J. Verbr. Lebensm. (2011) 6 (Suppl 1):S3–S13
DOI 10.1007/s00003-011-0689-0

Journal für Verbraucherschutz und Lebensmittelsicherheit
Journal of Consumer Protection and Food Safety

CONFERENCE PROCEEDINGS "DECISION MAKING AND SCIENCE"

Prioritising biosecurity investment between agricultural and environmental systems

David C. Cook · Rob W. Fraser · Jeffrey K. Waage · Matthew B. Thomas

Published online: 3 May 2011

Abstract This paper is motivated by the observation that there is a difference between the time paths of damage valuations for invasions which affect agricultural compared with environmental systems. In particular, unlike agricultural systems, social valuation of an environmental system is likely to be exponentially positively related to the extent of its deterioration. This paper explores the implications of this difference in determining biosecurity investment priorities where criteria for decision-making are relatively narrow. It is concluded that because of this difference an environmental system will often not be prioritised for such protection over an agricultural system even though its ultimate social value exceeds that of the agricultural system. For this reason a broader set of decision criteria are needed that enable decision-makers to learn more about the context of biosecurity investment decisions.

Keywords Biosecurity · Invasive species

Conference Proceedings: "Decision Making and Science—The Balancing of Risk Based Decisions that Influence Sustainability of Agricultural Production". 7th and 8th October 2010 in Berlin, Germany. Sponsored by the OECD Co-operative Research Programme.

D. C. Cook (✉)
CSIRO Ecosystem Sciences, GPO Box 1700, Canberra, ACT 2601, Australia
e-mail: david.c.cook@csiro.au

D. C. Cook
Cooperative Research Centre for National Plant Biosecurity, LPO Box 5012, Bruce, ACT 2617, Australia

R. W. Fraser
Department of Economics, The University of Kent, Canterbury CT2 7NZ, UK

J. K. Waage
London International Development Centre, 36 Gordon Square, London WC1H 0PD, UK

M. B. Thomas
Department of Entomology and Center for Infectious Disease Dynamics, Pennsylvania State University, University Park, PA 16802, USA

1 Introduction

Biological invasions have long had important economic implications for agriculture (Office of Technology Assessment 1993; Pimentel et al. 2000, 2005). Alien insect pests of crops, plant and animal diseases and weeds can cause outbreaks that spread and reduce agricultural production over broad areas (Lonsdale 1994; Mumford et al. 2001; Stansbury and Pretorius 2001; Wittwer et al. 2005). Regulatory institutions have been developed to prevent introductions of these "agricultural invasives", backed up by tools like chemical pesticides and biological control, for their eradication or control (GATT 1994). Nonetheless, these problems remain considerable, with economic costs arising from losses of production, costs of control and losses to trade for invasive species which are banned under international agreements (Fraser et al. 2006).

In the 1990s, research by ecologists revealed the dramatic potential environmental impacts of invasive alien plants, animals, and micro-organisms (Williamson 1996), and this led to the international agreement, embodied in the Convention on Biological Diversity (1991), that countries should prevent, eradicate or control species which threaten local species, habitats or ecosystems. This gives recognition to the dual effects of environmental bioinvasions—reduction of native biodiversity (including the extinction of native species), and the disruption of ecosystem service, e.g. when an invasive alien tree disrupts fire regimes and water and nutrient cycling in native grasslands.

While agricultural and environmental systems both face growing threats from Invasive Alien Species (IAS),[1] government responses are often profoundly different between these two sectors because of the need to prioritise public expenditure. With rising populations and increasing demands on government to provide public goods and correct market failures, the opportunity costs of money devoted to IAS control are also increasing. Moreover, the relative visibility of agricultural damage attributable to IAS compared to environmental damage often (but not always) translates into greater investment in agricultural protection than environmental. Certainly, institutional development has been most evident in the agricultural arena rather than the environmental arena (Agtrans Research and Dawson 2005).

Non-agricultural risks tend to receive less attention. A lack of credible quantified biological risk assessments has had severe consequences as risk assessment has developed into a dominant force in resource allocation within regulatory circles (Simberloff 2006). For example, following a Canadian request to access Australia's salmon market in 1994, Australian quarantine authorities commissioned the Australian Bureau of Agricultural and Resource Economics to prepare an economic analysis indicating potential economic damages that could result from diseases considered an importation risk (McKelvie et al. 1994). No environmental risks were considered. Instead, the threat of sizeable damage to commercial fisheries that might be caused by the diseases Furunculosis and Infectious Haematopoietic Necrosis were used as the basis for the Australian government to refuse Canada's request, which prompted Canada to take the matter to the World Trade Organization's Dispute Settlement Body. After formal consultations failed to resolve the issue, a dispute resolution panel was formed which subsequently ruled against Australia's salmon import ban. Upon appeal by the Australian government on behalf of its aquaculture industries, The Appellate Body upheld the panel's decision and Canada began exporting salmon to Australia in May 2000 (Andrée 2000). Throughout this long-running case, Australia continued to import aquarium fish and herring bait, which carry far greater disease risks than salmon to both commercial fisheries and native marine and estuarine ecosystems (Booth 2008). These risks were not subjected to the same amount of scrutiny, and did not garner the same sort of political and social attention in Australia as the risks to commercial fisheries.

While some international risk assessment processes incorporate environmental and social risks associated with IAS (e.g. Pheloung et al. 1999; Baker et al. 2008), a lack of quantitative information about their extent can mean deferral to economic risk assessment. While measures of economic risk might not necessarily be derived from detailed simulation models of spread and impact, annual gross value and volume statistics are readily available to indicate the relative size of agricultural assets placed at risk by an IAS (e.g. in the case of Australian industries, see AG-WEST Trade and Development 2003; ABS 2004; ABARE 2006, 2010). Environmental and social assets do not have the same easily-expressed annual values. It follows that a quantitative economic criterion tends to carry greater weight in terms of influence over strategic investments in IAS risk mitigation since it can be readily informed with data. Indeed, in the case of agricultural R&D organisations prioritising investment opportunities (e.g. Cook et al. 2010a) or State departments of agriculture (e.g. Cook 2003), it can be the only criterion considered.

The merger of environmental and agricultural ministries in some countries, and the agreement to coordinate international activities between, for instance, the International Plant Protection Convention and the Convention on Biological Diversity, create opportunities for a more cohesive approach to risk analysis (Bowornwathana 1996). Ultimately, policy makers often need to make difficult decisions between management actions to prevent, eradicate and control IAS threatening agriculture and/or the environment. These may be different threats, or the same, e.g. an invasive weed that both affects grassland ecosystem services and displaces grazing livestock. These sorts of decisions require the input of economists with necessary tools like benefit cost analysis and cost effectiveness analysis to help to identify expenditure options with the highest expected social welfare gains or lowest management costs over time.

[1] The term IAS applies when the abundance and distribution of a non-native (or alien) organism exceeds a defined and accepted environmental standard, resulting in a net negative effect on social welfare (Cook et al. 2010b).

A pressing issue for economists dealing with natural systems involves the questionable reliability of prices as measures of willingness to pay (or willingness to accept). The average consumer or producer does not possess sufficient data, expertise or inclination to factor in the potential invasive species damage costs that might result from a consignment of imported commodity being biologically contaminated. For any rational, profit maximising individual entering into a contract to supply or purchase such a commodity on an international market, it is impossible to account for every eventuality within the contract itself given the uncertainty surrounding the distribution of expected profits (Scholz and Stiftel 2005). While the impacts of a particular species on an agricultural industry poses no particular methodological problems (beyond determining expected supply curve shifts), non-market impacts are more complex.

The challenge associated with eliciting values for environmental flow-on effects is well documented. The large growth in the literature following the Exxon Valdez disaster is without precedent (Adamowicz 2004), but several problems with stated and revealed valuation techniques persist. It is difficult to understand and appreciate the willingness of an economic agent to pay to protect an environmental good (or to guard against changes in its wellbeing) without sociological information involved in that agent's decision-making process (Cook and Fraser 2008). The income elasticities associated with environmental goods are thought to be significantly positive, implying income has a relatively large influence on a person's willingness to pay to protect the environment (Whitby 2000). Non-use values for environmental amenities are also important. While an individual may lack financial incentives to invest in activities promoting the protection of ecosystems, their utility function may be partially dependent on environmental variables. As a result they may be prepared to forgo other consumption possibilities in order to gain utility from merely knowing the environment or a component of an environmental system remains in a favourable state.

However, for species invading the environment, where the impact will probably be on biodiversity or ecosystem services, it is likely that a proportionately greater amount of spread and damage must be incurred before a negative effect is perceived. This problem has been identified in the context of environmental valuation, with researchers attempting to elicit values which are contingent on the state of environmental deterioration of habitats or species. For example, Blamey et al. (2000) asked survey respondents to distinguish between "non-threatened" and "endangered" species in eliciting valuations, while Hanley et al. (2003) evaluated respondents' views on protecting "all goose species" compared with "endangered goose species". In such cases the findings support an exponential dependence of social valuations of environmental goods on the extent of damage to those goods. It follows that the social valuation of an environmental good is likely to be not just positively but also exponentially related to the time path of its deterioration.

Moreover, this time-dependence of environmental values represents a contrast to values in agricultural systems, where the extent of deterioration in production capacity simply determines the extent of import substitution of agricultural goods, resulting in a linear positive relationship between damage value and the extent of deterioration over time. As a consequence, in situations where a government is attempting to prioritise investment in biosecurity measures between the protection of agricultural and environmental systems, these different time paths of damage values may play an important role in determining such investment priorities.

These differences in the impacts of agricultural and environmental IAS over time highlight potential problems involved in simply using an 'expected damage' criterion to guide biosecurity investment decisions, and suggest the need for a broader set of investment criteria. Issues such as irreversible consequences and damage to irreplaceable social and environmental amenities are difficult to incorporate into conventional economic analyses. In recognition of this fact, recent developments in structured decision making offer a way forward. In particular, group-based multi-criteria decision analysis techniques that blend simulation modelling with expert elicitation and social learning experiments (e.g. Cook and Proctor 2007; Liu et al. 2010) show considerable promise in terms of providing adaptable and practical decision-support tools.

The aim of this paper is to explore the implications of the basic difference between the time paths of damage valuations of agricultural and environmental systems in order to determine its role in influencing biosecurity investment priorities. Our hypothesis is that this difference leads to a general investment bias towards preventing invasive species incursions in agricultural systems over environmental systems because of their more immediately observable damage costs. However, we also expect a sensitivity of this bias to the set of parameters contained in the decision-making framework. We then explore a way forward in terms of removing this bias from biosecurity resource allocation using group-based decision facilitation techniques that incorporate a broader set of decision criteria extending beyond expected economic impact.

The structure of the paper is as follows. Section 2 sets out the bioeconomic model of the biosecurity investment decision-making framework. It characterises the decision problem for both agricultural and environmental systems, including the specification of the time path of damage costs for each system. In so doing it also identifies the set of parameters which are expected to influence investment priorities if the potential value of total damage prevented is the sole criterion used. Section 3 then undertakes a numerical analysis of this model, including a sensitivity analysis of investment priorities to the model's set of parameters. As a consequence of this analysis, clear implications are identified for government policy design for biosecurity investment decisions. Section 4 discusses an alternative method of biosecurity prioritisation using group-based decision support tools.

2 The bioeconomic model

Biosecurity investments related to specific species are often guided by predictive models of impact (Cook et al. 2007; Cook and Matheson 2008; Hodda and Cook 2009; Liu et al. 2010). Generally, these combine a biological spread model with a hazard (or damage) function to predict the losses potentially inflicted over time as a species enters a new region and begins to spread. The predictive models can at times be relatively simple (e.g. Waage et al. 2005), and sometimes highly complex (e.g. Wittwer et al. 2005; Cook et al. 2010a). However, all share the common goal of condensing a range of information into a tangible measure of potential pest impact to decision makers to aid their resource allocation deliberations.

The illustrative bio-economic model we use assumes that an IAS establishes in a region and then spreads over time to infest a particular commodity, which may be agricultural (e.g. a nation's potato crop) or environmental (e.g. a region's wetland habitats). The rate at which this happens depends on the biology of the invasive species. The potential economic loss from bioinvasion has a maximum value, as there is a maximum amount of agricultural or environmental good which can be affected. This may comprise of a loss in market value (in the case of the agricultural good) or in non-market value (in the case of the environmental good). As the invasive species spreads, it infests a greater proportion of that total area and reduces asset value until this maximum is reached.

It is further assumed that once an alien invasive species becomes established in the region it will inevitably spread to carrying capacity in this new environment. Eradication programs, be they localised or regionalised, are not considered. Hence, we essentially model a "prevention only" policy approach to invasive species.

More specifically, assume the region for each good is circular in shape with an area of A, and that each new introduction occurs at the centre of this circle and achieves the same radial rate of spread, r.[2] On this basis the section, s_t, occupied by the invasive species at time t is described by:

$$s_t = r^2 t^2 \pi \quad (1)$$

Hence, the proportion of total area affected at time t is $\frac{s_t}{A}$.

Consider next the cost of the invasion. In the case of the agricultural good it is assumed that each unit of production lost to the invasion is valued at the import replacement cost (V^a) and is constant over time. Also assuming a one-to-one relationship between invaded area and production lost means that the cost of the invasion at time t $\left(C_t^a\right)$ is given by:

$$C_t^a = V^a r^2 t^2 \pi \quad (2)$$

which has a maximum value of AV^a when the invasion is complete.

In the case of the environmental good, account needs to be taken of the assumed increase in the social value per unit of the good as the extent of the invasion increases. In what follows this is done by assuming:

(i) a maximum social value per unit of the environmental good at the point of extinction (V^e);
(ii) a social value per unit of the environmental good at time t which is a function of this maximum value and the proportion of the total area invaded at time t.

This specification means that the social value of the environmental good per unit lost to the invasion at time t $\left(V_t^e\right)$ is given by:

$$V_t^e = V^e \left(\frac{s_t}{A}\right) \quad (3)$$

which has a maximum value of V^e when the invasion is complete. By combining this per unit cost of the

[2] This specification of uniformity is made to simplify the biological component of the model and will be reviewed in section three.

invasion with the specification of its rate of spread, the cost of the invasion at time t $\left(C_t^e\right)$ is given by:

$$C_t^e = V^e\left(\frac{S_t}{A}\right)r^2t^2\pi \quad (4)$$

which has a maximum value of $V^e.A$ when the invasion is complete.

Given these specifications of the annual cost of the invasion for both agricultural and environmental goods, the discounted present value (*PV*) of the total damage cost over the decision-making time horizon (*T*) can be represented (respectively for the agricultural and environmental goods) as:

$$PV(C^a) = \sum_{t=1}^{T}\left(\frac{V^a r^2 t^2 \pi}{(1+d)^t}\right) \quad (5)$$

and:

$$PV(C^e) = \sum_{t=1}^{T}\left(\frac{V^e\left(\frac{S_t}{A}\right) r^2 t^2 \pi}{(1+d)^t}\right) \quad (6)$$

where *d* is the rate of discount of future values. Given this specification, if:

$$PV(C^a) > PV(C^e) \quad (7)$$

then biosecurity investment in protecting the agricultural good will be prioritised. While if:

$$PV(C^a) < PV(C^e) \quad (8)$$

then biosecurity investment in protecting the environmental good will be prioritised.

Finally, it follows from (5) and (6) that the relative size of $PV(C^a)$ and $PV(C^e)$, and therefore the priority for biosecurity investment, depends on the various parameters of the bioeconomic model: *r*, *A*, V^a, V^e, *d* and *T*. A numerical analysis of the role of these parameters in determining priorities for biosecurity investment is presented in the next section, and illustrates the problem with using a single criterion (i.e. present value of total damage cost, *PV*) to make biosecurity resource allocation decisions.

3 Numerical analysis

In order to undertake a numerical analysis of the bioeconomic model of prioritising biosecurity investment developed in the previous section, consider first a base case set of values for the parameters of the model. As previously stated in relation to the biological parameters, it is assumed that the total susceptible area of the agricultural and environmental goods (*A*) is identical, and that the rate of spread of the invasive species (*r*) is the same for both host goods:

$$A = 10{,}000\,\text{ha}$$

and:

$$r = 2.5.$$

In addition, the decision-making parameters for the Present Valuation of damage cost, specifically the time horizon (*T*) and the rate of discount (*d*), are set to:

$$T = 30\,\text{years}$$

and:

$$d = 3\%.$$

Finally, the per unit social value of the environmental good at the point of extinction (V^e) is set to: $V^e = €8.00$ while the (constant) per unit value of lost agricultural production (V^a) is set to: $V^a = €6.00$

Note that these two settings imply the social value of the environmental good at its point of extinction exceeds the value of lost agricultural production.

Given this set of parameter values, Table 1 contains the base case results for the present value of total damage cost for both the agricultural and the environmental goods. The results show that the biological spread of the invasive species through each good's total susceptible area takes 23 years to complete. In addition, during this period the annual damage cost of the agricultural good invasion exceeds that for the environmental good invasion until year 20 (at which point 78.5% of *A* is invaded), after which the annual damage cost for the environmental good invasion is larger in every year until the time horizon is reached at year 30. Also in this context, note that the maximum annual damage cost for both goods occurs in year 23, after which there are no increments to the areas damaged and so the discounting of annual damage costs results in a gradual decrease in the present value of these costs. Finally in relation to Table 1, this base case set of parameter values results in the present value of total damage cost for the environmental good invasion exceeding that for the agricultural good invasion (i.e. €498,123 vs. €493,713). As a consequence, in this example the priority for biosecurity investment based on the *PV* criterion would be given to protecting the environmental good from invasion.

Consider next a sensitivity analysis of the parameters of the model in relation to the base case set of results. In what follows each of the

Table 1 Base case results for the present value of total damage cost

Time	Area affected (Ag. good)	Area affected (Env. good)	Ag. annual damage cost (€)	Env. annual damage cost (€)
0	–	–	0	0
1	20	20	114	0
2	79	79	444	5
3	177	177	970	23
4	314	314	1,675	70
5	491	491	2,541	166
6	707	707	3,552	335
7	962	962	4,694	602
8	1,257	1,257	5,952	997
9	1,590	1,590	7,314	1,551
10	1,964	1,964	8,766	2,295
11	2,376	2,376	10,298	3,262
12	2,827	2,827	11,899	4,486
13	3,318	3,318	13,558	5,998
14	3,848	3,848	15,266	7,833
15	4,418	4,418	17,014	10,022
16	5,027	5,027	18,794	12,596
17	5,674	5,674	20,599	15,585
18	6,362	6,362	22,421	19,018
19	7,088	7,088	24,254	22,922
20	7,854	7,854	26,091	27,323
21	8,659	8,659	27,928	32,244
22	9,503	9,503	29,758	37,707
23	10,000	10,000	30,402	40,535
24	10,000	10,000	29,516	39,355
25	10,000	10,000	28,656	38,208
26	10,000	10,000	27,822	37,096
27	10,000	10,000	27,011	36,015
28	10,000	10,000	26,225	34,966
29	10,000	10,000	25,461	33,948
30	10,000	10,000	24,719	32,959
Present value of total damage costs			493,713	498,123

$A = 10{,}000$, $r = 2.5$, $T = 30$, $d = 0.03$, $V^a = €6.00$, $V^e = €8.00$

Table 2 Sensitivity analysis of the base case results

	PV (Ag. damage) over 30 years (€)	PV (Env. damage) over 30 years (€)
(a) Base case (parameters as above)	493,713	498,123
(b) $r = 2.0$	363,773	303,925
(c) $A = 11{,}000$	513,830	503,959
(d) $d = 0.04$	404,731	398,971
(e) $T = 28$	443,533	431,216
(f) $V^a = €6.40$	526,627	498,123

parameter values are varied in magnitude such that biosecurity investment to protect the agricultural good becomes prioritised over biosecurity investment to protect the environmental good. On this basis it will be possible to demonstrate the role of each of the model's parameters in determining this investment priority. In particular, Table 2 contains results of the effects of such a sensitivity analysis on the present value of total damage cost for each good where:

(a) parameter values assume their base case values (as above);
(b) the rate of spread of the invasion has been reduced from $r = 2.5$ to $r = 2.0$;
(c) the total susceptible area for invasion has been increased from $A = 10{,}000$ to $A = 11{,}000$;
(d) the rate of discount of future damage costs has been increased from $d = 3\%$ to $d = 4\%$
(e) the time horizon for the present valuation has been reduced from $T = 30$ to $T = 28$
(f) the ratio $\frac{V^a}{V^e}$ has been increased from 75% to 80% (i.e. V^a increased from €6.00 to €6.40; $V^e =$ €8.00).

Figure 1 plots the present value of total damage to the agricultural good and the environmental good over time under each of these scenarios, with each panel corresponding to the scenarios listed above.

More specifically, Table 2 shows that if the rate of spread of the invasive species is smaller (i.e. $r = 2.0$ instead of 2.5), or the total susceptible area is larger (i.e. $A = 11{,}000$ ha instead of 10,000 ha), then in both cases the relative size of the present value of total damage cost for the agricultural and environmental goods is reversed, and biosecurity investment in protecting the agricultural good becomes prioritised over protecting the environmental good. The effects of these scenarios on Total Damage Cost over time is illustrated in panels (b) and (c) of Fig. 1, while the base case appears in panel (a). In the case of both an increased spread rate or an increase in susceptible area the explanation for the priority reversal can be attributed to the time-dependent variation in the social value of the environmental good—specifically the dependence of this value on the proportion of the total susceptible area that has been

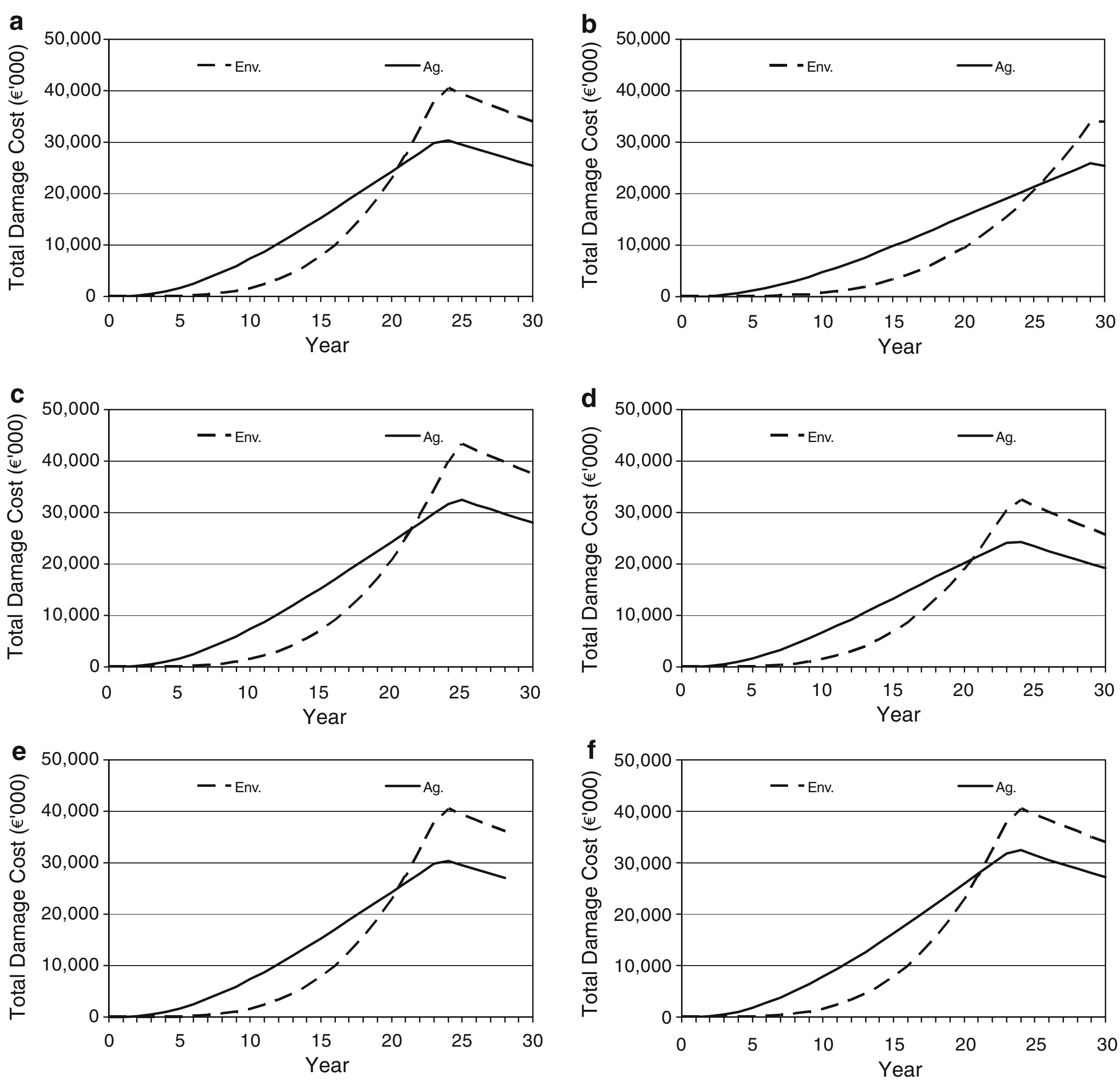

Fig. 1 Sensitivity of total damage costs over time

invaded. For example, in the case of a slower rate of spread, it is not until year 25 that the annual damage cost of the environmental good exceeds that of the agricultural good (compared with year 20 in the base case). While in the case of the larger total susceptible area, this reversal does not occur until year 21.

In addition, Table 2 shows that if the rate of discount of future damage costs is increased (i.e. $d = 4\%$ instead of 3%), or if the time horizon for decision-making is decreased (i.e. $T = 28$ instead of 30), then in both cases the relative size of the present values of damage costs is also reversed. Panels (d) and (e) of Fig. 1 illustrate the effects of these scenarios on Total Damage Costs over time. In both cases biosecurity investment in protecting the agricultural good again becomes prioritised. And once again the explanation for this reversal can be attributed to the time-dependent variation in the social value of the environmental good. However, in these cases, while there is no change in the biological consequences of the invasions, the changes to the decision-making framework act to reduce the relative importance of high annual damage costs further into the future, thereby tilting the priority for biosecurity investment away from protecting the environmental good.

Finally, Table 2 shows that if the per unit damage cost of lost agricultural production is a larger proportion of the social value of the environmental good at the point of extinction (i.e. 80% instead of 75%), then the annual damage costs of lost production of the agricultural good are across-the-board larger and, as previously, the priority for biosecurity investment is reversed. Panel (f) of Fig. 1 demonstrates the effects of this change on Total Damage Cost over time.

In summary, it can be seen from this analysis that the prioritisation of investment to protect the environmental good based on *PV* alone is vulnerable to any change in the model's parameter values which means that the higher annual costs of damage to the environmental good further into the future are less important in the decision-making process. In particular, if the biological parameters of the invasions are such that damage to the environmental good is less noticeable until further into the future, or if the decision-making framework focuses more heavily on short-term annual damage costs, then biosecurity investment to protect the environmental good is less likely to be prioritised over that for the agricultural good, even if the social cost of damage to the environmental good as it nears extinction exceeds the value of lost agricultural production.

Of course, the model and numerical example presented above are purely hypothetical. In reality, policy-makers face uncertainty about the model parameters, and consequently the value of potential environmental losses relative to agricultural losses. However, our stylised discussion suggests a need to investigate the time-dependence characteristic of environmental system values on a case-by-case basis. Unless this information is taken into account in biosecurity resource allocation decisions environmental systems stand to receive a disproportionate amount of protection from invasive species relative to agricultural systems.

4 Discussion

Given the possibility that the present value of total damage, or *PV*, criterion can deliver perverse resource allocation decisions when used in isolation, there is a clear need to enlarge the set of decision criteria used by decision-makers. The task of resource allocation in biosecurity is multi-faceted when both agricultural and environmental IAS impacts can result from invasion, and involves a variety of stakeholders with different priorities or objectives (Linkov et al. 2004). Economic analyses using a narrow single commodity method of assessing risk must be supplemented by other information. Generally, the difficulties involved in quantifying the non-market impact of invasive pests (described above) prevent their inclusion in economic analyses of quarantine strategies. However, if policies directed by such analyses are to reflect social welfare preferences, a more formal recognition of potential non-market damage is needed.

It should be noted that in addition to environmental consequences of invasion, other non-market goods that receive little attention in the literature often need to be considered by policy-makers. These can involve the socio-economic disposition of rural communities. In the same way an environmental resource may have an existence or moral value, so too might a rural community or a historic township. As such, a majority of the community may be willing to pay to preserve it even if they spend most of their time in urban areas and have few social or economic ties to the threatened community itself. Bennett et al. (2004) presents evidence to this affect in three very different regions of rural Australia.[3] Animal welfare

[3] Here the maintenance of rural populations is associated with environmental damage mitigation, so it is difficult to draw conclusions about the willingness of society to pay for the preservation of rural communities per se due to embedded environmental values.

too has also emerged as a non-market good requiring greater attention. Evidence presented in Frank (2008) suggests positive income elasticities for animal welfare, possibly attributable to scientific, philosophical and theological advances over the past three decades, as well as an increased number of companion animals in the developed world.

Given the complex mixture of market and non-market IAS impacts, Multi-Criteria Decision Analysis (MCDA) techniques may offer a practical solution to the dilemma facing biosecurity policy makers by stimulating discussion amongst the decision-making group about possible resource allocation choices, trade-offs and uncertainties. The stated purpose of using MCDA models is to find solutions to complex and uncertain decision-making issues, characterised by multiple alternatives/options that can be evaluated using weighable criteria (Jankowski and Nyerges 2001). For the weighting process to function effectively, decision-makers should have access to a wide range of information and decision support tools so that they may arrive at an appropriate set of weights through a process of consultation and deliberation (Jankowski and Nyerges 2001; Burgman et al. 2006). Policy options can then be evaluated by comparing the relative performance of each against the weighted criteria set, thereby allowing policy-makers to rank options, identify single optimal alternatives, sort alternatives into groups, provide an incomplete ranking or differentiate between acceptable and unacceptable alternatives (Roy 1985; Linkov et al. 2004).

There is a growing trend toward the use of participatory, or group-based MCDA approaches to create a more democratic and open process of resource allocation with respect to multi-faceted problems like biosecurity (Gilmour and Beilin 2006). This is particularly true within government sectors where there is a substantive, instrumental and normative rationale for stakeholder involvement in MCDA processes (Gilmour and Beilin 2006): *substantive* in the sense that these stakeholders combine to form an otherwise absent multidisciplinary local knowledgebase incorporating natural, physical, and social sciences, politics and ethics (McDaniels et al. 1999); *instrumental* in terms of diverse stakeholder groups being more likely to accept decision outcomes from a transparent process that gives voice to their respective concerns (Gilmour and Beilin 2006); and *normative* due to the tendency for decisions to determine usage of common resources, and to therefore involve opportunity costs (Linkov et al. 2004). Certainly, there is a danger that such inclusiveness can lead to a convoluted criteria selection and evaluation process (Dragan et al. 2003), but by focusing on underlying concerns and reasoning rather than their entrenched positions, MCDA stakeholders can engage in integrative bargaining and find creative ways to help work toward consensus (Fisher and Ury 1991).

Experimental use of participatory MCDA methods in facilitating IAS investment decisions has recently been made. Cook and Proctor (2007) use a decision-making group to prioritise a list of IAS according to a set of agreed criteria and criteria weights. These comprised of species with a wide variety of impacts, ranging from those of a purely agricultural significance to those with substantial environmental or social implications. The decision-making group comprised of representatives from a variety of government, industry and community groups that might be affected in the event of an IAS incursion. Liu et al. (2010) uses a similar technique and group composition to rank a set of management options for a single IAS with economic and social implications. This study specifically deals with uncertainty in impact predictions and uses both deterministic and stochastic spread model predictions to compare stakeholder preferences in light of this information. Both studies conclude that although there is more research to be done in terms of the presentation of complex information to stakeholder groups in an MCDA context, there is strong evidence of group-learning and the revision of stated preferences as a result of deliberation.

5 Conclusion

This paper has addressed the apparent difference between the time paths of damage valuations for invasions which affect agricultural compared with environmental systems. In particular, the per unit damage valuation for lost production from agricultural systems is typically based on the associated cost of import replacement, and is therefore largely unrelated to the extent to which the agricultural system is damaged. However, studies have shown that the per unit social valuation of damage to environmental systems is likely to be exponentially related to the extent of its deterioration. As a consequence, the aim of this paper has been to explore the implications of this basic difference between the time paths of damage valuations for agricultural and environmental systems in order to determine its role in influencing biosecurity investment priorities.

To do this a bioeconomic model of prioritising biosecurity investment between protecting an agricultural and an environmental system on the basis of present valuation of total damage cost was developed in Sect. 2. Then in Sect. 3 this bioeconomic model was subjected to a sensitivity analysis of the role of the parameters of the model in influencing investment priorities. Overall it was shown that because the environmental system only displays relatively high annual damage costs well into the future, a decision to prioritise its protection on the basis the single damage cost criterion is vulnerable to any change in the model's parameter values which means that these future damage costs are less important in the decision-making process.

From a biosecurity policy perspective, it follows that unless this time-dependent characteristic of the social value of environmental systems is clearly recognised in the investment prioritising process, environmental systems will be less well-protected even though their ultimate social value exceeds that of agricultural systems. A means of overcoming this problem and structuring biosecurity investment decisions more appropriately may involve participatory MCDA. Applications of this technique, discussed in Sect. 4 have demonstrated that through a process of deliberation and group-learning, policies more reflective of social values can be facilitated.

Acknowledgments We are very grateful to two anonymous reviewers for their helpful comments and suggestions. We would like to acknowledge the support of the Cooperative Research Centre for National Plant Biosecurity, established and supported under the Australian Government's Cooperative Research Centres Program.

Conflict of interest The author D. C. Cook declares that his contribution to the research was sponsored by the Cooperative Research Centre for National Plant Biosecurity and that he has no conflict of interest. The authors R. W. Fraser, J. K. Waage and M. B. Thomas declare that their contributions to the research were not sponsored and that they have no conflict of interest.

References

ABARE (2006) Australian Commodity Statistics. Australian Bureau of Agricultural and Resource Economics

ABARE (2010) Australian Farm Survey Results, 2007–08 to 2009–10. Australian Bureau of Agroicultural and Resource Economics, Canberra

ABS (2004) AgStats Integrated Regional Database, Catalogue No. 1353.0. Australian Bureau of Statistics, Canberra

Adamowicz WL (2004) What's it worth? An examination of historical trends and future directions in environmental valuation. Aust J Agric Resour Econ 48:419–443

Agtrans Research, Dawson N (2005) Review of progress on invasive species. Final report to Department of Environment and Heritage. Agtrans Research, Brisbane

AGWEST Trade and Development (2003) AGTRADE Database. Government of Western Australia—Department of Agriculture, South Perth

Andrée P (2000) The supervision of health and biosafety regulation by world trade rules. Tul Envtl L J 13:271–302

Baker RHA, Black R, Copp GH, Haysom KA, Hulme PE, Thomas MB, Brown A, Brown M, Cannon RJC, Ellis J, Ellis M, Ferris R, Glaves P, Gozlan RE, Holt J, Howe L, Knight JD, MacLeod A, Moore NP, Mumford JD, Murphy ST, Parrott D, Sansford CE, Smith GC, St-Hilaire S, Ward NL (2008) The UK risk assessment scheme for all non-native species. In: Neobiota—from ecology to conservation: the 4th European conference of the working group NEOBIOTA on biological invasions, Vienna, Austria, pp 46–57

Bennett J, van Bueren M, Whitten S (2004) Estimating society's willingness to pay to maintain viable rural communities. Aust J Agric Resour Econ 48:487–512

Blamey R, Rolfe J, Bennett J, Morrison M (2000) Valuing remnant vegetation in Central Queensland using choice modelling. Aust J Agric Resour Econ 44:439–456

Booth C (2008) Submission to the quarantine and biosecurity review. Invasive Species Council, Brisbane

Bowornwathana B (1996) The Phenomenon of new ministries and the politician-bureaucrat perspective: the case of Thailand. Asian Rev Public Adm 8:23–32

Burgman M, Fidler F, McBride M, Walshe T, Wintle B (2006) Eliciting expert judgements: literature review. Australian Centre of Excellence for Risk Analysis, Melbourne

Convention on biological diversity (1991) Convention text. Convention on biological diversity

Cook DC (2003) Prioritising exotic pest threats to Western Australian plant industries. Government of Western Australia—Department of Agriculture, Bunbury

Cook DC, Fraser RW (2008) Trade and invasive species risk mitigation: reconciling WTO compliance with maximising the gains from trade. Food Policy 33:176–184

Cook DC, Matheson AC (2008) An estimate of the potential economic impact of pine pitch canker in Australia. Aust For 71:107–112

Cook DC, Proctor WL (2007) Assessing the threat of exotic plant pests. Ecol Econ 63:594–604

Cook DC, Thomas MB, Cunningham SA, Anderson DL, De Barro PJ (2007) Predicting the economic impact of an invasive species on an ecosystem service. Ecol Appl 17:1832–1840

Cook DC, Hurley M, Liu S, Siddique A-BM, Lowell KE, Diggle A (2010a) Final report CRC10010 enhanced risk analysis tools. Cooperative Research Centre for National Plant Biosecurity, Canberra

Cook DC, Liu S, Murphy B, Lonsdale WM (2010b) Adaptive approaches to biosecurity governance. Risk Anal 30:1303–1314

Dragan M, Feoli E, Fernetti M, Zerihun W (2003) Application of a spatial decision support system (SDSS) to reduce soil erosion in northern Ethiopia. Environ Model Softw 18:861–868

Fisher R, Ury WL (1991) Getting to yes: negotiating agreement without giving in, 2nd edn. Penguin, New York

Frank J (2008) Is there an "animal welfare Kuznets curve"? Ecol Econ 66:478–491

Fraser RW, Cook DC, Mumford JD, Wilby A, Waage JK (2006) Managing outbreaks of invasive species: eradication versus suppression. Int J Pest Manag 52:261–268

GATT (1994) Agreement on the application of sanitary and phytosanitary measures. The results of the Uruguay Round of multilateral trade negotiations: the legal texts. General Agreement on Tariffs and Trade Secretariat, Geneva, pp 69–84

Gilmour J, Beilin R (2006) Stakeholder mapping for effective risk assessment and communication. Australian Centre of Excellence for Risk Analysis, Melbourne

Hanley N, MacMillan D, Patterson I, Wright RE (2003) Economics and the design of nature consewrvation policy: a case study of wild goose conservation in Scotland using choice experiments. Anim Conserv 6:123–129

Hodda M, Cook DC (2009) Economic impact from unrestricted spread of potato cyst nematodes in Australia. Phytopathology 99:1387–1393

Jankowski P, Nyerges T (2001) GIS-supported collaborative decision-making: results of an experiment. Ann Assoc Am Geogr 91:48–70

Linkov I, Varghese A, Jamil S, Seager TP, Kiker G, Bridges T (2004) Multi-criteria decision analysis: a framework for structuring remedial decisions at contaminated sites. In: Linkov I, BakrRamadan A (eds) Comparative risk assessment and environmental decision-making. Kluwer Academic, Amsterdam, pp 15–54

Liu S, Proctor W, Cook D (2010) Using an integrated fuzzy set and deliberative multi-criteria evaluation approach to facilitate decision-making in invasive species management. Ecol Econ 69:2374–2382

Lonsdale W (1994) Inviting trouble: introduced pasture species in northern Australia. Aust J Ecol 19:345–354

McDaniels TL, Gregory RS, Fields D (1999) Democratizing risk management: successful public involvement in local water management decisions. Risk Anal 19:497–510

McKelvie L, Reid C, Haque M (1994) Economic impact of salmonid diseases: furunculosis and infectious haematopoeitic necrosis (IHN): report to the Australian quarantine and inspection service. Australian Bureau of Agricultural and Resource Economics, Canberra

Mumford JD, Knight JD, Cook DC, Quinlan MM, Pluske J, Leach AW (2001) Benefit cost analysis of mediterranean fruit fly management options in Western Australia. Imperial College, Ascot

Office of Technology Assessment (1993) Harmful non-indigenous species in the United States, OTA-F-565. US Government Printing Office, Washington, DC

Pheloung PC, Williams PA, Halloy SR (1999) A weed risk assessment model for use as a biosecurity tool evaluating plant introductions. J Environ Manag 57:239–251

Pimentel D, Lach L, Zuniga R, Morrison D (2000) Environmental and economic costs associated with non-indigenous species in the US. Bioscience 50:53–65

Pimentel D, Zuniga R, Morrison D (2005) Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecol Econ 52:273–288

Roy B (1985) Méthodologie multicritere d' aide à la decision. Roy B (1985). Méthodologie multicritere d' aide à la decision Paris, Economica, Paris

Scholz JT, Stiftel B (2005) Introduction: the challenges of adaptive governance. In: Scholz JT, Stiftel B (eds) Adaptive governance and water conflict: new institutions for collaborative planning. Resources for the Future Press, Washington, DC, pp 1–11

Simberloff D (2006) Risk assessments, blacklists, and white lists for introduced species: Are predictions good enough to be useful? Agric Resour Econ Rev 35:1–10

Stansbury CD, Pretorius ZA (2001) Modelling the potential distribution of Karnal bunt of wheat in South Africa. South Afr J Plant Soil 18:159–168

Waage JK, Fraser RW, Mumford JD, Cook DC, Wilby A (2005) A New Agenda for Biosecurity Department for Food, Environment and Rural Affairs, London

Whitby M (2000) Challenges and options for the UK agri-environment: presidential address. J Agric Econ 51:317–332

Williamson M (1996) Biological invasions. Chapman and Hall, London

Wittwer G, McKirdy S, Wilson R (2005) Regional economic impacts of a plant disease incursion using a general equilibrium approach. Aust J Agric Resour Econ 49:75–89