

PREDICTING THE ECONOMIC IMPACT OF AN INVASIVE SPECIES ON AN ECOSYSTEM SERVICE

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Abstract. Quantifying the impact of alien invasive species on ecosystem services is an essential step in developing effective practices and policy for invasive species management. Here we develop a stochastic bioeconomic model that enables the economic impact of an invasive pest to be estimated before its arrival, based on relatively poorly specified ecological and economic parameters. We developed the model by using a hypothetical invasion of the varroa bee mite (*Varroa destructor*) into Australia and the negative flow-on effects that it would have on pollination by reducing honey bee populations, giving rise to a loss of pollination services, reduced crop yields, and additional production costs. If the mite were to continue to be prevented from entering the country over the next 30 years, we estimate that the economic costs avoided would be US\$16.4–38.8 million (Aus\$21.3–50.5 million) per year. We suggest that current invasion response funding arrangements in Australia, which do not acknowledge these avoided damages, require amendment.

Key words: *Apis mellifera*; Australia; bioeconomic model; economic impact assessment; European honey bee; invasive species; parasitic bee mite; pollination; *Varroa destructor*.

INTRODUCTION

Defining the extent of an alien invasive species' current or future impact is an important step in developing appropriate management responses. Without a quantified assessment of potential impact, there is little basis for determining what level of prevention or mitigation response is warranted (i.e., how much should be spent to maintain a service under threat), or who is responsible and ultimately bears the cost for this response. In this paper we use a hypothetical invasion of a parasitic bee mite, *Varroa destructor* (Anderson and Trueman 2000) into Australia, as a case history to present a method that can be generally applied to quantitatively estimate the private (or market) benefits of excluding invasive alien species of economic significance. Stringent biosecurity protocols are currently in place to protect Australia's freedom from *V. destructor* (along with a host of other exotic bee pests), but specific quantitative evidence of damage avoided as a result is lacking. This is a particularly interesting case study because the species directly threatened (the European honey bee *Apis mellifera* Linnaeus) is itself an invasive species. We are, in effect, estimating the economic benefits to be gained by protecting a naturalized invasive species through the exclusion of a potential new invader.

Invasive species are now acknowledged as a source of tremendous cost to the global economy. In the United

States alone, the Office of Technology Assessment (1993) estimated that 79 species had caused US\$97 billion in damage between 1906 and 1991 (all costs in the paper are expressed in US\$ unless otherwise stated). More general assessments of losses considering the effects of >50 000 pest species have estimated annual losses of US\$120–138 billion (Pimentel et al. 2000, 2005), while global losses are believed to be on the order of US\$1.5 trillion per annum (Pimentel et al. 2002). In addition to their economic significance, invasive alien species are also recognized as one of the greatest threats to biodiversity (Wilcove et al. 1998) and, hence, to ecosystem services and functioning. Crop pollination is recognized as a key ecosystem service provided by biodiversity (Daily 1997, Klein et al. 2007), and there have been numerous attempts to quantify its economic value (e.g., Cheung 1973, Johnson 1973, Robinson et al. 1989, Costanza et al. 1997, Burgett et al. 2004, Losey and Vaughan 2006), developed in the spirit that such services will be better managed if their benefit to the human economy is known. Estimates of the nonmarket benefits and costs associated with *V. destructor* exclusion, which would also provide valuable information to resource managers, are not available.

These aggregated estimates have been important in raising awareness of the potential magnitude of the problem, which is no doubt enormous, but they do not constitute a solid platform for the development of policy responses to specific species invasions (Waage et al. 2005). Importantly, from a management perspective, point estimates of aggregate economic damage do not provide policy makers with recommended investment

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options, nor are they intended to. Similarly, numbers that reflect aggregate ecosystem service values provide no information about how specific changes in ecosystem states might affect social interests and values (Toman 1998). Some of these previous valuations have been criticized on technical grounds (Turner et al. 1998, Daily et al. 2000), but it is the lack of specificity that we address in this paper.

Australia finds itself in a position of being able to observe the impact of *V. destructor* in other countries, and to make a decision about the level of investment it should make in trying to maintain its area freedom. The emergence of *V. destructor* as a parasite of *A. mellifera*, having transferred from the Eastern honey bee *Apis cerana* Fabricius (Anderson and Trueman 2000), has resulted in significant impact on honey bee populations throughout the world (Martin 1998, Sammataro et al. 2000, Vetharaniam and Barlow 2006). In North America, where it is thought to have arrived in the mid-1980s by way of illegal commercial bee movements from Europe and South America (de Guzman et al. 1997), it has since caused severe losses of both feral and commercial *A. mellifera* colonies (Watanabe 1994), with most colony deaths probably resulting from the effects of viruses that are activated and transmitted by the mite (Martin 2001, Chen et al. 2004, Sumpter and Martin 2004). Total damages have been estimated at US\$0.6–14.6 billion (Robinson et al. 1989, Muth and Thurman 1995, Morse and Calderone 2000). The mite also has been recently introduced to both the North and South Islands of New Zealand (Munn 2004).

The impact that *V. destructor* naturalization would have on the delivery of pollination services in Australia is expected to be particularly severe, because most of the viruses that the mite transmits, and that cause high colony mortality, are present in the local honey bee population (Anderson and Gibbs 1988). In addition, the Australian native bee fauna includes no other social Apidae, except for the Meliponinae (restricted to tropical and subtropical regions; Heard [1999]), and introduced Bombinae (restricted to Tasmania; Semmens et al. [1993]). *Apis mellifera* was introduced into Australia soon after the arrival of the first Europeans (Paton 1995, Cunningham et al. 2002) and has become widespread, sometimes reaching very high population densities (Oldroyd et al. 1994, 1997). The success of this invasion is probably due to the generally favorable climate and the abundance of relatively accessible flowers that produce generous amounts of nectar (Ford 1985). Over much of the landscape, honey bees have no competitors of comparable efficiency. For these reasons, insect-pollinated crops in Australia commonly get a very high level of service from feral *A. mellifera*, particularly in the case of horticultural crops.

To estimate the pollination benefits of excluding *V. destructor* from Australia, we develop a relatively simple stochastic bioeconomic model that can be used generally to simulate the spread and impact of invasive species

that may occur in the future. Following introduction and establishment, spread is assumed to continue until the species becomes naturalized. Naturalization is complete when a species spreads to its full capacity within an environment, such that descendants of the original specimens introduced into that environment become permanent, non-spreading members of the biota (Mack 1996, Mack and Lonsdale 2001). The model estimates the economic costs throughout this process by making simple predictions about the commercial behavior of affected producers and what behavioral change is likely to cost. In the *V. destructor* example, we consider these costs in relation to producers who rely on wild honey bee pollinators. We thus depart from methods used in past assessments for this species in the Australian context, e.g., Gill (1989) and Gordon and Davis (2003), which placed a value on honey bee pollination services and assumed an immediate 100% loss of those services in the presence of *V. destructor*. We focus specifically on those pollination service losses directly attributable to the expected spread of the mite over time.

METHODS

Bioeconomic model

The stochastic simulation model determines total expected (or probability-weighted) damage from an invasive species over a specified period of time. Uncertain or variable parameters are specified as probability distributions, and 10 000 model iterations are run with values randomly sampled across the range of each distribution, using Monte Carlo simulation. Our analysis of *Varroa destructor* is limited to the pollination benefits accruing to private agricultural industries. The share of these benefits enjoyed by plant industries is determined using expected cost implications of the mite arriving in Australia and gradually eliminating the feral *Apis mellifera* population.

The model of invasion employed here treats an outbreak as being the product of two distinct events: arrival and establishment. The process of detecting an outbreak is not considered separately from arrival (thus prohibiting an examination of different surveillance and detection policies that may reduce the time until discovery). The probability of a successful outbreak, or the transition between a “with pest” (call it event *a*) and “without pest” (event *b*) state, is described as a regular Markov process such that the probability of event *a* occurring in any given time period will reduce to a constant value after several periods. Each element of the transition matrix

$$\mathbf{P} = \begin{pmatrix} P_{aa} & P_{ab} \\ P_{ba} & P_{bb} \end{pmatrix}$$

where *a* defines the row and *b* the column, provides an indication of the invasibility of the ecosystem concerned (Perrings 1998). The elements of the transition matrix

are expected to be at least partly determined by investment in avoidance technologies and procedures.

Australia operates a national sentinel hive program, with sentinel hives located near major seaports being regularly monitored for the presence of exotic honey bee pests. Given the generic nature of these activities and difficulties in assigning specific attribution rates to the biosecurity effort directed at *V. destructor*, we do not formalize the relationship in the model. Instead, we use deterministic transitional probabilities, with p_{ab} specified as the initial arrival probability, and p_{aa} by an initial establishment probability. The remaining elements are $p_{ba} = (1 - p_{aa})$ and $p_{bb} = (1 - p_{ab})$.

Having successfully arrived in Australia, it is assumed *V. destructor* will become naturalized over time. No large country or region has successfully removed the mite once it has become established, and we are therefore compelled to assume that this would also be the case in the Australian context. This is not to say that there will be no action taken on behalf of beekeepers to protect their bee colonies from *V. destructor*. However, as a base scenario, we assume no large scale government eradication or containment campaign to bolster private control efforts. The benefits of localized control by beekeepers will mainly flow on to plant industries paying for commercial pollination services, and will do little to preserve the pollination service from the feral *A. mellifera* population (i.e., the number of commercial hives is very small in comparison to feral hives, and therefore subtracts little from the total susceptible number of colonies). Simulated damages therefore represent maximum losses over time, which are used in subsequent sections to indicate the returns to strategic pre-entry investments and a coordinated response to an outbreak.

Becoming a naturalized species in our model involves a site of introduction and a number of satellite sites that subsequently develop. After entry and establishment has occurred, the total damage attributable to any one of these sites j in time period t is estimated by

$$D_{jt} = \mathbf{P}(d_j A_{jt} N_{jt}) \tag{1}$$

where D_{jt} is the total damage inflicted by site j at time t ; d_j is the marginal damage cost of the pest in site j and time t ; A_{jt} is the area affected in site j at time t ; and N_{jt} is pest density within site j at time t .

Applying this damage function to *V. destructor* is difficult because the mite itself has no direct effect on pollination-reliant crops, so it is the indirect (or secondary) effects that we are modeling. We therefore assume that $d_j = (1 - r_j)y_j + c_j$, where r is a parameter defining the rate at which feral *A. mellifera* (which would have provided pollination services to that area of crop had they not been affected by *V. destructor*) can be substituted for by commercial pollination services while maintaining the same level of output (respectively) for host industries with a pre-invasion yield y . In their absence, costs c are incurred in the form of commercial

pollination services purchased by growers to replace the service from feral *A. mellifera*. Assuming that each site involves a single crop, r_j varies from site to site depending on the relative dependence on feral *A. mellifera* pollination. We assume that at the margin, commercial and feral bees are perfect substitutes (i.e., the first commercial honey bee introduced to an area is just as effective at providing pollination services as the last feral honey bee was before it fell victim to *V. destructor*), but that $r_j < 1$ (see Table 1). In the remainder of the paper, we refer to r as the marginal rate of pollinator substitution between feral and commercial bee pollination services.

After an arrival event, the spread of the outbreak to other susceptible areas is modeled as a logistic function. The area occupied by the population in one site j expands according to

$$A_j = \frac{A_{\max}}{1 + \left(\frac{A_{\max}}{A_{\min}} - 1\right) e^{-gt}} \tag{2}$$

where A_{\max} is the maximum area occupied by the pest (estimated by the total area of affected crop); A_{\min} is the area affected by the pest immediately upon establishment (assumed as 1 acre of affected crop [SI conversion: 1 acre = 0.4 ha]); g is the population growth rate; and t is the number of time periods after arrival.

We assume that in each unit of area affected by the expanding *V. destructor* population, the local population density (N) within site j also grows logistically to the carrying capacity of the environment such that

$$N_j = \frac{K}{1 + \left(\frac{K}{N_{\min}} - 1\right) e^{-gt}} \tag{3}$$

where K is the carrying capacity, or maximum density of infestation attainable per unit of area; and N_{\min} is the pest density immediately upon establishment.

The total number of component sites (s) making up an outbreak at any point in time is determined as

$$s_t = 1 + \frac{S_{\max}}{1 + \left(\frac{S_{\max}}{S_{\min}} - 1\right) e^{-\mu A_{t-1}}} \tag{4}$$

where S_{\max} is the maximum attainable number of satellite sites; S_{\min} is the minimum number of satellite sites; and μ is the intrinsic rate of satellite generation. Thus, as the total area affected by mites increases and the mite population density within that area increases, so too does the likelihood of a random satellite outbreak some distance from the original site.

Spread area, population density, and the number of sites can now be combined with the probability of entry and establishment in an expression of probability-weighted, or expected, damage over time. Assuming a discount rate α , the present value of expected damage

TABLE 1. Crop statistics, production cost increases, and marginal rate of pollinator substitution.

| Crop | Total area (ha)† | Gross value (US\$)† | Pollinator reliance (%)‡ | Additional hives (no./ha)§ | Marginal rate of pollinator substitution |
|----------------|------------------|---------------------|--------------------------|----------------------------|--|
| Almond | 4430 | \$31 380 715 | 100 | 2–5 | 0.7–0.9 |
| Apple | 13 260 | \$284 386 310 | 90 | 2 | 0.8–1.0 |
| Apricot | 1085 | \$23 664 145 | 70 | 1–2 | 0.9–1.0 |
| Avocado | 4000 | \$59 700 005 | 100 | 2 | 0.7–0.9 |
| Blueberry | 510 | \$20 157 025 | 100 | 1–2 | 0.7–0.9 |
| Canola | 1 909 730 | \$1 129 200 045 | 15 | 0 | 0.95–1.0 |
| Cherry | 1270 | \$32 184 430 | 90 | 1–2 | 0.8–1.0 |
| Cucumber | 1205 | \$12 422 140 | 100 | 1–2 | 0.8–1.0 |
| Field pea | 422 675 | \$74 217 510 | 50 | 0 | 0.9–1.0 |
| Lemon and lime | 1785 | \$18 428 350 | 20 | 0.5 | 0.95–1.0 |
| Lupine | 1 347 180 | \$205 052 935 | 10 | 0 | 0.95–1.0 |
| Macadamia nut | 14 000 | \$38 080 795 | 90 | 2–5 | 0.8–1.0 |
| Mandarin | 4895 | \$64 840 715 | 30 | 0.5 | 0.95–1.0 |
| Mango | 2650 | \$758 870 670 | 50 | 2 | 0.9–1.0 |
| Nectarine | 985 | \$86 070 740 | 60 | 1–2 | 0.9–1.0 |
| Orange | 30 560 | \$223 799 350 | 30 | 0.5 | 0.95–1.0 |
| Peach | 1885 | \$63 816 890 | 60 | 1–2 | 0.9–1.0 |
| Pear | 3025 | \$79 798 405 | 50 | 2 | 0.9–1.0 |
| Plum | 835 | \$33 212 615 | 70 | 1–2 | 0.9–1.0 |
| Pumpkin | 8995 | \$44 909 400 | 90 | 1–2 | 0.8–1.0 |
| Rockmelon | 3940 | \$78 281 210 | 100 | 1–2 | 0.8–1.0 |
| Strawberry | 905 | \$113 371 335 | 40 | 0 | 0.9–1.0 |
| Sunflower | 161 545 | \$38 172 960 | 100 | 2–5 | 0.7–0.9 |
| Watermelon | 4950 | \$51 143 565 | 100 | 1–2 | 0.8–1.0 |
| Zucchini | 1955 | \$24 234 590 | 100 | 1–2 | 0.8–1.0 |

† Annual value based on 5-year average (Australian Bureau of Statistics 2004).

‡ Based on pollinator reliance figures in Free (1993) and Cunningham et al. (2002).

§ Based in part on Ministry of Agriculture and Forestry (2000).

after n time periods $[PV(ED_n)]$ is

$$PV(ED_n) = \sum_{t=0}^n (1 + \alpha)^{-t} \sum_{j=1}^{s_t} D_{jt}. \quad (5)$$

This expression provides us with a probability-weighted estimate of invasion-induced revenue losses over time, and therefore provides an indication of its economic significance. It is *not* a measure of what damage will be inflicted by a species if it is introduced to a region tomorrow. Rather, it provides a measure of expected damage, taking into account uncertainty in the time of arrival and change in abundance and distribution over time after arrival.

Probability of entry and establishment

It is difficult to accurately estimate probability of entry and establishment; the openness of an economy, composition of trade flows, culture, regulatory regimes, the importance of agriculture, forestry, and tourism, and habitat fragmentation, conversion, and disturbance can all influence invasion probability and success (Williamson 1996, Perrings et al. 2002, Levine and D’Antonio 2003). With the exception of well-known, high-profile invasive species, quantitative arrival estimates are difficult to come by and often are poorly specified. We use an estimate put forward in Cook (2003) for the likelihood of *V. destructor* arrival, which in turn uses Australian government guidelines for risk analysis (Biosecurity Australia 2001) to represent a combined probability of entry and establishment using a uniform

distribution with a minimum value of 0.2 and a maximum value of 0.7 (i.e., Uniform (0.2, 0.7)).

Cost increments and the marginal rate of pollinator substitution

We considered 25 crops for which there are estimates of yield gains attributable to insect pollination services (Free 1993, Cunningham et al. 2002). We placed affected crops in one of four categories for the marginal rate of pollinator substitution (0.7–0.9, 0.8–1.0, 0.9–1.0, and 0.95–1.0), the choice of which was dependent on both pollinator reliance and the number of additional hives required to replace feral *A. mellifera* within the crop concerned (see Table 1).

All resultant future costs were discounted. Discounting was used because a dollar available for investment in the present is more valuable than a dollar that will not become available until a later period. The future dollar has an opportunity cost associated with it, i.e., investment opportunities we have had to forgo while we were waiting for it to become available for spending. The problem is that it is difficult to identify exactly what return could have been earned on those forgone investments. In the absence of definitive information on opportunity costs relevant to a specific project like *V. destructor* exclusion, we again cite Australian government guidelines, which recommend a standard discount rate of 8%. This consists of a margin of 3% on top of a real risk-free rate of 5% (Department of Finance 1991). We revised this risk-free rate, which applies to streams of

TABLE 2. Biological parameters.

| Parameter | Definition | Assumed parameter value |
|--------------------|---|--|
| A_{\min} (acres) | area affected by pest at establishment | PERT (10, 30, 50) |
| A_{\max} (acres) | maximum area occupied by pest | 10 042 475 (Australian Bureau of Statistics 2004) |
| g | population growth rate | PERT (0.20, 0.35, 0.50) (Harbo and Harris 2001)† |
| N_{\min} | pest density at establishment | PERT (5.0%, 7.5%, 10.0%) |
| $K(N_{\max})$ | carrying capacity at maximum density of infestation | PERT (70%, 85%, 100%) |
| S_{\max} | maximum attainable no. satellite sites | PERT (30, 40, 50) |
| S_{\min} | minimum no. satellite sites | PERT (0, 5, 10) |
| μ | intrinsic rate of satellite generation | PERT (1.0×10^{-3} , 5.95×10^{-3} , 1.0×10^{-2}) |

Note: The term “PERT” is an acronym for the Program Evaluation and Review Technique, used to form a special case of the beta distribution using lower boundary (minimum), modal (most likely), and upper boundary (maximum) parameters (Vose 2000).
† Approximation using information from Harbo and Harris (2001).

uncertain benefits adjusted for the cost of risk-bearing to risk-averse individuals, to 7% to reflect a precautionary attitude to radical ecosystem changes. Hence, our standard discount rate was set at 7% per annum.

Public, nonmarket effects of *V. destructor* exclusion, including existence and bequest values associated with *A. mellifera* displacement of native bee species (see Kremen et al. 2002), were not subtracted from our estimated cost increments.

We also omitted detailed changes in the beekeeping industry within Australia, but assumed that the price increment per unit of pollination service provided would increase. Although the industry is currently valued at around US\$48.8 million (Aus\$65.0 million) per year, paid pollination services represent a mere US\$2.5 million (Aus\$3.3 million) (Centre for International Economics 2005). *Varroa destructor* establishment and spread would probably lead to the expansion of this portion of the industry. Projections of likely price rises are not available. However, using the New Zealand experience as a benchmark, we assumed that the price per commercial hive will rise by between US\$55 and \$85 (Aus\$70–\$110) per hive as *V. destructor* establishes (Ministry of Agriculture and Forestry 2000). Resultant gains may be offset by restricted opportunities for exports of queen and packaged bees (particularly to the (potentially) large and expanding Canadian and U.S. markets, with the arrival of Africanized bees). There are no official statistics available on either current or projected values of these exports (Centre for International Economics 2005).

Other model parameters

Table 2 provides details of the biological parameters used. In each of the model iterations, one value from each probability distribution in the model is used. The average annual expected damage cost associated with the invasive species concerned is thus presented as a distribution.

RESULTS

Applying the model to *V. destructor* in Australia, we conservatively estimate that the mean of the distribution of expected pollination benefits preserved through

successful exclusion over a 30-year period is approximately US\$27.5 million (Aus\$36.7 million) per year. For convenience, a normal distribution was fitted to the histogram of expected exclusion benefits (Fig. 1). With 90% confidence intervals, we estimate that avoided damages will be US\$16.4–38.8 million (Aus\$21.3–50.5 million) per annum. Note that benefits are expressed as annual average damages avoided over a 30-year period, i.e., $(PV(ED_n)/n)$.

Our results imply that efforts to maintain area freedom from *V. destructor* can cost a substantial amount before they completely offset the benefits of exclusion. At best (i.e., taking the 5% confidence interval; Fig. 1), risk mitigation and management activities could cost up to US\$16.4 million per year (in real, or present-value terms) before the ratio of market benefits to costs becomes negative, and at worst, US\$38.8 million.

The choice of discount rate is crucial to the analysis, as Fig. 2 demonstrates. The economic significance of an invasive species like *V. destructor* depends on the total length of time over which simulations are carried out and at which specific point in time the measure of expected damage is taken. Changing the choice of discount rate from the base rate (7%) by small amounts has a large influence on results. Although a rate of 5% implies that the annual expected benefits of exclusion could be as high as US\$62.3 million projected out to 20 years (and US\$162.8 million by the 30-year mark), a rate double that implies that market benefits may only be US\$5.5 million (and may have fallen to a mere \$4.8 million by year 30).

DISCUSSION

To demonstrate how our model might be used to inform risk management decisions, let us look more closely at incursion response arrangements and how they relate to an invasive species like *Varroa destructor*. The importance of responding immediately to the presence of potentially harmful organisms as soon as they are detected requires governments and other bodies to establish an easily accessible funding mechanism for emergency action (Meyerson and Reaser 2002). This is accomplished in Australia via a cost-sharing agreement

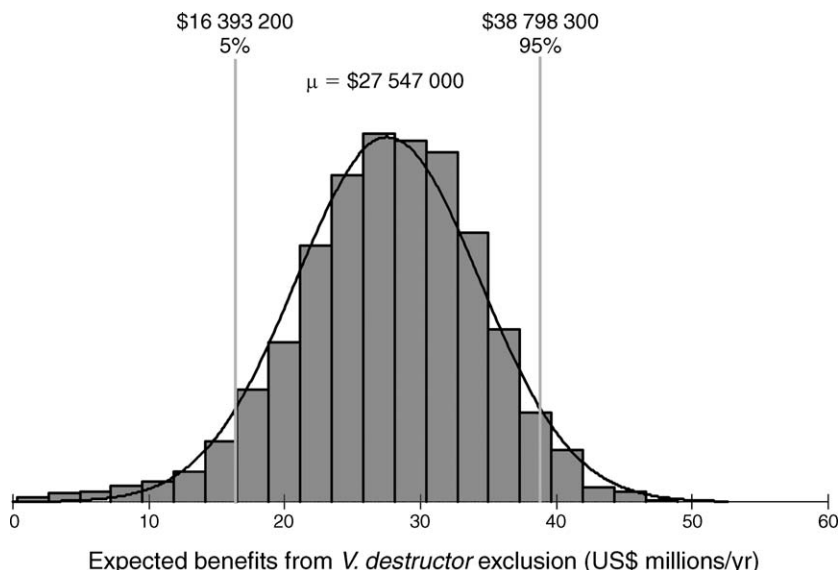


FIG. 1. Distribution of average annual expected benefits of *Varroa destructor* exclusion from Australia over a 30-year time period.

between the national government, state governments, and national industry groups, known as the Emergency Animal Disease Response Agreement, EADRA (Animal Health Australia 2002). In it, the proportion of public and private exclusion benefits for pests and diseases of quarantine significance are specified. Similar cost-share arrangements are being developed in the United States (National Invasive Species Council 2001). Under the EADRA, 20% of benefits produced through the exclusion of *V. destructor* from Australia are deemed to accrue to private entities and 80% to the public, under an agreement. In the event of an incursion, this means that 80% of any eradication cost is to be provided by government sources, while the remainder is to be funded by benefiting industries that are signatories to the agreement. In its existing form, the only EADRA signatory that stands to benefit from any future eradication campaign against *V. destructor* is the Australian apiculture industry.

Plant industries that stand to gain substantial pollination benefits if eradication is successful, as we have clearly demonstrated, are not signatories to this agreement. The omission of such a large flow-on effect in benefit-cost analyses places a strong negative bias on the calculated net benefits expected to result from the successful eradication of an outbreak, and may therefore lead to inadequate response efforts. Although more detailed assessments would be required, in practice, to assign specific benefit shares, we can use the model in its current form to provide an indication. Our estimates suggest that the largest industry-specific benefits are enjoyed by the sunflower industry, followed by the avocado, strawberry, and apple industries (Fig. 3).

Individual crop varieties, in fact, may be different in terms of pollinator dependence (Klein et al. 2007), and

development of new varieties is ongoing in many industries. Furthermore, the substitutability (or lack thereof) of one crop for another, and therefore the use of substitution as a management response to *V. destructor*, is difficult to determine. For these reasons, and because of the shortage of relevant pollination research, data are not available for a more accurate description of pollinator reliance. Nevertheless, of the 25 crops used in the simulations, 13 derived notional (theoretical) benefits from *V. destructor* freedom of over US\$751 500 (Aus\$1 million) per year.

The exclusion of nonmarket impacts from our model does not allow us to comment on the appropriateness of the 20/80 ratio between private and public eradication resources applied to *V. destructor*. Private benefits include the preservation of feral *A. mellifera* pollination services to national parks, public amenities, and household gardens, and consumer surplus losses resulting from

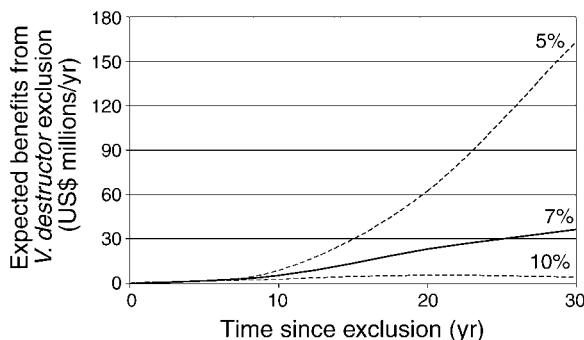


FIG. 2. Sensitivity of the expected exclusion benefits to changes in the discount rate. The three lines track the mean of expected exclusion benefits over 30 years discounted at 5%, 7%, and 10%, respectively (see Eq. 5).

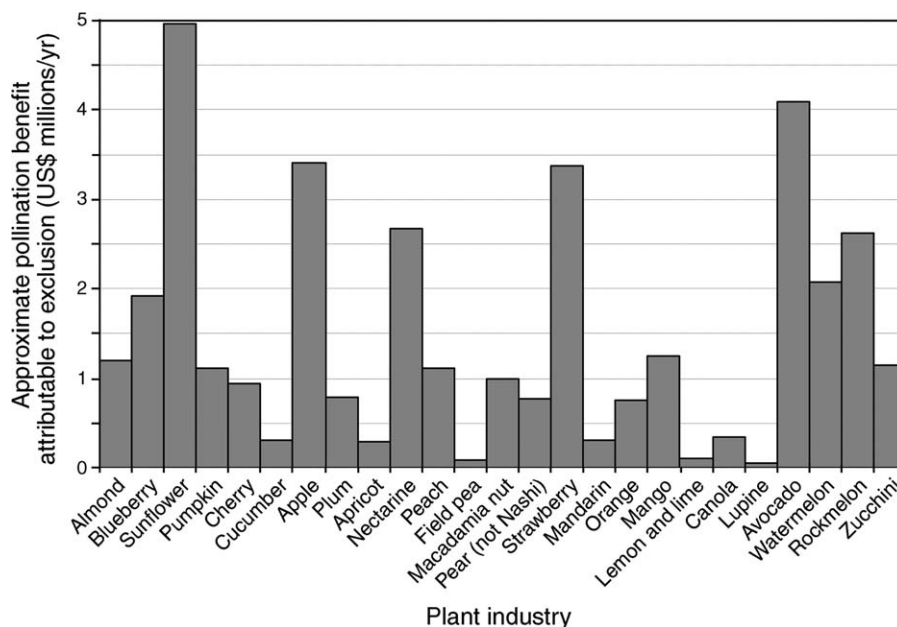


FIG. 3. Estimated private benefits accruing to plant industries as a result of *V. destructor* exclusion from Australia over 30 years.

increased pollination costs being passed on in commodity prices also. Offsetting these are public costs relating to: (1) the displacement of native pollinators, the extent of which is ambiguous (see Paton 1995, and Pyke and Balzer 1985); (2) the displacement of hollow-dependent native fauna by feral bee hives; and (3) human health effects related to injuries sustained as a result of bee stings. Indeed, between two and three fatalities attributable to bee stings are recorded every year in Australia (Harvey et al. 1984). Cook and Proctor (2007) present a method in which both quantitative and qualitative nonmarket and market information can be used to inform invasive species risk management decisions.

Sensitivity analysis of the model indicates that the variables with the greatest influence on results are the probability of arrival probability and the predicted rate of inter-hive spread of *V. destructor* in Australia. This suggests that the largest returns to investment in risk-reduction procedures or technologies involve targeting these parameters. To use a hypothetical example, if the probability of arrival can be lowered by 25% as a result of an additional US\$375 000 invested in pre-border and border security measures to avoid *V. destructor* arrivals, we predict a benefit:cost ratio (BCR) of 2.2:1. In the absence of a hazard rate function relating expenditure in prevention measures to the arrival probability, or to reduction in the rate of growth brought about by research and development activities to reduce spread, these predictions are questionable. In any event, practical limitations mean that it is unlikely that the arrival probability can be reduced below its current level. Indeed, further efforts to reduce it through pre-border and border security measures may, in fact,

achieve the opposite effect by raising compliance costs above the expected value of penalties for noncompliance, thereby increasing incentives to bypass regulated entry points (Department of Primary Industries and Energy 1988).

Other investment opportunities may lie in influencing different parameters important in the spread process, such as breeding resistance, to influence the maximum number of susceptible hives (A_{\max}). Resistance may be possible by breeding virus-resistant honey bees. Behavioral resistance may present another breeding option where grooming or hygiene habits of bee colonies restrict the destructive effects of *V. destructor* (Boecking and Spivak 1999). The most resistant honey bees will probably incorporate several resistant characteristics (Harris and Harbo 1999). If we assume hypothetically that we were to make a one-off investment of US\$750 000 and successfully find and reproduce *V. destructor*-resistant bees to the point at which A_{\max} falls by 25%, the benefits generated would be on the order of US\$5.6 million (i.e., BCR \sim 7.5:1). Research on host resistance continues, and our model has highlighted the potential benefits that may stem from it.

Host resistance research may be bolstered by efforts to reduce dependence on *A. mellifera* by encouraging populations of other wild pollinators through better landscape management (e.g., Cunningham et al. 2002, Klein et al. 2007) and the development of alternative managed pollinators (e.g., Hogendoorn et al. (2006)).

CONCLUSIONS

Using a stochastic impact simulation model with minimal information, we have been able to demonstrate

the substantial benefits of maintaining an important ecosystem service through the exclusion of an alien invasive species from a region. In total, we estimate that for every year Australia remains free of *Varroa destructor*, a group of 25 plant industries receives pollination benefits of US\$16.4–38.8 million per year. Specific information of this kind can inform management decisions affecting the provision of ecosystem services over time. Our results imply that large expenditures to either keep *V. destructor* out or to remove its lethal effects can be justified. We have also demonstrated that the private benefits of *V. destructor* exclusion extend well beyond the bee industry. As such, the current cost-sharing arrangements for incursion responses in Australia can be questioned on equity grounds.

The “blunt” nature of aggregate assessments of the value of ecosystem services makes it very difficult to use them in a management context. Decisions relating to who should pay to preserve a particular service within a particular region, for instance, cannot be informed by an interregional assessment of all ecosystem services. Although this broad assessment approach may be successful in improving general awareness of environmental benefits, we suggest that targeted case-specific bioeconomic models can be used to estimate economic impacts that are clearly linked to current management decisions and policy making.

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