blatantly biased than the mortality studies discussed above. Many of these examples effectively selected themselves, when population level effects were noticed, or came as a response to the introduction of agents that are perceived as very risky. So, on the one hand there is a considerable level of under-reporting (because population reductions in most species would not be noticed at all), while on the other, when a species is selected for more detailed study, there must be a strong suspicion that it is being affected. The negative examples mostly come from studies of native coccinellids in the USA, which were monitored when the exotic species *Harmonia axyridis* Pallas (Col., Coccinellidae) and *Coccinella septempunctata* L. (Col., Coccinellidae) invaded their areas (Elliott *et al.*, 1996; Colunga-Garcia & Gage, 1998; see below).

Only one example of extinction could be identified. This is surprising, given the strong suspicion about the role of biocontrol in species depletion in some island ecosystems. In particular, local entomologists have argued that the early history of biocontrol had a substantial impact on the native fauna (particularly Lepidoptera) in Hawaii (Zimmerman, 1948, 1958a, b, 1978; Gagne & Howarth, 1985; Howarth, 1985), where there have been an extraordinary number of biocontrol introductions per unit area (Greathead & Greathead, 1992). Gagne & Howarth (1985) actually argued that particular native lepidopteran species, most of which are suspected of extinction sometime in the early 20th century, owe their demise partially or completely to the introduction of biocontrol agents into Hawaii. They do not, however, specify which agents they allege to have been involved. While they quote Zimmerman's volumes on Hawaii as a source reference (Zimmerman, 1948, 1958a, b, 1978), only in a minority of the cases did Zimmerman indicate that biocontrol agents introduced into Hawaii were ever reared from the moths in question. Accordingly, it is hard to see exactly how Gagne & Howarth (1985) drew their conclusions. Also, in checking the rearing records of Zimmerman (1948, 1958a, b, 1978) it becomes clear that the accidentally introduced species of parasitoid found among the rearing records of these nontarget hosts outnumber substantially the deliberate introductions (see also Funasaki *et al.*, 1988). Some of these appear to have been recorded as biocontrol introductions elsewhere, but were apparently not deliberately introduced into Hawaii. A possible implication of this is that these were illicit or unofficial introductions. This would explain Zimmerman's attitude to biocontrol. However, there is no evidence for this, and this should in any case not reflect on the record of

legitimate biocontrol exercises. Overall, we cannot conclude that biocontrol was blameless, but simply that specific examples are lacking, and with the mixture of legitimate introductions, possible illegitimate introductions, accidental introductions, and other factors (such as habitat loss) contributing to the extinction of local fauna, it is hard to ascertain the precise role of biocontrol.

Zimmerman (1948, 1958a, b, 1978) certainly felt that the effect of biocontrol on Hawaiian fauna was substantial. His testimony was essentially based on comparison with the observations of previous entomologists in the area, such as Swezey, and a detailed knowledge of the local fauna. The general impression given by Zimmerman was that biocontrol was likely to have contributed to the extinction of many lepidopteran species in Hawaii during the period 1900-1930 (Zimmerman, 1948). He made occasional references to whole genera (such as *Oeobia* and *Hedylepta* (Zimmerman, 1958b)), commenting on the severity of parasitism by foreign parasitoids, but the records provided on individual species betrayed the lack of more specific information. These records and the detailed review of Funasaki *et al.* (1988) in fact suggested that the role of biocontrol, while not negligible, was minor compared to that of accidental introductions.

The one relatively clear example of extinction of a moth seemingly caused by biocontrol is for Fiji, which has had far fewer introductions (Greathead & Greathead, 1992). This arose from the introduction of the parasitoid *Bessa remota* (Aldrich) (Dipt., Tachinidae) against levuana moth (*Levuana iridescens* Bethune-Baker; Lep., Zygaenidae), in 1925 (Tohill *et al.*, 1930; Robinson, 1975; Howarth, 1991). Robinson (1975) believed the extinction of the native moth *Heteropan dolens* Druce (Lep., Zygaenidae) was a result of this introduction, which also resulted in the apparent extinction (or near extinction) of the endemic target *L. iridescens* (the target may persist in some areas: see Paine (1994)). Again, corroborating evidence and an in-depth discussion of how this conclusion was arrived at is lacking, but the clear unequivocal statement specifying a target/agent/nontarget combination has allowed its inclusion in the database. However, whether some areas or islands still harbour the nontarget is uncertain, and the overall lack of detail given indicates that a severity level of 7 (rather than 8) is appropriate (see Table 1).

Another island example that is in some ways similar is the introduction of parasitoid *Tetrastichus dryi* Waterston (Hym., Eulophidae) into La Reunion in the Indian ocean, which drove its target extinct, but later inflicted heavy parasitism and population reductions on nontarget psyllid *Trioza eastopi* Orian (= *Trioza litseae* Bordaga) (Hem., Triozidae) (Aubert & Quilici, 1983). It seems that the extinction of the target of biocontrol is not only possible on island habitats, but may also be associated with major nontarget effects.

Biocontrol is implicated as a possible contributory factor in the extinction of the native harlequin bug *Murgantia histrionica* (Hahn) (Hem., Pentatomidae), which occurred following two introductions (of the parasitoids *Trichopoda pilipes* and *Trissolcus basalus*, see previous section) into Hawaii in 1962 (Howarth, 1991). A recent review of this case (Follett *et al.*, 2000) suggested that the evidence is essentially circumstantial. Moreover, *M. histrionica* was considered a pest itself, and its role as a potential host of one of the agents used was well known prior to introduction. There is some evidence that these introductions have also led to population consequences for the native koa bug *Coleotichus blackburniae*, as discussed above, but the presence of other substantial mortality factors and habitat loss has again made it impossible to establish biocontrol as the primary cause of population reductions in this species (Follett *et al.*, 2000). Nevertheless, the role of these agent species as contributory

factors, inducing quite severe mortality in this host, has allowed the inclusion of these examples (Table 2).

A more recent example of a possible extinction is the displacement of the native parasitoid, *Apanteles diatraeae* Muesebeck (Hym., Braconidae), by the introduced parasitoid, *Cotesia flavipes* Cameron (Hym., Braconidae) on the sugarcane borer, *Diatraea saccharalis* F. (Lep., Pyralidae). The native parasitoid could no longer be collected from the sugarcane borer in 1984-1985, implying severe population reduction and possible extinction (Bennett, 1993). Such cases of displacement are difficult to interpret, however. Firstly, extinction *per se* was not shown (for example, the natural habitat and native hosts were not examined). Secondly, a native species that maintains itself only by parasitizing a pest on an agricultural crop could be said to be 'living on borrowed time' in any case – habitat loss could be considered the underlying cause of extinction. So, the detection of extinction in natural habitats is perhaps more meaningful (but less likely to occur). Nevertheless, this example, and others that are similar, have been included (with a severity level of 6 in this case). Bennett (1993) himself gave three other relatively clear examples of displacement. Viggiani (1994) gave some examples involving aphelinid parasitoids in Italy. One of these was a report that the introduced parasitoid *Encarsia pergandiella* Howard (Hym., Aphelinidae) became a dominant parasitoid of *Trialeurodes vaporariorum* (Westwood) (Hem., Aleyrodidae) in some areas of Italy. However, as the population changes in other whitefly parasitoids were not commented on, there was no direct evidence of displacement or partial displacement via competition (and therefore no evidence of a nontarget effect). The introduction of *Cales noaki* Howard (Hym., Aphelinidae) into southern Italy against the woolly aphid, *Aleurothrixus floccosus* Maskell (Hem., Aleyrodidae), on the other hand, has apparently led to the complete displacement of all competing parasitoids in populations of the nontarget species *Aleurotuba jelinekii* (Frauenfeld) (Hem., Aleyrodidae) in some areas. This latter example has thus been included. Roberts (1986) reported the partial displacement of the native ichneumonid parasitoid, *Xanthopimpla rhopaloceros* Krieger (Hym., Ichneumonidae) by the tachinid agent *Trigonospila brevifacies* (Hardy) (Dipt., Tachinidae) in New Zealand, both of which parasitized 'endemic moths' (no particular species are mentioned). These native moths also seemed to suffer at a population level. Roberts (1986) also reported a marked reduction in the parasitism of the pest noctuid *Chrysodeixis eriosoma* Doubleday (Lep., Noctuidae) by native Trichogrammoidea after the introduction of the exotic egg parasitoid *Copidosoma floridanum* (Ashmead) (Hym., Encyrtidae) which can be considered a displacement. However, this may have occurred primarily through competition between parasitoids within the eggs of the host – it is thus possible that the overall abundance of these trichogrammoid species (which are probably generalists) was not radically altered. We give this example the benefit of the doubt, and a severity level of 5.

Exotic aphidophagous Coccinellidae in North America constitute another source of serious population-level nontarget effects, displacement of native coccinellid species being the primary concern. Obrycki *et al.* (2000) gave a good overview of these introductions and their effects, together with much relevant biological detail. Eight coccinellid species were established in North America in the 20th century (Obrycki *et al.*, 2000). In most cases it is uncertain whether or not the cause of establishment was a biocontrol introduction (there were normally a large number of these, over several states) or accidental importation at ports or shipping routes (there were usually several potential points of entry, such as large harbours and seaways), or both. In a sense, the exact manner of establishment is irrelevant, at least with respect to the species of most concern, *Harmonia axyridis* and *Coccinella septempunctata*. There were clearly concerted and repeated attempts to establish these coccinellids as biocontrol agents, as well

as to redistribute those that had already established within the continent to meet economic agendas. For example, releases of *C. septempunctata* in North America were made in over 50 places at various times (Schaefer *et al.*, 1987). So the biocontrol community should at least acknowledge the negative effects that such establishments might lead to. Several authors debated causes of establishment and documented the slow natural spread and more rapid redistribution of the various coccinellid species over the continent from their points of entry (Angelet *et al.*, 1979; Schaefer *et al.*, 1987; Day *et al.*, 1994; Tedders & Schaefer, 1994; Wheeler & Hoebeke, 1995), and a few quantified the effects that invasions of species into particular areas have had on native Coccinellidae (Wheeler & Hoebeke, 1995; Elliott *et al.*, 1996; Brown & Miller, 1998; Colunga-Garcia & Gage, 1998). Of the latter studies, some had equivocal results, indicating no real obvious effect on native coccinellid densities (Colunga-Garcia & Gage, 1998), or simply documented the displacement of one introduced coccinellid by another, without illuminating the original displacement of natives (Brown & Miller, 1998). Thus, while there has been much concern about the introduction (or accidental establishment) of *H. axyridis* into North America, there are no clear examples of population-level effects on native species. This is not the case for *C. septempunctata*, however. Elliott *et al.* (1996) clearly linked the 20-fold (or greater) decline in two native coccinellid species to the invasion of *C. septempunctata*. While Wheeler & Hoebeke (1995) were more circumspect about drawing such conclusions about the disappearance of native species *Coccinella novemnotata* Herbst (Col., Coccinellidae), noting that many factors may have caused its decline, they admitted that the invasion of *C. septempunctata* into the area was the most parsimonious explanation. Consumption of rare lepidopteran eggs is also a concern in the case of *C. septempunctata*, although no field data on actual mortality are available (Horn, 1991). Little information for either species exists on their effects on nontarget prey aphid species, apart from a list of prey utilized in the field by *H. axyridis* (Lamana & Miller, 1996).

The main reasoning behind worries regarding exotic coccinellids is that such species have the potential to radically change the structure of nontarget communities. Indeed, the community investigated by Brown & Miller (1998), albeit an agricultural one, was largely devoid of native coccinellid species, tending to be dominated by one introduced coccinellid or another. While the initial decline of such species was not documented, and there are no data with which to make comparisons of the situation before and after the arrival of *C. septempunctata*, this is hardly reassuring. The absence of evidence on the effect of introduced coccinellids on nontarget prey is not at all reassuring either. The negative interactions between competing coccinellids are likely to be mediated via prey depression (although intraguild predation may also have a large role). This itself must imply other, hitherto undetected, nontarget effects, and the possible broad-scale alteration of community composition. As Elliott *et al.* (1996) pointed out, the rationale behind some of these introductions appears questionable, in that their benefits for pest (aphid) control are not always evident. Elliott *et al.* (1996) were forced to reason indirectly to draw conclusions about the effect on control of *C. septempunctata*, suggesting that the lack of any change in predator (coccinellid) numbers implies no increase in predatory effect, therefore no change in pest control. However, Brown & Miller (1998) state clearly that the appearance (and community dominance) of *H. axyridis* was associated with considerable improvement in aphid control. There is the possibility that pest aphids, in agroecosystems such as that studied by Brown & Miller (1998), are well controlled by *H. axyridis*, whereas in natural arboreal habitats, with non-pestiferous prey, this species had much less effect. If so, however, this would have to be accepted as serendipitous. That these coccinellid introductions have had at least some environmental effects was predictable, in that it follows

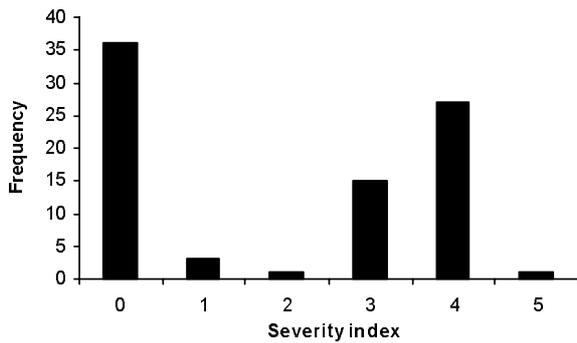
from their broad polyphagy and ability to disperse and establish. While there is a real case against *C. septempunctata*, the earlier introduction, the case against introducing *H. axyridis* is just being made – there is little evidence of nontarget effects and what exists is confined to agroecosystems (Brown & Miller, 1998; Colunga-Garcia & Gage, 1998). The only species *H. axyridis* has so far been unequivocally recorded having a major impact on is *C. septempunctata* itself (Brown & Miller, 1998). But the suspicion is that both species have had a significant impact on nontarget communities of aphids and their natural enemies, and it is hard to say for the moment whether or not the pest management benefits outweigh the environmental impacts.

Outside of North America, Goeden & Louda (1976) gave two examples of the introduction of the coccinellid *Cryptolaemus montrouzieri* Mulsant (Col., Coccinellidae) causing population level nontarget effects on *Dactylopius opuntiae* (Cockerell) (Hem., Dactylopiidae), a cochineal insect introduced for weed control. These were inferred from the detrimental effect noticed in the biocontrol of prickly pear by this species, in South Africa and Mauritius, and thus can be considered effects of some significance.

### The severity of impacts in augmentation and inundation

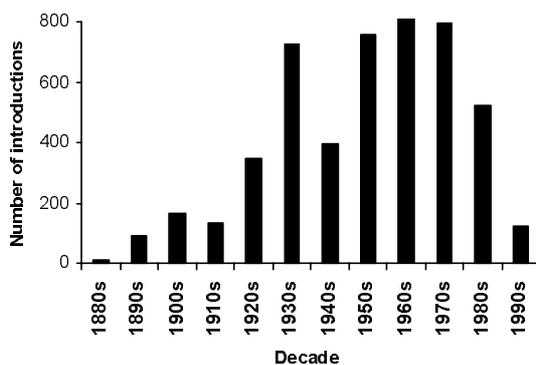
These studies, which mainly examine population reduction (in field trials or simulated scenarios), form a relatively unbiased body of data. Many of the agents are generalists, but their use is considered sufficiently safe due to their relative lack of persistence, and the consequent transience (or simply non-existence) of their nontarget impact. Field tests of pure host range and mortality are rare, which means that the number of cases of severity level 1 and 2 is greatly under-represented (Figure 3). The large number of cases of severity level zero appears from studies where (local) population effects have been looked for and none found. There is a substantial number of low intensity population level impacts (index = 3) and a few more serious ones. Over half (52%) of nontargets investigated suffered a population level impact or equivalent mortality (compared to  $\leq 9.5\%$  for classical biocontrol).

More detail on agents, agent types, and the general severity of their effects is given in Table 3. This table includes many of the most commonly used agents. There are very few such agents in each class, so making comparisons of severity levels between the classes is ill advised. For inundative use, only a few agent species dominate the usage in each class of agents, but there can be wide discrepancies between the species used in terms of their nontarget impact. For example, no nontarget impacts have been recorded in the field for the entomopathogenic fungus *Metarhizium anisopliae* var. *acridum* Driver & Milner (Deuteromycotina, Hyphomycetes), due to its specificity for grasshoppers and other acridids (but nontarget effects within this family, or affecting grasshopper nontargets, cannot be ruled out as yet) (Peveling *et al.*, 1999; Stolz, 1999). However, most other insect pathogens, such as certain *Beauveria bassiana* (Bals.) Vuill. (Deuteromycotina, Hyphomycetes) isolates, seem to frequently lead to nontarget effects, due to their relatively low specificity. These effects are sometimes quite extreme, albeit short-term and local (e.g. James *et al.*, 1995; Jaronski *et al.*, 1998). Many of the most commonly used agents have very wide host ranges because this provides a wide potential market, and thus justifies the research and development costs. However, occasionally products are developed with a specific pest or closely related set of pests in mind. In such cases, those developing the product have decided to limit the ultimate market for the product, in exchange for promoting the product over other methods of control (such as chemical pesticides or generalist agents), by means of emphasizing its environmentally friendly image. An example of the latter strategy is the use of *M. anisopliae* var. *acridum* against locusts and grasshoppers (Peveling *et al.*, 1999).

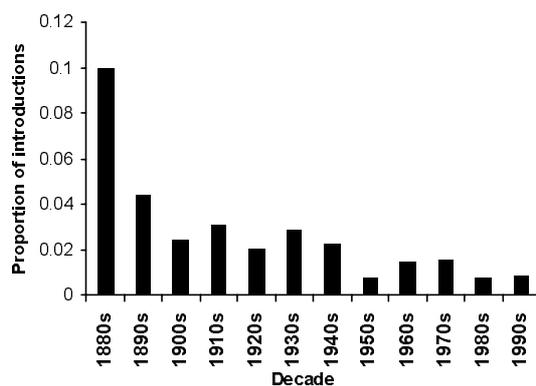


**Figure 3.** Severity index (see Table 1) in studies of augmentation or inundation, including all studies.

(a)



(b)



**Figure 4.** Patterns of nontarget effects over time. (a) Number of classical introductions of arthropod control agents per decade, as recorded in BIOCAT, (b) proportions of introductions for each decade leading to a recorded nontarget effect of any level.

While the impacts of some generalist inundative agents on a short-term local scale are clear, the interpretation of these impacts, in

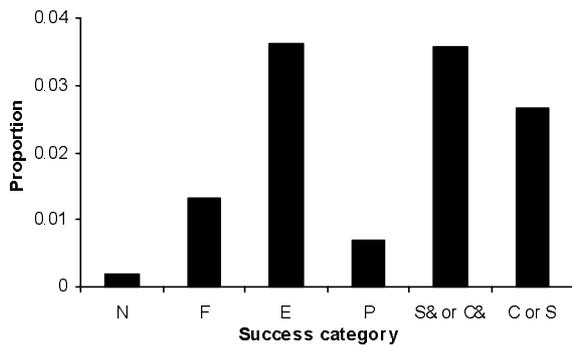
terms of more general environmental safety, is rather more subjective. Extrapolating from and interpreting such short-term experiments requires the tackling of a number of other ecological issues and a deeper understanding of the biology of the agent types. It is accepted, for example, that indigenous generalist nematodes used inundatively pose very little risk, for a series of specific biological reasons (Eulers & Hokkanen, 1996), despite examples of impacts.

However, some of the arguments about generalists seem to apply very broadly, to all agent types. The inundative use of such generalists is currently justified where they are either endemic to the area concerned and will 'decay' back to low endemic levels, or where they are exotic and it is known with some certainty that they will not persist. Thus while they clearly have nontarget effects, and sometimes quite profound ones, these are short-term and local, confined to the agricultural habitat of the pest. The environmental impact can be justified by its transience, and the assumption is that there will always be a pool of unaffected nontargets to recolonize the area. When an agent is used in a widespread and repeated manner over large areas, particularly natural habitats, the logic behind this justification may break down (as it does for chemical pesticides). The other obvious risk is that the agents are used in regions where they are not endemic, and they establish, because there may be no obligation on behalf of the end-user to undertake the necessary studies to ensure the safety of such use.

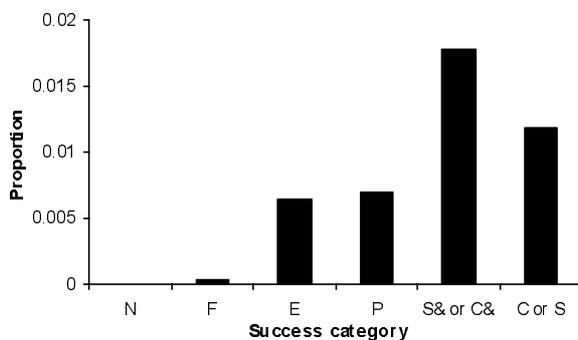
To conclude that inundation is safer, or less safe, than classical biocontrol would therefore miss the point. Of course, inundation may be safer, because it discourages the deliberate use and establishment of exotic agents, facilitating the use of indigenous agents instead. On the other hand, it sometimes may be less safe, because due to commercial pressures, and without effective regulation, it could encourage the widespread use of generalist agents, which may inadvertently be used where they are exotic. Also, there is the possibility that for some agents even normal use may be damaging if carried out in a blanket manner over vast areas of natural habitats for extended periods. Effective regulation could largely remove these risks.

Not all groups of species at risk from inundation have been given equal emphasis in these studies. The nontargets investigated in studies of inundation mainly consisted of insects, such as soil insects (Georgis *et al.*, 1991), environmentally important insects such as ants (Stolz, 1999) and forest Lepidoptera (Miller, 2000), and beneficials such as carabids, predatory bugs and Hymenoptera (Bathon, 1996; Peveling *et al.*, 1999; Stolz, 1999), including parasitoids of the target insects (Battisti, 1994). The need to look at these groups is obvious. The impacts on species similar to the agent, such as other pathogens or nematodes, however, are less often examined. For exotic uses of agents, competitive or indirect interactions leading to displacement of native species are obvious risks. In addition, such risks may even be an issue in short-lived inundative applications. Ishibashi & Kondo (1986) presented a laboratory study that examined the effects of applications of *Steinernema feltiae* (Filipjev 1934) Wouts, Mracek, Gerdin & Bedding 1982 (Rhabditida, Steinernematidae) and *Steinernema glaseri* (Steiner 1929) Wouts, Mracek, Gerdin & Bedding 1982 (Rhabditida, Steinernematidae) on the composition of nematode communities in soil. The densities of nontarget nematodes were affected when the agents were present in large numbers, but these densities quickly recovered. However, after recovery, the proportion of native rhabditids (as compared to other nematode groups) had increased. The reasons for this were unknown, highlighting the need for more research into such effects.

(a)



(b)



**Figure 5.** Classical biocontrol success and nontarget effects. (a) Success category (in BIOCAT) and proportion with a recorded nontarget effect of any level. (b) Success category and proportion with a recorded nontarget effect of severity  $>2$  (see Table 1). Success categories are defined as follows: N = unknown; F = agent failed to establish; E = agent established, but no recorded suppression of target (or other indicators of success); P = partial control, S& or C& = substantial or complete control in combination with another agent, C or S = substantial or complete control. See also Greathead & Greathead (1992).

### Patterns over time in classical biocontrol

The number of classical biocontrol introductions versus insects increased from the late nineteenth century to mid twentieth century (Figure 4). This was followed by a sustained period of high activity from the 1950s to the 1970s (see also Greathead & Greathead, 1992). The proportion of these introductions leading to (any) recorded nontarget effects was very much higher at the beginning of biocontrol history, and has tended to fall ever since (Figure 4b). However, a sceptic might discount the first two decades of this series (1880-1900), because sample sizes (of total introductions made) were low. This would still reveal a trend, but it would be much weaker. Certainly, the bar for the 1880s, in which there were only ten introductions, one of which led to a nontarget effect, adds to the visual impression of the decrease. In summary, we have some evidence that the chances of nontarget effects occurring for a given introduction have fallen over the years. This weak trend is perhaps all we could expect, given the nature of the data, but it is reassuring that it coincides with our expectations. There are too few examples of introductions leading to more serious effects (with mortality

$>40\%$ , or population reductions) to draw any conclusions about the temporal patterns of these.

### Biocontrol success and nontarget effects

An attempt was made to see if the number of recorded nontarget effects could be related to the success category of classical biocontrol (as stored in BIOCAT). The results are shown as proportions of introductions leading to effects of a given severity ( $>0$  and  $>2$ , giving Figures 5a and 5b, respectively). Few nontarget effects have been noted where the outcome of the introduction itself was not ascertained, which is not surprising. There are a substantial number of nontarget effects for introductions where agents did not seem to establish on the target pest (Figure 5a). This implies that some control agents have established as persisting populations on nontargets, and thus are likely to have a population level impact. There is little direct evidence of these impacts themselves, however (Figure 5b). Looking at all impacts (Figure 5a), we see a relatively small proportion of cases where agents leading to an only partial biocontrol success have been recorded utilizing nontargets (Figure 5a). We cannot think of a plausible explanation for this, aside from chance, however. It seems that more successful biocontrol introductions in general have a greater proportion of their potential nontarget effects (i.e. species attacked at all in the field) realised as major ones ( $>40\%$  mortality, population level impacts) (compare Figures 5a and 5b). Of course, there may be a sampling bias behind this relationship too, or it may simply be down to chance. The sample size of introductions with nontarget effects of severity  $>2$  is low (17), so the pattern should be treated very sceptically. Ecological theory (given certain assumptions) predicts that intermediate successes should lead to the greatest consequences for nontargets. Figure 5, however, provides no evidence for this, and indeed suggests that, if anything, more successful agents lead to more serious nontarget effects.

### Discussion

While only 1.7% of classical biocontrol introductions versus insects have associated with them some data on the existence of nontarget effects of any level, we can draw some conclusions about the frequency of nontarget effects indirectly. The quantitative study of mortality induced on nontargets has shown that the minor nontarget effects outnumber the major effects by nine to one, or more. Thus, only 9.5% of the instances of nontarget mortality in the database seem likely to lead to population-level effects. As most studies investigating population effects were chosen on the basis that nontarget effects were suspected, the real figure is perhaps much lower. For the sake of argument, however, we can continue to extrapolate using this figure. As 62.6% of releases have involved polyphagous species, attacking an average of about two nontarget hosts in the field each (Stiling & Simberloff, 2000), perhaps 11% of introductions have had serious population consequences for nontargets. We can tentatively estimate, using this logic, that up to around 628 nontargets have been affected at a population level over the history of biocontrol (again, the true number may be far lower).

Stiling & Simberloff (2000) suggested that the level of polyphagy in biocontrol agents generally is quite high. We have found that most agents in the database are recorded as utilizing or affecting the abundance of five or fewer nontarget species. The mean is 4.6, higher than that of Stiling & Simberloff (2000), but that is not surprising, considering that the proportion of polyphagous agents in a nontarget effects database is bound to be exaggerated. What is most important is that most of the effects observed were caused by a few very polyphagous agents (less than one-quarter of those in the database), which each had more than four nontarget hosts or prey in the field where introduced.

Thus, we have the impression that a minority of introductions, of polyphagous agents, has caused most nontarget effects. We can also say with confidence that many more such consequences have gone undetected. Out of a (worst-case scenario) prediction of 628 population-level nontarget effects, there are data only for 22. This means we have certainly missed some. The general impression from what we have is that the most destructive releases were of broadly polyphagous species, which could have been predicted. Not all of the introductions of polyphagous agents seem to have been justified by economic urgency or an entirely sober cost-benefit consideration, either (Elliott *et al.*, 1996).

Some of these introductions can certainly be excused on the basis that they were made a long time ago, before priorities had changed to take account of environmental impacts. This reasoning could be applied to the releases in the late 19th and early 20th centuries in Hawaii, which are suspected of contributing to decline of native moths (Zimmerman, 1948, 1958a, b, 1978; Gagne & Howarth, 1985; Funasaki *et al.*, 1988; Howarth, 1991). There were far more of such early insect biocontrol operations in Hawaii than in any other part of the world (189 had occurred by 1930, compared with 35 in mainland USA). It was effectively a testing ground for insect biocontrol, and an early form of biocontrol motivated primarily by economic grounds, with a low awareness or concern about environmental issues. Even there, however, in the worst imaginable scenario for native species, the negative effects of accidental introductions and habitat loss dwarfed those of biocontrol (Funasaki *et al.*, 1988; Follett *et al.*, 2000).

However, even in the 1980s and beyond, some polyphagous agents have been in regular use. Introductions of polyphagous coccinellids such as *Harmonia axyridis* in the USA, and some other polyphagous agents worldwide in recent times, indicate that the agencies involved are willing to risk an introduction which has a likelihood of leading to significant environmental effects when a substantial economic payoff or socio-economic urgency are involved. Overall, the general impression is that a great number of early nontarget effects, and even some recent ones, could have been avoided had the authorities involved simply adhered to broad guidelines such as the 1996 FAO Code of Conduct for the Import and Release of Biocontrol Agents (Schulten, 1997). The code (in sections 4.2.2 and 4.2.3) calls for background information, and experiments where necessary, to be used to assess host range prior to introduction. However, while the implication is that the use of very polyphagous agents should be limited to circumstances of extreme urgency, as the code only provides guidelines, it cannot rule their use out in all circumstances, or dictate to a country how serious a problem should be to justify the release. In practice the decision will come down to a cost-benefit analysis or value judgement on behalf of the authorities involved, which implies no common standard as such, but does mean that releases should be justified in writing. The lack of any rigid rules regarding releases is often seen as desirable, because the political nature of the decision to introduce is inescapable, and because the decision-making process needs to remain flexible to effectively reflect the political will and idiosyncratic circumstances in each case. However, this in turn implies that agents known to be or suspected of being environmentally destructive are occasionally released.

There are ambiguities surrounding the information provided by scientists, as well as the judgements made by decision-makers on the basis of that information. The broad guidelines and recommendations of the FAO code can be fulfilled at many levels, from the very superficial through to the very thorough. In addition, scientific uncertainty about the possible impact of an agent may remain large, even when a great deal of effort is made to predict it accurately. A great deal of information on the ecology of a parasitoid or predator can be required to achieve an accurate prediction about its field host range (Barratt *et al.*, 2000). Also

necessary is some knowledge of what the potential nontargets are – information that is not always readily available. While some very risky agents might be excluded on the basis of information already available on their host range (or simply on common sense grounds), for most an experimental approach is required. Such experiments have a number of difficulties (Sands, 1993; Barratt *et al.*, 1996; Onstad & McManus, 1996; Van Driesche & Hoddle, 1997). Overall, clear information on host range may be difficult to attain, and historically has not been the only factor in deciding whether a release goes ahead. The degree of acceptable environmental impact of a classical biocontrol introduction has thus varied, depending on a number of disparate factors. Among these are the attitudes of agencies involved, the urgency of the pest problem, the tractability of the ecology of the organisms, the level of knowledge of potential nontarget species, and the prevailing environmental attitudes of the time. In many problematic cases, such as the introduction of polyphagous coccinellids against Russian wheat aphid (*Diuraphis noxia* (Mordvilko); Hem., Aphididae), the urgency of the problem (as perceived by politicians and farmers) was the overriding factor in the introduction.

Inundative and augmentative methods of biocontrol have a different set of nontarget issues to classical biocontrol. The data regarding nontarget effects are also generally superior where available and often come from commercial trials. It seems that the developers and exporters of biopesticides in particular are effectively addressing section 4.5 of the FAO code, which calls (among other things) for information regarding nontarget effects in biopesticide usage. This is perhaps because biopesticides are also subject to another FAO code, that for the safe use of pesticides, and also (to a greater extent than classical biocontrol) the separate legislation of member governments. Historically, pesticides, and hence biopesticides, have been subject to blanket legislation, unlike classical biocontrol where each decision has been treated separately. Over 50% of the trials and tests on inundative methods actually show population level effects on nontargets. Most agents are considered 'safe', however, because they do not persist or disperse very far. The implicit assumption is that these agents will not be used repeatedly over large natural areas, and that any exotic use of agents will be carefully controlled. If these assumptions and conditions are met, their use can be considered safe, but the data indicate the potential of many agents to have some impact on nontargets if not used responsibly (i.e. if the above assumptions are not met, which seems quite conceivable). Most inundative agents are very broad generalists, which means that the risks are relatively high. The use of specialist agents, where possible, is always preferable, to limit the potential for nontarget effects. This obvious fact is usually in conflict with the economics of product development, however.

It is also important to admit our ignorance of the natural ecology of some inundative agents, such as insect pathogens and nematodes. Such ecological knowledge is needed so that we can understand the basis for the transience of any effects, and the conditions that prohibit exotic agents to establish permanently. This deeper knowledge is essential for prediction of how these processes may change under different application scenarios, and thus for a more circumspect assessment of risks. Assessment of risk for the exotic use of inundative agents generally can be quite difficult. For parasitoid and predator agents it covers issues such as host range and overwintering ability, which although relatively tractable still require much time and money to address properly. For nematodes and pathogens there are other issues, such as competition with native species within the soil or within low-density communities of native hosts. The ecological knowledge and risk assessment protocols to tackle these issues are

comparatively underdeveloped, but it is important not to ignore these risks.

Finally, we would all benefit from an adherence to sections 7.1.3 and 8.1.3 of the FAO code by all authorities involved in biocontrol, which indicate that biocontrol should be monitored to at least some degree for any nontarget effects, that this should be documented, and preferably published. This is the minimal requirement to ensure accountability in biocontrol for its environmental consequences.

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