

REVIEW AND SYNTHESIS

The economic benefits of invasive species management

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Abstract

1. Invasive species are known to cause significant negative impacts to ecosystems and to people.
2. In this paper, we outline the nature of these economic impacts, and then present a range of approaches for estimating the economic costs of invasive species (including impacts on biodiversity), and thus the benefits of management programmes. The importance of thinking clearly about the most appropriate context for valuation is stressed.
3. We provide examples of the application of non-market valuation approaches to invasive species management, and show how such methods can be used to measure public preferences over how control is undertaken.
4. We discuss some important problems in applying economic valuation methods in this context.

KEYWORDS

economic valuation, invasive species, non-market values

1 | INTRODUCTION

Invasive species are those introduced to a novel environment with negative ecological, economic or social impacts (Mooney, 2001). These negative impacts have been increasingly recognized in both the ecological and economic literatures as awareness of the impacts of invasive species grows, and as globalization increases the pathways and speed of invasions (Seebens et al., 2018; Smith et al., 2018). In this paper, we outline the approaches that economists take to measuring the costs of invasive species, including both commercially valued losses and “non-market” effects, while noting that economic benefits arise from non-native species in many instances. We also discuss some of the problems of applying economic valuation approaches to invasives.

Our understanding of the effects of invasive species is greatest for those that have impacts on agriculture and forestry (Vilà et al., 2010). However, the potential consequences of invasive species are wide ranging across ecosystems (Pejchar & Mooney, 2009). Invasive species

damage food production (Engeman et al., 2010) and can act as disease vectors (Medlock & Leach, 2015). Due to the complex nature of ecosystems it is likely that we do not yet understand their full impacts.

From a number of perspectives, controlling invasive species is becoming increasingly important for society. However, control often incurs high costs (Martins et al., 2006), and may be met with social opposition (Sheremet, Healey, Quine, & Hanley, 2017), particularly where invasive species have acquired cultural values (Roberts, Cresswell, & Hanley, 2018). Moreover, it has been argued that the introduction of some non-native species provides ecological benefits, for example in terms of providing shelter and food resources for native species (Schlaepfer, Sax, & Olden, 2011). When control efforts are unsuccessful, and/or where the damages associated with the invasion are low relative to the costs of control, then it may be socially desirable to abandon control measures and instead manage the resulting damage (Rolfe & Windle, 2014).

In this paper, we take the economic benefits of invasive species management to be equal to the avoided costs of damages from

invasives, were control not to be implemented. That is, one can contrast the stream of economic values which results from a business-as-usual or no intervention scenario, and compare this with the stream of economic values resulting when a package of interventions to manage the invasive species is in place. The size of benefits from control interventions thus depends on the speed of spread for new invaders, or area of invasion for established populations; the damages per “unit” (e.g. per possum, per infected km²); and how many people are affected by these damages. The **net** benefits of a management programme, in contrast, are equal to the value of avoided damage minus the costs of control, including any negative side effects of invasive species control (e.g. release of an invasive plant species from grazing pressure (Tye, Atkinson, & Carrión, 2007), or accidental poisoning of native (Lloyd & Mcqueen, 2000) or domestic species (Goh, Fearnside, Heller, & Malikides, 2005), and any benefits forgone that were provided by the invasive species (e.g. fuelwood production or hunting opportunities). Net benefits of interventions therefore also depend on the effectiveness of control options, when and where such options are implemented, and for how long they are implemented, since this will determine the time period over which (discounted) benefits and costs of control are added up (Figure 1).

Given the unpredictable nature of the side effects of invasive species control, incorporation of these costs is challenging. In some cases, the control efforts are able to learn from previous control. The release of blackberry *Rubus niveus* from grazing pressure following goat eradication in the Galapagos has informed future herbivore control efforts in the region (Carrion, Donlan, Campbell, Lavoie, & Cruz, 2011). Similarly a long history of the use of 1,080 poison for mammal control in New Zealand and Australia means that impacts on non-target species can be modelled before poisoning begins (Goh et al., 2005; Lloyd & Mcqueen, 2000). Nonetheless, side effects remain uncertain and hard to predict, and therefore are a source of uncertainty in estimating net benefits of invasive species control.

In this paper, we do not review evidence of the effectiveness of control options (e.g. Leppanen et al., 2019; Shine & Doody, 2011), and say little about costs. Instead, we focus on the value of avoided damages, and on public preferences for *how* control is undertaken. Indeed, there are many issues which economists have contributed to on this subject which we do not cover. We give no attention to the economically optimal level of management effort, the timing of control actions (Sims & Finnoff, 2013) or the use of economics in modelling invasive species spread. For an excellent overview of many of these issues, see Epanchin-Niell (2017). We also do not consider the question of how much resources to invest in biosecurity measures to try to prevent potentially invasive species from entering a country (Finnoff, Shogren, Leung, & Lodge, 2007; Florec, Sadler, White, & Dominiak, 2013; Rout, Moore, Possingham, & McCarthy, 2011). Finally, we do not provide a comprehensive description of the general principles of the economic valuation of environmental change, but direct readers looking for further guidance to Hanley and Barbier (2009).

2 | A GROWING PROBLEM?

The total number of invasive species is increasing worldwide (Huang, Haack, & Zhang, 2011; Seebens et al., 2017, 2018; Smith et al., 2018). The number of new invasions, as well as the number of individual species recognized as invasive, has increased steadily since 1800, with an increased rate of introduction after 1950 (Seebens et al., 2017, 2018). This rise is often linked to the expansion of global trade, specialization in production and increased connections to previously isolated locations (Seebens et al., 2018). Climate change also opens up new pathways for introduction and for range expansion of already introduced species. For example, in China, USA and UK, the number of invertebrate pests has increased with rising mean temperatures even after accounting for increased trade (Huang et al., 2011), while

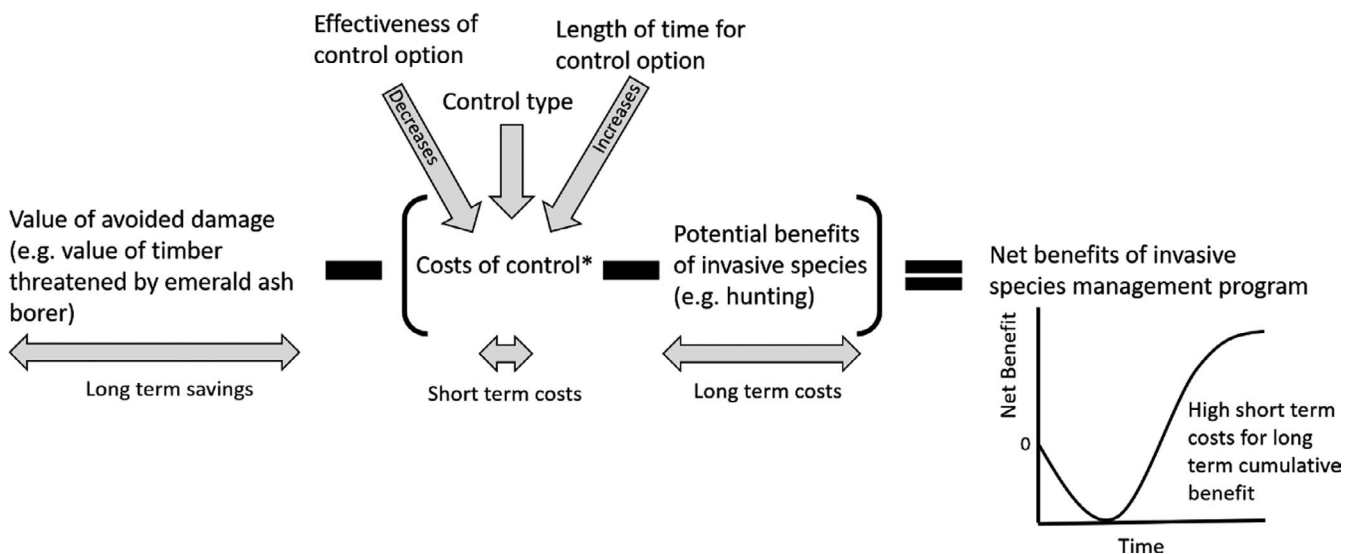


FIGURE 1 Calculation of net benefit of invasive species management programmes. A full economic evaluation of control options would discount the stream of predicted benefits and costs of control over time, using an appropriate social rate of discount (see Hanley & Barbier, 2009)

increased temperatures in Europe have led to the establishment of mosquito species and associated vector-borne diseases (Medlock & Leach, 2015). Although the trends for increasing numbers of invasive species seem widespread, it is important to account for increased effort in identifying invasive species which increases the probability of detection (Costello & Solow, 2003).

Invasive species are considered a key threat to the conservation of biodiversity. As Doherty, Glen, Nimmo, Ritchie and Dickman (2016) write, “*Thirty species of invasive predator are implicated in the extinction or endangerment of 738 vertebrate species, collectively contributing to 58% of all bird, mammal, and reptile extinctions.*” For many endangered species, invasives are the main factor driving populations towards local or global extinctions. Three examples from New Zealand are the kakapo *Strigops habroptilus*, kiwi *Apteryx* spp., and rock wren *Salpinctes obsoletus*, all threatened by predation of nests and young animals by stoats *Mustela erminea* (Robertson, Craig, Gardiner, & Graham, 2016; Weston, O'Donnell, van dam-Bates, & Monks, 2018). Efforts to safeguard the small remaining populations of these three birds thus depend on the success of efforts to remove the pressures from the invasive mustelid population by trapping or poisoning.

As invasive species increase, so do actions to control them. Costs of invasive species control are poorly reported, often contained within grey literature and limited in the species considered and in geographical scope (Brooke, Hilton, & Martins, 2007; Holmes et al., 2015; Iacona et al., 2018; Martins et al., 2006). Although cost reporting is scarce, those projects that do report costs generally show increased cost effectiveness over time, particularly where actions occur in the same geographic location (Carrion et al., 2011; Donlan & Wilcox, 2007; Martins et al., 2006). Despite increased efficiencies, the total costs of invasive species control are most likely increasing. This is largely due to control actions taking place in increasingly complex locations with multiple invasive species (Glen et al., 2013). As the number of invasive species and the complexities of control increase, so does the importance of prioritization of control actions. The most consistent predictor of costs is the size of control area, with larger areas having smaller per ha costs (Martins et al., 2006; Wenger et al., 2018), however this is not consistent across all studies (Holmes et al., 2015). Taxon (e.g. rodents vs. ungulates) can also have an impact, as can remoteness

(Holmes et al., 2015; Martins et al., 2006; Wenger et al., 2018). Prioritization of invasive species control needs to account for the costs of control, as well as the damages avoided, and any positive values of the invasive species itself (Cook, Thomas, Cunningham, Anderson, & De Barro, 2007; Donlan, Luque, & Wilcox, 2015; Roberts et al., 2018).

3 | TYPOLOGY OF ECONOMIC VALUES FROM INVASIVE SPECIES MANAGEMENT

The impacts of invasive species are associated with a range of costs and benefits, with many species having both positive and negative values depending on the context (Goodenough, 2010; Oreska & Aldridge, 2011; Shackleton, Shackleton, & Kull, 2018). Impacts vary through space and through time. Although it is widely recognized that invasive species have many potential impacts on human well-being, our understanding of these impacts from an ecological and economic point of view is very incomplete (Vilà et al., 2010). In this section, we use an economics perspective to categorize the types of impacts arising from invasive species.

For an impact of an invasive species to be economically relevant, at least one of two possible cases must hold. The first is that there is some effect, positive or negative, on the well-being or *utility* of people. The second is that there is some effect, positive or negative, on the profits of firms due to the invasive species. In this paper, we term these two types of value *utility impacts* and *production effects*. Table 1 summarizes this classification system. Any given invasive species may bring about economic costs or benefits from its effects on utility and/or on profit. In the rest of this section, we give examples of these types of effect. We also discuss how a value can be assigned to such impacts, based on the perspective of the analyst.

Invasive species can have a **positive utility impact** on people through provision of resources (Shackleton et al., 2018). Common Pheasants *Phasianus colchicus* in the UK countryside provide a source of enjoyment for hunters. The species was introduced to the UK from China as a game species in the 18th century (The statistical Accounts of Scotland, 1791–1799). Large numbers of pheasants are now bred and released each year on UK sporting estates (Robertson,

TABLE 1 Typology of economic impacts of invasive species

| Classification | Example | Are the example impacts valued by the market? |
|--|---|---|
| Positive utility effects on human well-being | Increased recreational hunting opportunities due to introduction of deer in New Zealand | Partially, if land managers charge hunting fees |
| Negative utility effects on human well-being | Effects of an invasive species on native biodiversity, for example, effects of stoats on kiwi | No: the value of loss is not provided by a “market price of biodiversity” |
| Positive impacts on production | Use of non-native species (e.g. sitka spruce) in UK timber production | Yes: timber markets reflect the value of yield increases from using non-native species |
| Negative impacts on production | Effects of invasive pests and pathogens on timber yields from woodlands in UK | Yes: timber markets reflect the fall in output from damages due to invasive pathogens and pests |

1996). *Rhododendron ponticum* was first introduced to the UK as an ornamental plant, and has value for gardeners due to its easy care, evergreen nature and bright flowers. Although attitudes towards *R. ponticum* have changed in recent years due to its fast spread and ability to outcompete native plants, removal projects are often met with public resistance (Williamson, 2006).

Invasive species can also have **negative utility impacts**. As well as being a game bird, pheasants are hosts to UK vectors of Lyme disease, most commonly the sheep tick *Ixodes ricinus*. Because pheasants remain infectious for up to 10 weeks, the species acts as amplifiers for the disease in the environment, increasing risk of transmission to humans (Kurtenbach, Carey, Hoodless, Nuttall, & Randolph, 1998). The impact of invasive species as vectors of disease is predicted to increase as a result of climate change. Invasive mosquitos *Aedes* spp. have increased in mainland Europe, bringing with them chikungunya and dengue fever, with cases reported in Italy and France. Climate models predict that climate change will lead to increased potential for these species to invade the UK (Medlock & Leach, 2015).

Invasive species may also have impacts on the **production** of goods and services. For example, an invasive pathogen may reduce the quality of agricultural output, or reduce yield, or require the increased use of costly inputs such as pesticides to sustain production levels. As such, the invasion increases the costs of producing a given level of output, or alternately reduces the potential output from a given amount of resource input (e.g. per hectare of land). These are **negative production effects**. Such potential effects can be studied through the means of a production function for output, which includes the effect (positive or negative) of the invasive as desirable or undesirable inputs, alongside other inputs such as labour and land. The most commonly studied impacts of invasive species are indeed those on agriculture and forestry. Early back-of-the-envelope calculations for US agriculture show losses due to the effects of non-native weeds, pests and pathogens of around \$65 million (Pimentel, Zuniga, & Morrison, 2005), while invasive pests and pathogens such as emerald ash borer *Agrilus planipennis* and *Phytophthora ramorum* cause damage to commercial forestry (Freer-Smith & Webber, 2017). Invasive species can also disrupt services: in the UK, rail services spend money to control *R. ponticum* growing near to railway lines to prevent service disruption (Williamson, 2006). Zebra mussels *Dreissena polymorpha* growing on boat hulls reduce fuel efficiency, thus effecting shipping costs (Oreska & Aldridge, 2011). However, there has been great use in agriculture and forestry of non-native species as a way of increasing outputs, generating a **positive production effect**. Obvious examples are the introduction of non-native tree species to increase timber yields from forests, for example, *Eucalyptus* spp. introduced into South Africa (Forsyth, Richardson, Brown, & van Wilgen, 2004).

For both utility and production impacts, economists think about the valuation of these from mainly two perspectives, according to whose values or which decision-maker one is thinking about. The two typical perspectives are private and social values. Consider the case of an invasive pest which has negative effects on timber yields, but which also results in forests becoming of lower recreational value

and to have lower ability to sequester carbon dioxide from the atmosphere. From the perspective of the land owner, the private costs of this arrival are those which effect their own well-being or profit: for example the loss of timber sales, or the reduction in hunting permits on their land which can be sold. But there are a wider set of impacts which affect the well-being of wider society: the loss of recreation values to non-hunters, and the loss of carbon sequestration benefits meaning higher costs from alternative means to reduce net greenhouse gas emissions. Society's perspective on what benefits and costs are relevant is much wider than that of the private landowner.

Any economic analysis of damage costs should be clear about which perspective—private or social—is being considered. From the viewpoint of what actions the landowner takes to control the invasive species, then almost certainly only private costs are relevant to them in determining the extent of these actions, unless society is using a financial subsidy to reward landowners for taking the wider set of impacts into their decision making (Macpherson, Kleczkowski, Healey, & Hanley, 2017). On the other hand, cost-benefit analysis of public policy choice takes the social perspective on economic values (Hanley & Barbier, 2009).

Given that we have now been clear on what potential types of impact are relevant to economic valuation, and the importance of whose perspective on value we take in any analysis (private costs and benefits or the wider set of social costs and benefits), it is next important to think about how these impacts can be valued in monetary terms. The most useful distinction is between those impacts which can be reasonably measured using market prices, and those which require non-market valuation methods. By “reasonably measured”, we mean approaches which provide useful information to decision-makers and other stakeholders on the private or social costs, and private or social benefits, of an invasive species.

Markets are good at signalling the value of a large set of goods and services which might be affected by an invasive species. Market prices reflect the interaction of supply and demand, for example for timber, wheat or farmland. Supply curves provide information on the incremental cost of producing goods (e.g. timber from a forest), and how this cost varies across producers. Demand curves show how much buyers (consumers or other firms) are willing to pay for goods and services: that is, the value that they place on having such goods and services made available to them. Thus, the demand curve for timber shows how much potential buyers are willing to pay for this good, which reflects the value of the good to them. Market forces mean that prices move to equilibrate demand and supply, so that the market price at any point in time shows both the incremental cost of producing the good and its incremental value to buyers. This means that when markets work well, market prices provide valuable signals on both the private and the social costs or benefits of changes in output. Thus, one estimate of the economic costs to the UK of a 10% loss of agricultural output due to an invasive pest is the market value (price times quantity) of this change in output, less the costs of producing this crop. A better estimate using the same data would be to also consider the effects on prices paid by consumers and received

by producers, as well as the changes in output quantity. Note, however, that actually establishing and then identifying the size of the causal link between the introduction of a pest/pathogen and the effects on agricultural profits may be challenging. Moreover, farmers may change their management in response to an invasive species, so such simple calculations can give a misleading estimate of the true social costs of lost output. We need to factor in any such behavioural responses to changes in production risk in measuring economic damages.

Markets are thus a good basis for thinking about the private costs and benefits to landowners or land managers, or to other firms such as transport companies, of an invasive species. However, market prices do not send good signals about social costs and benefits when demand and supply curves do not reflect all of the costs associated with producing a good (e.g. when dairy farming leads to increased water pollution) and/or all of the benefits of producing the good (for instance, when people use forests for recreation as well as for timber production). In such cases, private costs and benefits as measured using market prices are no longer the same as social costs and benefits. Such circumstances extend to cases where markets are simply missing for certain benefits and costs, such as the impacts of the invasive species on native biodiversity. Economists refer to both of these instances as a case of “**market failure**,” and for more than 100 years have known that market failure implies that market prices no longer provide adequate information on the social costs or benefits of changes in output.

Indeed, market failure characterizes many of the situations in which invasive species generate impacts on human well-being. The spread of mosquito vectors of dengue fever brings about increases in morbidity which typically are not valued by markets: extra cases of dengue fever in Spain over the next 10 years can be partly valued although increases in treatment costs to the Spanish health services, but generate wider effects of people's well-being which such approaches undervalue. The introduction of stoats to New Zealand in the 1880s (to control rabbits) brought about significant and ongoing predation losses to ground-nesting birds such as the kakapo and takahe *Porphyrio hochstetteri* (King, 1984). These social costs are not priced well by markets. The introduction of possums *Trichosurus vulpecula* to New Zealand from Australia has led to increased bovine TB outbreaks in dairy cattle (the value of which is recognized by markets for cattle and for dairy products) and to ecological damages to native forest species such as rata *Metrosideros umbellata*, which go completely unvalued by the market (Department of Conservation, 2008).

Differences in perspective on values often translate into differences in incentives. For example, a dairy farmer in New Zealand might consider spending money to help fund a regional possum control programme since this would reduce the private costs they might expect to experience from higher risks of bovine TB transmission to their cows (since possums are a vector). Imagine that these avoided costs from bovine TB are equal to \$200,000 per year to the farmer. It would be irrational for the farmer to spend more than this on the control programme solely from the perspective of losses in farm profits. But possums also have negative effects on native plants and birds: the utility of local people might

well increase if these negative effects were reduced or avoided, if they care about, and thus are willing to pay for, the avoidance of losses to native biodiversity due to possums. Suppose the aggregate willingness to pay of the local population is \$100,000 per year. The social benefits of control are equal in this case to the sum of the avoided damages to biodiversity plus the avoided impacts on dairy production, worth \$300,000 per year. The private benefits are lower than this, at only \$200,000. This means that the degree of control that is preferred if we think about social benefits from avoided damages is greater than if we only think about the private benefits. Of course, the farmer may also care about wildlife on her land, but the key point here is that the difference in perspective leads to a difference in what we see as the economic benefits of a control strategy, and thus the likelihood that we would choose to implement a strategy of a given degree of stringency.

Many of the effects on utility from invasive species will not be adequately reflected by markets. The negative impacts of invasive species on native ecosystems or species are a good example. Bird, reptile and mammal populations worldwide are well known to be adversely impacted by domestic cats *Felis catus* (Woods, McDonald, & Har Ris, 2003). In Guam, the brown tree snake *Boiga irregularis* is credited with the devastation of native bird species due to nest predation (Pimentel et al., 2001). Ash dieback resulting from the introduced fungi *Hymenoscyphus fraxineus* has led to large reductions in the ash population within the UK (Freer-Smith & Webber, 2017), which leads to losses in well-being for people who enjoy walking in ash woodlands. The loss of ash trees *Fraxinus excelsior* also represents a loss of pollution sinks, linked with increases in human cardiovascular and respiratory mortality (Jones & McDermott, 2017). Loss of ash trees due to the emerald ash borer in the USA has been associated with a reduction in life satisfaction due to the loss of locally valued woodlands (Jones, 2017).

Because invasive species are often prolific within the introduced system, they can change the native ecosystem functioning. The introduction of purple loosestrife *Lythrum salicaria* and water hyacinth *Eichhornia crassipes* structurally changes wetlands in Europe and North America (Pimentel et al., 2001). These changes can have significant impacts on people who care about the ecological quality of waterbodies, or whose recreational experiences are diminished due to the adverse impacts of such invasives on swimming or boating opportunities. The introduction of feral pigs *Sus scrofa* into the California Channel Islands led to increases in native Golden Eagle *Aquila chrysaetos* populations. This in turn decreased native fox *Urocyon littoralis* populations through increased predation, leading to increases in native skunk *Spilogale gracilis amphiala* populations due to declines in predation by foxes (Roemer, Donlan, & Courchamp, 2002). This change in ecosystem dynamics away from the “natural” ecosystem may be seen negatively for those individuals who value the unique ecosystems of the California Channel Islands, however increases in Golden Eagle populations may also be valued by bird watchers.

When invasive species have social costs and benefits which are not well-reflected by market prices, then a means must be found of valuing these non-market values. Such “non-market valuation methods” are described in the next section.

TABLE 2 Methods for valuing the economic impacts of invasive species

| Method | Type of impact valued | Advantages | Disadvantages | Examples |
|---|---|---|---|---|
| Stated preference: choice experiment | Non-use; recreation; landscape; biodiversity; cultural heritage | Wide range of applications; hypothetical markets means can be used to value planned control measures or estimate costs of potential invasive species. Can value multiple attributes and compare preferences. | Hypothetical markets not based in real payments. High cognitive burden for participants. | Preferences for control of invasive tree diseases (Sheremet et al., 2017) |
| Stated preference: Contingent valuation | Non-use; recreation; landscape; biodiversity; cultural heritage | Wide range of applications. Hypothetical markets means can be used to value planned control measures or estimate costs of potential invasive species. Lower cognitive burden for participants than choice experiment. | Hypothetical markets not based in real payments. Good can (typically) only be valued as a whole, not by individual attributes. | Values of delayed arrival date of invasive species (McIntosh et al., 2010) |
| Revealed preference: Travel cost models | Recreation | Based in real behaviour. Where multiple alternative sites exist can value individual attributes. | Limited number of scenarios in which this can be applied. Relies on existing markets and presence of alternative sites. Underestimate of value as cannot account for non-use values. | Recreation impacted by feral herbivore grazing (Peh et al., 2015) |
| Revealed preference: Hedonic pricing | Housing markets mainly. | Based in real behaviour. Potential to value individual attributes. | Limited scenarios in which method can be applied, very rarely used in invasive species research. Underestimate of value as cannot account for non-use values. | Invasive aquatic plants impact on waterfront house prices (Zhang & Boyle, 2010) |
| Production function | Crop production; Production forests; Livestock; Human health | Based in market values, except for applications to human health. Sees the environment as an input to production, | Limited scenarios in which applicable. Cannot account for many effects on utility. | Impact of ragweed on costs of allergies (Richter et al., 2013) |

4 | METHODS FOR VALUING NON-MARKET IMPACTS

Since the mid-1960s, economists have built up a tool kit of methods for estimating non-market values attached to the environment. Initially developed in the context of national park planning, water quality enhancements and public forest management, these methods have now expanded to be able to value changes in a very wide range of environmental benefits and costs, from changes in urban air quality to the conservation of wetlands. All of these methods can be used to value the non-market impacts of invasive species. Non-market valuation methods are usually categorized into three types (Hanley & Barbier, 2009):

- Stated preference approaches
- Revealed preference approaches
- Production function methods

All are based on the notion of maximum willingness to pay (WTP) as a standard measure of the economic value of a good to

individuals, since in economics the value someone places on any good or service depends not just on their preferences, but also on how much they are willing to give up to obtain it. Table 2 summarizes the applicability of each of the methods described below to valuing the economic damages or benefits associated with invasive species. We note that economic value is only one aspect of “value” which is relevant for society in thinking about the benefits of invasive species management: multiple definitions and frameworks for values exist which are relevant to the problem (Chan, Satterfield, & Goldstein, 2012). However, this paper takes a solely economics perspective.

Stated and revealed preference approaches estimate the effects of environmental change on individual well-being (i.e. the direct effects on utility). **Stated preference** methods ask individuals to make choices between different levels of ecosystem or environmental quality and the cost of provision. In this way, people show the value they place on, for example, avoiding damage by possums to native forests, or funding a fire ant control programme. Two stated preference methods dominate the literature: contingent valuation and choice modelling (Hanley & Czajkowski, 2019). In **contingent**

valuation, people vote on whether they agree with a specific change in the provision of an environmental good (e.g. delaying invasive species arrival to maintain recreation opportunities in waterbodies) at a specific cost to them (e.g. a payment of \$48 to delay arrival for one year, McIntosh, Shogren, & Finnoff, 2010). In choice modelling, people make choices between different “bundles” of environmental goods—such as different measures for invasive species control—as a function of the attributes of this good (e.g. forest ownership, type of forest, control action) where one of these attributes comprises a cost of providing the good (e.g. an increase in local taxes). One of the attributes of the good over which people make choices could be populations or spread of an invasive species, or the impacts of a species on, for instance, forest quality. Attributes could be also used to describe the different potential components of a management plan (see the next section).

An example of a contingent valuation study of the benefits of invasive species control is McIntosh et al. (2010). Costly control measures may only delay the arrival of an invasive species in an area, rather than guarantee it will never arrive. A nation-wide survey of US households elicited their maximum WTP to delay the arrival of aquatic invasives such as fish (common carp *Cyprinus carpio*), molluscs (zebra mussels), crustaceans (rusty crayfish *Orconectes rusticus*) or water plants (Purple loosestrife) to inland water bodies in “regional” lakes and rivers in the USA, defined as places to which the respondent could drive in no more than 2 hours. Scenarios presented included delaying invasions by 1 year or 10 years for high or low levels of impact. Impacts were described in terms of effects on human health, the economy, recreation and navigation. Results showed that mean WTP to delay impacts by 10 years was five times greater than that to delay impacts by only 1 year. The main policy conclusion was that even short delays in arrival could generate significant economic benefits (around \$4–\$5.5 billion).

Another contingent valuation study is reported in Meldrum, Champ, and Bond (2013). The authors estimate the non-market benefits of managing white pine blister rust *Cronartium ribicola* in the US forests. This invasive fungus has caused significant ecological damages to high-elevation forests in Western North America and Quebec. The authors explore how respondent attitudes to forest protection and use affect their WTP to reduce the spread of this pathogen. Responses from a random sample of Western US households were used to elicit mean WTP estimates for a white pine blister rust control programme across a varying part (30%–70%) of 2 million acres of high-altitude pine forest. Just under half of respondents had a positive WTP for the programme. Mean WTP for the protection programme was around \$300 across all of those respondents with a positive WTP. The study shows how respondents’ attitudes to why an invasive species control programme should be implemented can be incorporated into models of the estimated benefits of such a programme (Meldrum et al., 2013).

Choice modelling asks respondents to make choices between different bundles of (environmental) attributes, allowing the researcher to infer the economic value which people place on each of these attributes (Hoyos, 2010). The method has been applied to

uncover aspects of a control programme most valued by citizens, such as the spatial targeting of control measures against invasive fire ants in Queensland (Rolfe & Windle, 2014). An example of a choice card (a card illustrating the features of goods to be compared) used in the study by Sheremet et al. (2017) on public WTP for programmes to counter invasive forest pests and pathogens is included in the Supporting Information. The choice modelling method has been extensively used in a wide range of environmental and conservation management contexts (e.g. Roberts, Hanley, & Cresswell, 2017).

Chakir, David, Gozlan, and Sangare (2016) apply the approach to quantify the impacts of the invasive Asian ladybird species *Harmonia axyridis* on French citizens. The Asian ladybird was deliberately introduced to France as a bio-control measure for aphid management in agriculture in the 1990s. It has since spread rapidly, and is associated with undesired negative impacts on native biodiversity, housing (due to overwintering in large numbers) and wine production (due to tainting). However, the presence of the Asian ladybird allows farmers to use lower volumes of pesticides for aphid control. Chakir et al included the following attributes in their experimental design, to represent the environmental management problem at hand (levels of the attributes are shown in parentheses):

- level of pesticides used in agriculture (status quo, a 3% increase over 5 years, a 3% decrease, according to populations of Asian ladybird present)
- population level of the native 2-spotted ladybird *Adalia bipunctata* which is adversely affected by the Asian ladybird (levels: not present in France; rare; abundant)
- damages to humans due to the presence of Asian ladybirds overwintering in houses (defined as % of housing affected varying from 1% to 15%)
- cost of an Asian ladybird control programme to the French taxpayer.

Results showed that across the 464 respondents who completed all of the choice tasks, people were willing to pay to protect the native 2-spot ladybird *A. bipunctata* and to reduce nuisance to householders; but they were also willing to pay to reduce pesticide use. This means that the French population would value a programme to protect/restore native biodiversity by reducing Asian ladybird populations, but would require compensation to make up for any increase in pesticide use for aphid control that this made necessary. Interestingly, WTP to remove the negative effects of Asian ladybirds was higher than the compensation needed to offset increases in pesticides. The results also show support for public research programmes into alternative ways of controlling this invasive species (Chakir et al., 2016).

Stated preference methods have the disadvantage that they are not based on actual payment for the good. However, they offer many advantages: widespread applicability and the ability to measure both non-use and use values (Hanley & Barbier, 2009). Considerable effort has been devoted to understanding how best to design such studies, how to minimize the extent of hypothetical market bias, and what kinds of econometric model are most appropriate given the nature of the data and the range of information processing and choice

processes that individuals may employ in responding. For example, the Sheremet et al. (2017) paper noted above focusses on the issue of heterogeneity in peoples' preferences for invasive control strategies in UK woodlands when analysing the stated choice data.

Revealed preference methods are based on actual behaviour rather than stated choices. The analyst searches for a "behavioural trail" in markets which are somehow related to the non-market environmental good of interest (Champ, Boyle, & Brown, 2003). **Travel cost models** use people's expenditures on outdoor recreation trips (e.g. mountaineering day trips, fishing trips, bird watching visits) to infer the demand for the natural resources (mountains, rivers, wetlands) which are the destinations of these trips. More relevantly, if there is a quality change at a given site (e.g. a loss of tree cover due to an invasive pest) or if a site is no longer available (e.g. if a suburban forest site is closed to recreationalists because of the presence of oak processionary moth caterpillars), then the economic losses of this closure of a site or due to a decline in site quality can be estimated. Similarly, the economic benefits of an increase in deer numbers which allows hunters in New Zealand to "consume" more days of hunting recreation could be valued using this approach.

It is more difficult to apply travel cost models than stated preference approaches to invasive species management, since one has to find a way of specifying a quantitative relationship between the abundance or spatial distribution of the invasive species, and recreational site quality. Then, one needs to find a relationship between the number of visits individuals make to the recreational site and this invasive-dependent site quality index. Perhaps due to these requirements, it is hard to find examples in the literature of fully developed travel cost model applications to the benefits of invasive species control. One partial analysis is presented in Peh et al. (2015). The authors study the effects of feral goats *Capra aegagrus hircus* and pigs *Sus scrofa domesticus* on ecosystem quality in the Centre Hills, Montserrat, and try to quantify the economic benefits of ongoing management of these feral species. Feral pigs and goats have adverse impacts on forest understorey through grazing which affects endemic and rare bird species such as the Montserrat oriole *Icterus ober* and facilitates the rapid spread of invasive non-native plants such as guava *Psidium guajava*. The authors consider three benefits of feral livestock control: enhanced carbon storage, nature-based tourism, and hunting. For nature-based tourism, a travel cost analysis was undertaken, based on interviews with overseas visitors. Spending on accommodation, meals and car rental was used as a measure of the total expenditure by overseas visitors. Sampled visitors were asked how likely it was that they would cease visiting if ecological quality in the Centre Hills declined (specifically, if "...the unique animals of Montserrat have disappeared"). These responses were used to produce an estimate of lost tourism values if feral livestock control were abandoned. However, as the authors note, this was an incomplete application of the travel cost approach, since only parts of travel costs were measured, while no allowance was given to tourists being willing to pay more than their actual travel costs to visit the reserve. More recently, Zipp, Lewis, Provencher, Zanden,

and Vander (2019) use travel cost models to look at spillover effects from milfoil invasions across a system of lakes in Wisconsin. Their results suggest that intervening early is the best strategy when "high value" lakes are impacted following an initial episode in a lower economic value lake (Zipp et al., 2019).

The second revealed preference method is the **hedonic pricing approach**. This examines the role of spatial and temporal variations in environmental quality in markets which are somehow related to the environmental good in question. An attribute-based approach is used to explain variations in prices in such markets, where one or more of the attributes is an environmental good (or bad). The most often used market is the housing market. Here, the analyst focusses on variations in house prices, based on the assumption that people are willing to spend extra on a house, all else being equal, to "buy" better local environmental quality. Thus, houses closer to urban green spaces, or with better air quality, or lower noise levels, will on average attract higher bids from house buyers than properties further away from green space, or with higher pollution levels, or which are in noisier neighbourhoods. Thus, behaviour in a market for a related good (housing, or agricultural or forest land) can be used to infer values placed by people acting in that related market on changes in a non-market environmental good such as urban air quality or landscape quality.

Hedonic price applications to measuring the benefits of invasive species control have been limited to instances where the invasive species affects house prices. For instance, the spread of an invasive aquatic plant can change the benefits of living at a lakeside location if this means that recreational opportunities are reduced. Zhang and Boyle (2010) study the relationship between house prices at lakeside locations in Vermont and the spread of Eurasian watermilfoil *Myriophyllum spicatum*. This plant spreads rapidly, crowding out native water plants and reducing recreational opportunities (swimming, fishing) as it forms dense mats. What is especially interesting about Zhang and Boyle's work is that they make use of two indexes for water quality: one based solely on the abundance of milfoil, and one based on the abundance of all water plants including milfoil. The paper is also unusual in being able to make use of property-specific values for milfoil and total aquatic plant abundance, rather than more spatially aggregated measures. Two alternative functional forms are used to represent the possible effects of milfoil abundance on house prices, quadratic and exponential. Results showed that whilst milfoil abundance on its own had no significant effects on house prices, total aquatic plant abundance (including milfoil) did. Marginal effects of increasing total aquatic plant coverage along a 6 point scale were computed, showing that, for example, reducing the coverage of aquatic plants from the highest level by one scale point would increase average house prices by around 20%. For a more recent hedonic price application of the same invasive species for a lake in Northern Idaho, see Liao et al. (2016).

Production function methods link invasive species population changes to impacts on commercial crops and livestock, or to human health outcomes. Figure 2 shows some of the possible linkages. The general idea is to evaluate how changes in environmental status or

ecological condition effects the production of some marketed good or on the “production” of a health status, which may be moderated by management choices. The right and left arrows show epidemiological models which translate the change in the “arrival” of an invasive pathogen, for instance, into its effects on commercially grown crops. Crop losses can be valued using market prices. For forests, the arrival and spread of the pathogen may change the optimal management of the forest in terms of the optimal rotation period and/or the optimal planting mix of species (Macpherson et al., 2017; MacPherson, Kleczkowski, Healey, & Hanley, 2018). Moreover, if we think about the potential irreversibility of certain invasive species control options (e.g. the introduction of a natural predator), and the likelihood that we will learn more about the epidemiology and impacts of the invasive over time, then real options models can be used to estimate the costs of acting too soon or too late (Sims & Finnoff, 2013).

For human health effects—for example, in terms of cases of dengue fever in a country due to the arrival of the mosquito *Aedes aegypti* which spreads the dengue virus—several valuation methods exist, including the use of stated preference methods to measure WTP for reducing disease risks, and the Costs of Illness approach, which sums medical system care costs and lost earnings due to sickness. Note that the Cost of Illness approach will often under-state the most people are willing to pay to avoid an episode of ill health, so that the method yields under-estimates of the economic costs of illness. Production function approaches can also be used, where the invasive species is a negative input to the “production” of a particular level of health or health status. Here, an epidemiological model links changes in the invasive species to changes in human health status; and then an economic valuation is placed on this change in health status. Richter et al. (2013) estimated the economic costs of allergic reactions arising from ragweed *Ambrosia artemisiifolia* throughout Austria and Bavaria, modelling costs under alternative climate scenarios for the period 2011–2050. To estimate the population affected by ragweed the authors first model the spread of ragweed under different climate scenarios. Under current climatic conditions, the mean annual cost to Austria and Bavaria of ragweed, in terms of health and productivity losses, is estimated to be €291 million Euro per year between 2011 and 2050, rising to €333–365 million per year under climate change scenarios. Because the cost estimates from this study are linked to modelled expansion of ragweed range, the models also predict that costs will increase over time, rising from €133 million in 2005 to €422 million in 2050 under current climate.

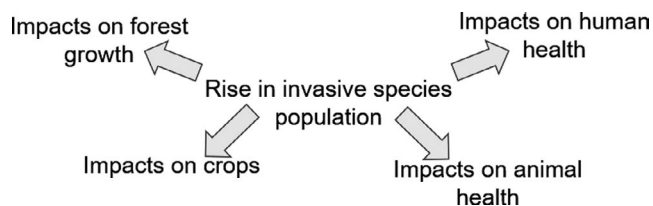


FIGURE 2 Link between invasive species population changes to impacts on commercial crops and livestock, or to human health outcomes, as used in production function approach

5 | ECONOMIC VALUES ASSOCIATED WITH HOW INVASIVE SPECIES ARE MANAGED

Multiple options are often available to managers to respond to an invasive species, once this species has arrived. Examples of damage reduction activities include lethal controls (e.g. poisoning), the fencing of vulnerable habitats, and felling or spraying of invasive pests and pathogens which affect forests. The public may well have preferences over these control options which should be taken into account in any Cost-Benefit Analysis of control measures (Bremner & Park, 2007; Sheremet et al., 2017). That is, if people would rather animals were not controlled by lethal means, then their WTP to avoid this measure being implemented is a non-market cost of the damage reduction activity which should be added to the financial costs of lethal programmes. Although there are a wide range of methods available to understand public preferences (see Hall, McVittie, & Moran, 2004 for an overview of methods with regards to general environmental management), we have focused only on WTP because this is best suited for inclusion in the cost-benefit analysis framework. Concerns over the acceptability of management measures do show up in a WTP framework, as the examples provided here show; but other value frameworks will give a wider view of public acceptability than the economic frame adopted here. Even if a full Cost-Benefit Analysis is not undertaken, one can take the perspective that damage reduction activities should be chosen with a view to somehow balancing their social acceptability with their ecological effectiveness and financial cost (Roberts et al., 2018).

There is evidence that the public care about which invasive control measures are used or proposed (Bremner & Park, 2007). Lethal control is an obvious example. Hanley, MacMillan, Patterson, and Wright (2003) used a choice experiment to show that the willingness to pay for a goose *Anser albifrons* and *Branta leucopsis* management programme on the island of Islay (Scotland) was significantly reduced if the control programme included shooting. This result was found for the Scottish general public and for visitors to the island. Interestingly the WTP of residents of Islay was not significantly reduced by the use of shooting (Hanley et al., 2003). For forest disease control, Sheremet et al. (2017) found that the UK public had a negative WTP for control options which consisted of either clear-felling infected forests, or which made use of chemical or biocide spraying. Jepson and Arakelyan (2017) found a negative attitude to the use of GM breeding methods as a response to ash die-back, and that this attitude varied significantly according to respondent's age and education, and according to where GM-modified ash trees were planted in the landscape (Jepson & Arakelyan, 2017). Finally, Fleischer, Shafir, and Mandelik (2013) used choice modelling to study the preferences of Israeli citizens for different control options designed to respond to invasion by the Dwarf honey bee *Apis florea*. The attributes used in the design were impacts on two native plant species (*Calotropic procera* and *Lupinus pilosus*); the nature of a pesticide-based control programme targeted at the dwarf honey bee; and

donations to a fund to pay for the programme. People were willing to pay between US\$6–\$17 per month for a control programme, but this declined for around 25% of the sample when a pesticide was used (Fleischer et al., 2013).

6 | DISCUSSION

In the preceding sections, we saw how a number of different tools are available for estimating the economic benefits of invasive species management. In this final section, a short discussion is provided on some of the main challenges in applying these methods in this specific context, again focussing on non-market impacts.

The first problem to note turns on the issue of how much we require people to know about an issue before “counting” their preferences as part of decision making. Clearly, the vast majority of the populace will not understand the complex web of factors determining the nature of species invasions, their impacts on ecosystems or production, or the nature of control options available. In such circumstances, how much weight should be given to the values “poorly informed” people place on control options? It is certainly useful to know something about how people’s WTP depends on what they know about the problem: Sheremet et al. (2017), for instance, show how people’s understanding of invasive forest pathogens is related to their WTP for different management measures. Bremner and Park (2007) find a strong association between knowledge and support for invasives control programmes. This problem reflects a much more general issue in environmental economics and cost-benefit analysis when we apply the principles of economic valuation to issues such as biodiversity decline, about many people will not know as much as experts (La Riviere et al., 2014). Welfare economics (that part of economics underlying cost-benefit analysis and valuation) state that the economic votes of everyone within the “relevant population” should count, no matter how much people know about the good in question. Thus, finding methods to help respondents understand the implications of invasive species management options before measuring their preferences might be viewed as a sensible approach in this regard, and economists have been investigating the ways in which deliberative mechanisms can be combined with economic valuation approaches in such situations (Lienhoop, Bartkowski, & Hansjurgens, 2015). However, these kinds of participatory approaches create aggregation problems, since now the values of observed subjects will likely differ substantially from the population from whom they are drawn.

Clearly, telling people more about the likely effects of an invasion, what contributes to its spread, or how the spread can be managed, will be helpful in terms of better public policy decision making and more effective management (e.g. for fire ants in Queensland: see <https://www.daf.qld.gov.au/business-priorities/plants/weeds-pest-animals-ants/invasive-ants/fire-ants>). But ultimately, what drives economic valuation is changes in “end-points” that make a difference to people’s well-being, whatever the nature of the complex

mechanism that delivers these changes in end points. So what people care about is changes in the nature of their recreational experience in a forest, not how an invasive species produces these changes.

The second issue to ponder is concerned with scientific uncertainty over the rate of spread of an invasive species, its impacts on ecosystems, and the effectiveness of control measures. Scientists will often be very unsure about these parameters, especially in the early stages on an invasion (Lodge et al., 2016). From a valuation perspective, taxpayers may well care about the nature and extent of such uncertainty, in terms of their willingness to pay to support a control strategy. Sheremet et al. (2017) included uncertainty over speed of spread, extent of damage and efficiency of control measures in their choice experiment on invasive forest pathogens, but found no significant effects of such uncertainty on public WTP for a control programme (although there was a statistically significant variation with regards to how much importance people attached to uncertainty over speed of spread). More generally, however, we know that taxpayers often care about the uncertainty attached to predicted environmental policy outcomes (Lundhede, Jacobson, Hanley, Strand, & Thorsen, 2015). Thus, it seems preferable to effectively communicate scientific uncertainty over invasive species management to households and firms when trying to estimate the benefits of control. Being able to quantify this uncertainty in a way in which ordinary people can understand is a key challenge for ecologists.

Third, there are issues around the treatment of irreversibility and the timing of actions. In many cases although the impacts of invasive species may be uncertain, they may also be irreversible (e.g. the extinction of an endemic species; Finnoff, McIntosh, Shogren, Sims, & Warziniack, 2010). In these cases it has been suggested that the most appropriate question may not be “how much is society willing to pay” but “how much can society afford to lose”. Acting to prevent species invasions, or quick action once an invasive species is discovered, can prevent high negative impacts and ensure that the widest range of options are available going forward. However, the costs of prevention of irreversible changes need to be considered. Moreover, if we can learn more about damage costs and about the effectiveness of control measures as time passes, then waiting itself generates an option value which should be considered in deciding when to act (Sims & Finnoff, 2013).

While we have focussed on the economic damage caused by invasive species, in many cases invasive species also have positive cultural or social values. Failing to account for such values can undermine control efforts (Estévez, Anderson, Pizarro, & Burgman, 2015). In Hawaii, 12% of the population were in favour of maintaining a feral cat population, rising to 50% of people involved in animal welfare organizations. The reasons for such opinion was related to enjoyment of seeing feral cats, and an intrinsic value of knowing feral cats persist, even if not seen (Lohr & Lepczyk, 2014). Similarly in Bonaire (an island in the Caribbean), positive public attitudes towards feral donkeys restricted the possible control measures available, as any lethal control programme would be met with high social resistance (Roberts et al., 2018). Many of these cultural

values associated with the presence of an invasive species could be hard to measure using the economic principle of willingness to pay, and might be better understood using alternative notions of “value” (Chan et al., 2012). Understanding the positive values associated with invasive species can be central to designing effective invasive species control measures, particularly where control is sanctioned by governing bodies who must respond to multiple competing agendas, and where invasive species control can become a politicised issue.

Finally, in a rapidly changing climate the very concept of invasive species becomes problematic. Huang et al. (2011) illustrate that increasing temperatures are associated with increases in invasive species in the UK, USA and China, whilst Medlock and Leach (2015) show the increased potential of mosquito invasion into the UK under climate change. Climate change is associated with changing ranges for many species (Aguilée, Raoul, Rousset, & Ronce, 2016; Giezendanner, Bertuzzo, Pasetto, Guisan, & Rinaldo, 2019; Parmesan, 2006). While controlling invasive species arising from climate change may be of high importance to safeguard native species, so too is enabling species to adapt their ranges to survive a changed climate. For species where natural range expansion is limited, such as by island size or due to other environmental barriers including urban areas or mountain ranges, translocation to novel environments may be required to ensure persistence (Braidwood, Taggart, Smith, & Andersen, 2018; Vitt, 2016), although this method of species conservation is not without controversy (Bucharova, 2017; Vitt, 2016). Appropriate measures to prevent invasive species, and control measures to tackle invasive species, must therefore allow for, and in some cases manage, natural range expansion. Which species people consider to be “native” and thus “worth saving” will likely influence the public acceptability of management measures in such cases (Lundhede et al., 2015).

7 | CONCLUSIONS

In this paper, we have illustrated the range of economic benefits from managing invasive species, including safeguarding biodiversity, reducing losses from forestry and agriculture, and improving ecosystem health. As invasive species and their impacts continue to increase, so does the need to develop appropriate policy and management responses. Recognizing the economic benefits of control provide vital information to policy makers and practitioners to prioritize invasive species control actions. Benefits of managing invasive species are not limited to those associated with market-valued goods such as crops, but should include increased exposure to disease and disruption to ecosystem service supply and impacts on biodiversity. We made clear the importance of thinking about the most appropriate context, whether private or social, in measuring the benefits of management, and in predicting whether private landowners apply sufficiently robust controls from the perspective of society. Decision making also needs to account for the impact of uncertainty over the outcomes associated with control of invasive species, and the potential irreversibility of control actions.

This paper contributes to consolidating an understanding of the economic benefits of invasive species control. What economic valuation currently demands of ecologists in this regard is simple to set out. These demands include being able to quantify the impacts of invasives on end-points which people care about, or end-points related to producer profits; and the extent to which specific management actions mediate such undesirable effects on production and utility. Given the fast-changing landscape of invasive species management, these demands are certainly not trivial.

However, this is only half of the equation, as policy makers and practitioners also need to account for the costs and effectiveness of control when making management decisions. We would direct readers to the growing literature on the effectiveness of invasive species control options (e.g. IUCN, 2017; Simberloff, Keitt, & Pickett, 2018; Simberloff, 2001) and encourage full consideration of costs in combination with the economic benefits of control we present here. Control of established invasive species is also only one tool in reducing impacts of invasive species, as enhanced biosecurity could arguably have more benefits than control efforts following establishment (Rout et al., 2011). However, biosecurity measures also have potentially high costs, and trade-offs related to risk of invasion, spread, and potential severity of damage must be considered alongside costs (Epanchin-Niell & Liebhold, 2015; Rout, Moore, & Mccarthy, 2014).

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CONFLICT OF INTEREST

None declared.

AUTHORS' CONTRIBUTIONS

Both authors shared equally in planning and writing the paper.

DATA ACCESSIBILITY

No data were used in writing this paper.

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