Final report of ACERA project 1002

Comparing multi-criteria analysis and cost benefit analysis for biosecurity

Procedures, applications and the measurement of consequences

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Executive summary

The application of decision support tools is an emerging field in biosecurity management. The goal of this report is to compare two well-known decision frameworks - cost benefit analysis (CBA) and multi-criteria analysis (MCA) - to support biosecurity decision-making.

Sharing common conceptual roots, the ultimate reason for applying both frameworks is to maximise net benefit or utility in decision-making. While CBA aims to achieve economic efficiency, MCA's primary concern is effectiveness. Such differences triggered a debate over CBA and MCA in the 1960s. However, over time the "either…or" debate has been replaced by a more constructive discussion of a complementary nature. Therefore, instead of arguing, "Which framework is more universally preferred," we attempted to answer the questions of "When is it better to use CBA or MCA" and "How can the two frameworks be jointly applied and mutually enhanced in biosecurity management?"

We form a list of potential factors that will influence the choice and procedural details of CBA and MCA from a literature review. We then identify ten factors that are most relevant to biosecurity management. These factors include, urgency of the decision problem, whether the invasive species and/or its management effort has any non-market impact, whether there are well-defined pre-existing biosecurity management options, level of scientific uncertainty, availability of non-market valuation studies, funding, technical capacity, stakeholders' interest, degree of public agreement, and political mandate .

Among these factors, we believe 'political mandate' will favour the choice of CBA, and '(low degree) of public agreement' and '(high level) of scientific uncertainty' can nudge decision-makers in the direction of MCA. The remaining seven factors will only have an effect on the specific procedures of biosecurity decision-making *after* either CBA or MCA is chosen.

Most importantly, we develop a conceptual framework to combine the features of both CBA and MCA in their "pure" form to create a win-win situation for biosecurity management. In essence, this framework integrates the 'breadth' and representativeness of CBA and the 'depth' in analysis and deliberation of MCA. We demonstrate how to apply this framework with two decision trees and one case study. One tree features CBA as the dominant tool, complemented by four elements of MCA, while, in the other, MCA is the main decision-aid and is complemented by three CBA components. The case study integrates choice modelling and MCA and offers an innovative decision-making model for biosecurity management by combining the CBA feature of broader representativeness and the MCA characteristic of open and in-depth group process.

Table of Contents

Chapter 1 Literature review

1.1 MCA application in biosecurity

Background

Emerging in the 1960s and 1970s, partly as a result of the rapid growth of operations research in WWII, multi-criteria analysis (MCA) provides a framework to facilitate making difficult decisions (Hajkowicz and Collins, 2007). MCA is an evaluation method that ranks or scores the performance of decision options against multiple objectives or criteria. Each decision option is rated against each criterion using performance measures, which collectively form an evaluation matrix. The criteria are weighted to represent their importance. The weights are combined with the evaluation matrix to attain an overall rank or score for each decision option.

Over time MCA has become an established methodology with dozens of books, thousands of applications, dedicated scientific journals, software packages, and university courses (Figueira et al., 2004). A recent review of MCA applications in the arena of environmental management showed that the number of peer-reviewed papers has been growing significantly over the last two decades, and this growth was attributed to both increased decision complexity and the push for transparency in the decision-making process by stakeholders and regulators (Huang et al., 2011). A review of regulatory and guidance documents reveals that regulatory agencies in the United States (e.g. U.S. Environmental Protection Agency) and Europe (e.g. European Union) also implemented MCA in their decision-making process (Kiker et al., 2005).

In the context of environmental management, MCA has been applied to the fields of agricultural resource management (Hayashi, 2000) , energy planning (Pohekar and Ramachandran, 2004), water resources management (Hajkowicz and Collins, 2007), and fisheries (Kjærsgaard, 2007). Only recently has it been *explicitly* applied to assist biosecurity decision-making (Bax et al., 2001; Cook and Proctor, 2007; Maguire, 2004). Since the early 1990s, though, there has been a class of risk assessment models in the biosecurity area that *implicitly* utilise quantitative and qualitative information as primary inputs to MCA, such as the much cited Australian Weed Risk Assessment (AWRA) (Pheloung et al., 1999).

1.1.2 Methodology

We processed over 300 abstracts from the ISI Web of Science on 2 February 2013 with no restriction on publication year, using the search keywords ((nonindigenous or non-indigenous or non-native or nonnative or alien or exotic or invasi* or noxious or weed*) AND (multicriteria or multi-criteria or AHP or "multiple criteria" or “risk assessment”)). We then refined the results by searching for relevant research areas such as environmental sciences and ecology. Next, we examined each publication according to the following selection criteria: 1) the study documented the MCA approach in enough detail. At a minimum, it reported all MCA criteria. 2) The study was not a review article or purely theoretical. 3) Only publications with a major focus on non-native species are included. Those with a more general focus on natural resources management (and non-native species related criteria was only one of many MCA objectives) were rejected from this screening step, for example (Kim et al., 2003).

This screening resulted in the majority of the 300 studies being rejected and only 63 studies met the above criteria (Appendix 1). In order to elucidate the development and current status of how MCA has been applied in biosecurity management, we present an overview of the literature by describing each paper by its date of publication and study area.

We classify decision problems of non-native species into two categories: (1) Decisions about potential non-native species before they arrive in a certain country or region – pre-border biosecurity, and (2) decisions about response actions to non-native species after they have arrived or post-border biosecurity (Maguire, 2004).

This set of 63 studies is not exhaustive and further MCA applications do exist in the grey literature (e.g. governmental papers, consultancy reports, and dissertations). We focused on peer-reviewed literature because they tend to have higher quality.

1.1.3 Result

The first MCA application in the biosecurity field was a 1992 study on developing a rating system for potential exotic bird and mammal pests in California (Smallwood and Salmon, 1992). The most recent papers of the 63 were published in 2012. In this period of 20 years, even though the number of case studies oscillated from year to year, there is clearly an increasing trend over time. Figure 1 demonstrates the trend in terms of cumulative publications in each of the selected periods. The average number of publications per year also increased from about one to nine case studies in the last 20 years.

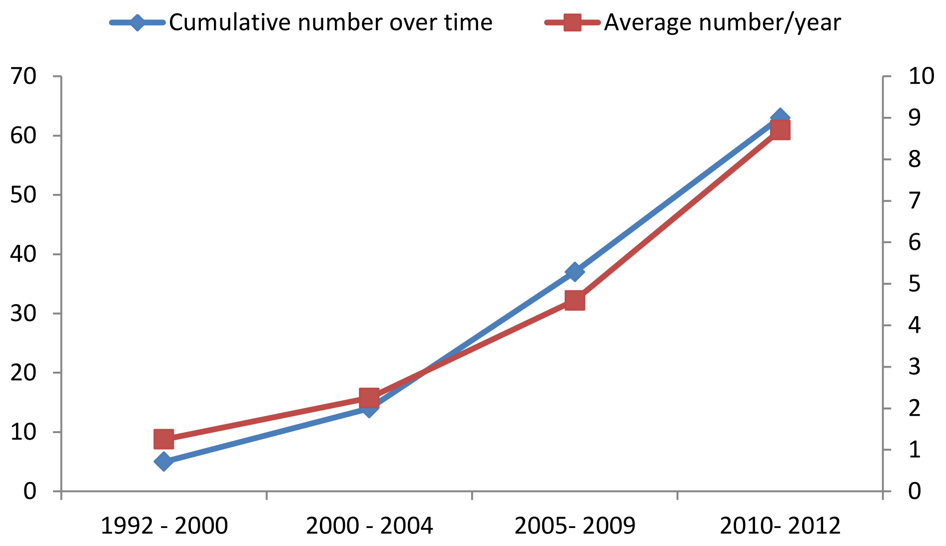


Figure 1. Total and average number of MCA publications from 1992 to 2012.

We also analysed the distribution of MCA papers by geographic region. Each publication was assigned a country and a continent based on its study area. Figure 2 shows the result of this analysis. The distribution is organised by continent. North America, Europe and Oceania dominated the publications of MCA studies in the biosecurity management field, and over 80% of the studies were from these three continents. In Africa, North America, and Oceania, the vast majority of the papers examine a single country: South Africa (100%), United States (95%), and Australia (87.5%), respectively. In Europe though, MCA applications tend to have study areas composed of multiple countries.

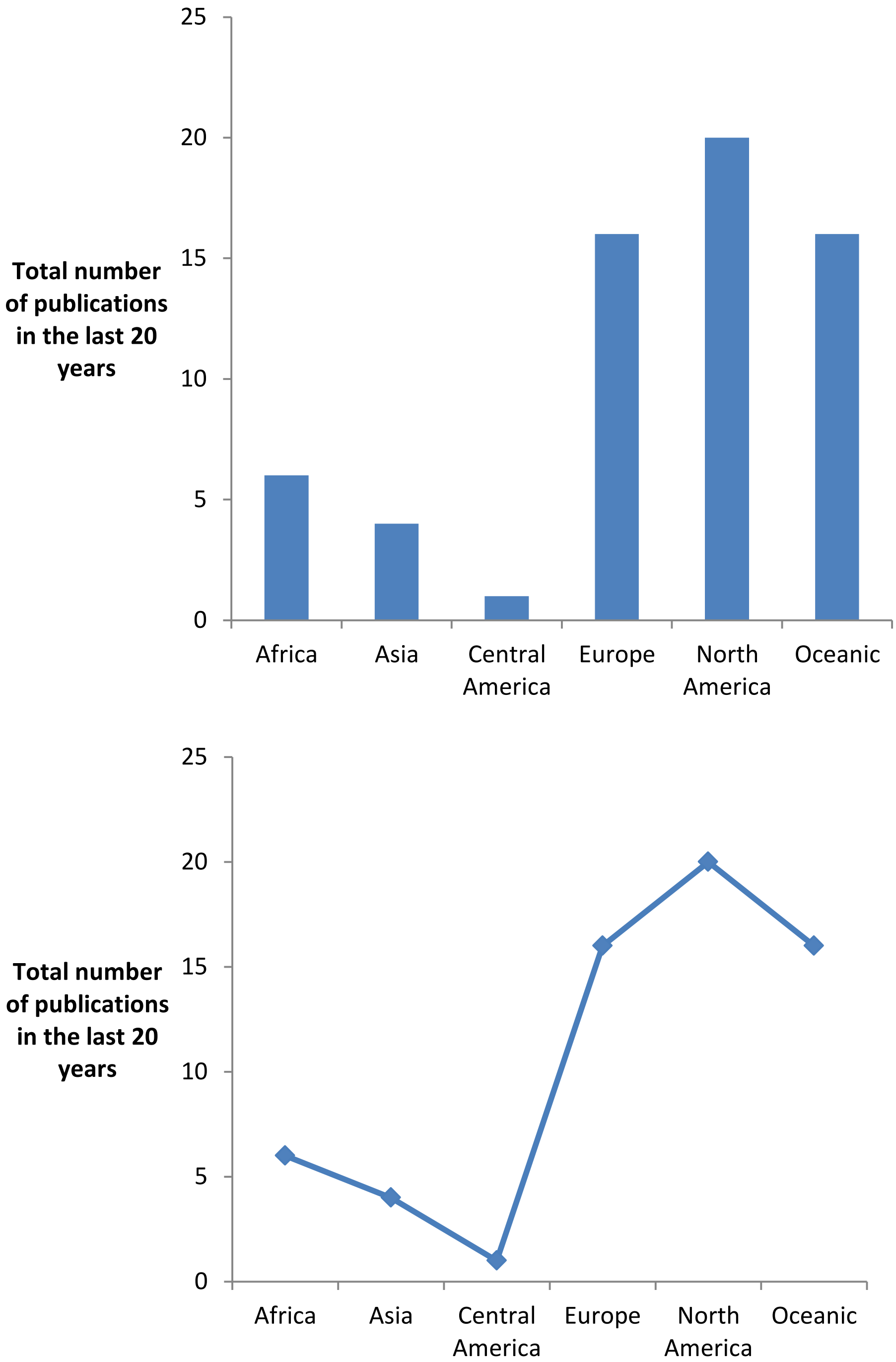


Figure 2. The distribution of MCA publications by continent.

Another way to examine the trend of MCA applications is by examining the temporal and spatial information we assigned to each study. Figure 3 shows the percentage of case studies for a given time period that were conducted in each continent. Countries in North America, Oceania, and Africa pioneered the application of MCA in biosecurity management before 2000, and the methodology was gradually adopted in other parts of the world, in particular, Europe, which has a steady growth in its "market share" in the following three periods. On the other hand, since

2005, the dominance of both North American and Australasian studies has decreased in the last two time periods.

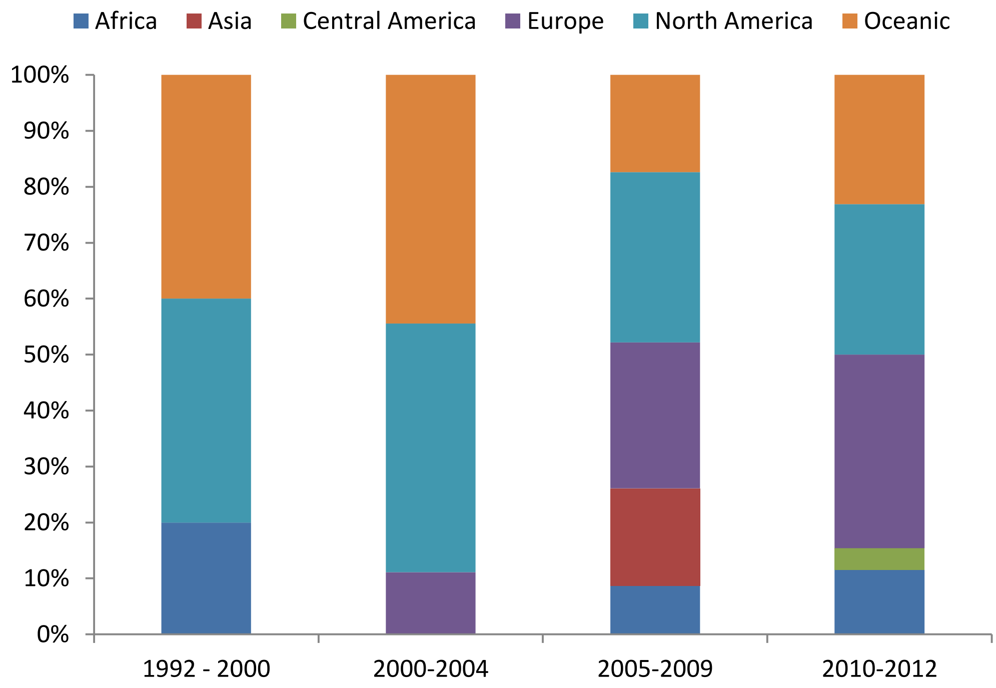


Figure 3. Total number of publications from each continent over time.

In terms of decision points, slightly more than half of the 63 studies were pre-border studies (54%), two studies addressed both pre- and post-border decision problems and the rest were post-border studies. Among the pre-border studies, most belong to a class of risk assessment models such as the AWRA (Pheloung et al., 1999). These risk assessment MCA studies usually assign equal weights to different criteria (e.g. weed attributes and impacts). They have been developed for several taxa including plants (Crosti et al., 2010), fish (Copp et al., 2009), and terrestrial vertebrates (Bomford and Hart, 1999). They also appear to identify non-native species and non-invaders with consistent accuracy across all geographical ranges tested (Gordon and Gantz, 2008).

1.2 CBA applications in biosecurity

Biosecurity is defined as measures to reduce the risk of disease incursion and or spread of invasive species that affect human well-being. Consequently, CBA applications in biosecurity focus on the cost and benefits of implementing these measures. In other words, a biosecurity CBA will compare the cost of one or a combination of these measures to the obtained benefits in the form of the avoided losses that would otherwise have occurred if certain biosecurity actions were not implemented.

While CBA is one of the earliest and the most widely used concepts in economics, its application to biosecurity faces several challenges. First, the benefits of biosecurity measures cannot often be directly measured, generating an inadequacy in our epidemical knowledge or technical capacity to generate widely agreed on predictions or measures of biosecurity actions. The majority of biosecurity CBAs are indeed done via simulation techniques relying on fundamental assumptions that cannot always be satisfactorily verified.

The second difficulty is that biosecurity measures applied in one sector can influence other sectors of an economy, so the costs and the benefits are not often comprehensively evaluated. For example, the loss caused by an animal disease outbreak can be much more significant than simply multiplying a constant average value of an animal with the quantity of infected animals. The disease also has an impact on other sectors of the economy (e.g., food suppliers, domestic consumers, exporters and so on). Some authors, such as Wittwer et al. (2005), Smith et al. (2011), Keogh-Brown et al. (2009), Gohin and Rault (2013), address this issue by analysing the problem in a computable general equilibrium (CGE) model, but most other works keep the analysis within a partial equilibrium by looking at one sector only.

The third challenge is the difficulty in measuring the benefits of biosecurity relating to a human disease or the environment. For human diseases, some authors use the concept of Quality Adjusted Life Years (QALY- see, for example, Prieto and Sacristan, 2003) to calculate how 'many life years' would be saved, but the monetary value of a QALY is still under debate (Nord et al. 1999). Regarding the environment, it is usually difficult to obtain an exact measure for the environmental service saved by a biosecurity measure, simply because there is no market price for it. Choice experiment techniques have been applied to provide statistical estimates (see, for example, Wang et al. 2007 or Choi et al. 2010), but the high cost of implementing a study of this kind and inherent econometric issues are still key obstacles.

Our first report (Appendix 3) aims to provide an estimate of non-market values of controlling invasive pest species in Queensland's bioregions. Six out of 13 bioregions of Queensland were selected for the study. The monetary values of non-market consumption are estimated by the choice experiment technique which allows analysts to estimate the values associated with different attributes of an environmental good or service relative to a biosecurity measure.

The results show that each household in the sample is willing to pay AU$7 to eliminate weed cover from landscape and water bodies, and from AU$93 to AU$232 to reduce the risk of ants and other biting insects. In addition, households are willing to bear between A$100 to A$235 (0.15-0.35% of income) per year to support changes to the existing biosecurity measures. This result suggests that enhanced biosecurity measures are likely to improve household welfare by better protecting the bioregional attributes from an invasive species threat.

Another important challenge in biosecurity CBA is that the benefits of biosecurity measures are usually inter-related. For example, implementing a very effective quarantine program for an imported disease can reduce the attraction of a vaccination program, because both can reduce the probability of an adverse event. In another example, the early detection benefit of a surveillance program depends on the effectiveness of the prior vaccination program and whether post-surveillance eradication programs can effectively remove the disease. Because of this complication, biosecurity CBA usually focus on a single measure at a time by assuming other measures are at 'business as usual' levels.

We will look at each biosecurity measure, from an economic point of view, in more detail, highlighting the relevant literature.

1.2.1 CBA of pre-incursion quarantine

Pre-incursion quarantine is designed to reduce or eliminate the probability of an incursion. Apart from administrative expenditures, consumer welfare effects caused by restricted trade flows are significant costs (Mumford, 2002). Some authors claim that this cost of restricting trade flows may be so significant that bans should be lifted until a virtual risk is confirmed. An example is a partial equilibrium analysis on the Australian banana market by Anderson and James (1998), who show that the benefit of free trade is, on average, enough to compensate damages even if the whole domestic industry is wiped out by an imported disease. This is supported (somewhat) in another analysis by Leroux and MacLaren (2011) who show that the net cost of restricting flows of goods is, on average, positive but reduced by uncertainties and the irreversibility of ban removals. In these works, costs are more than the benefits, so it is not cost-effective to have a quarantine program.

Other studies provide more constructive evidence. For example, Cook et al. (2011) show that the price differential between imported apples and the autarkic price is insufficient to outweigh the increase in expected damage resulting from increased fire blight risk. Soliman et al. (2013) calculate that the economic impacts of the invasion of the plant pathogenic bacterium in Europe is, on average, 114 million EUR/year more than the benefit of completely non-quarantined flows of trade. Another estimate from Breukers et al. (2008) reveals that reducing monitoring frequency in brown rot quarantine increases the costs to EUR 12.5 million/ year in Dutch potato production chain, 60% of which are export losses. According to these authors, benefits exceed costs, so establishing a quarantine program can improve human welfare.

1.2.2 CBA of post-incursion control measures

Once a disease or pest escapes from established quarantine measures (if any) and its entry is confirmed, there are two basic strategies to deal with it, namely eradication and doing nothing. A combination of these two options, containment (Cacho et al., 2008), is to eradicate a sub-area and keep the disease inside the remaining area, usually referred to as containment zones. The benefit of both (complete) eradication and containment strategies is the avoided loss caused by the spread of the disease, so the spread rate is always a key variable in a CBA of a control measure. There is a rule-of-thumb that if the spread rate is always larger than the discount rate, the net benefit of eradication is always positive (Harris et al., 2001).

The cost of an eradication program normally includes (i) the cost of the actual removal, or slaughter, in the case of animals, which may have to be repeated due to re-incursion and (ii) the cost of monitoring activities to make certain that eradication objectives are met. For example, Schoenbaum and Disney (2003) calculate the cost of slaughtering FMD herds in the US is around US$17/animal, plus US$ 5000-7000/farm for post-slaughter cleaning. In addition, the cost for testing and a monitoring visit is between US$ 200-500/farm. Hinrichs et al. (2006) estimate that the costs for culling H5N1 are US$0.25/bird in Vietnam and $1USD/bird in Nigeria. For animal diseases, the eradication cost is often calculated by multiplying the effected number with a constant unit cost, plus an extra expenditure for culling-related activities.

For plants and small pests, calculating eradication costs is more complicated and available estimates are often too broad. It is generally agreed that the cost of eradication increases exponentially with the infestation size, due to likely re-invasion and after some point of spread, an eradication program becomes practically unaffordable (Adamson et al. 2000; Hester et al., 2004; Harris and Timmins, 2009). Some rough estimates for the unit cost of a successful eradication do not exhibit this trend (see Rejmanek and Pitcairn, 2002; Cunningham et al., 2003; Woldendorp and Bomford, 2004), but they are considered to be biased as they ignore the eradication feasibility issues. Taking into account the eradication feasibility, Panetta et al. (2011b), estimate that among 41 invasive plants listed as Class-1 declared pests under the Queensland Land Protection Act 2002, one cannot be practically eradicated, 12 could be eradicated with substantial investment and 40 could be eradicated with less AU$1 million.

One feature of the eradication cost for plants is that it often includes various cost components over a long period of time because eradicating an invasive plant often takes more than 10 years (Panetta and Lawes, 2007). For example, Panetta et al. (2011a), with a stochastic dynamic model, estimate that to eradicate branched broomrape in South Australia, it will take, on average, 73 years (starting in 2008) at a net benefit of AU$68 million. For other weeds (Cunningham et al., 2003; Woldendorp and Bomford, 2004), the time horizon may not be such a length, but a significant component of the total eradication cost should be devoted to monitoring activities over a certain period of time.

1.2.3 CBA of biosecurity surveillance

Biosecurity surveillance is the search to detect unknown incursions. It can be implemented either when there are yet to be known incursions, as a complement to quarantine, or when some known incursions are already confirmed, in parallel with eradication measures. The cost of surveillance is the money spent to implement the search and the benefit is 'early detection', or the possibility of introducing control actions that can avoid extra losses. Thus, surveillance has a benefit only when post-incursion actions, such as eradication or containment, are also desired or implemented.

> Our second report (Appendix 4) analyses the cost-effectiveness of a fire ant eradication program in Queensland with two options, namely surveillance to detect unknown infestations and extending treatment methods beyond detection points to increase the successful eradication probability. The analyses focus on the effects of changes in the program budget and its allocation between surveillance and broadcast treatment.
>
> We find that allocating a larger proportion of the budget to surveillance substantially increased the estimated probability of eradication unless surveillance sensitivity is much lower than expected. In the latter circumstance, a doubling of the budget could achieve a high eradication probability provided that a substantial share is allocated to broadcast treatment. Our analysis also demonstrates the existence of a minimum budget below which eradication is probably infeasible.

The comparison between the cost and benefit of a surveillance program (when desired) depends on how 'early' the search can detect an unknown incursion if it is present. Very ambitious surveillance programs can detect a disease very early but it is often too expensive to do so, while a cheap program may not have a satisfactory detectability. Quantitatively, the relationship between surveillance effort and detectability plays a key role in determining the cost-effectiveness of a surveillance measure.

A number of authors have estimated this quantitative relationship between cost and detectability (e.g., Moore et al. (2011), Hauser and McCarthy (2009), Bogich et al. (2008), Cacho et al. (2007), Cacho et al. (2004), Sharov and Liebhold (1998), Chen et al. (2009) and Kotani et al. (2009)). Most of them use an exponential formulation[1] for the effort-detectability relationship with parameters specified by the authors or estimated from particular experiments. Others use generalized linear mixed models[2] or borrow from CPUE (catch per unit effort) concepts in fisheries. However, questions remain over the use of these relationships outside of the context where they are specified. This is because - apart from the inherent difficulties in estimating the parameter of a probability distribution with limited data - many other non-quantified factors such as morphology, skills of observers (Garrard et al., 2008; Moore et al., 2011), geographical characteristics as well as the surveillance patterns themselves must be taken into account.

The third report (Appendix 5), as an application of cost-benefit analysis, addresses two questions regarding the economics of surveillance for invasive weeds, namely what determines the net benefit of surveillance, and how many financial resources should be allocated to surveillance measures. We build up a generic effort-detection relationship that can be applied to a wide variety of invasive weeds, guiding policy makers on economically best approaches to the management of invasive species, with a minimum set of parameters.

An application to Orange Hawkweed in Australia is provided as an example. The result shows that for a broad range of parameters, the annual surveillance budget for hawkweed should be between $4,000 and $20,000 for every 10,000 ha at risk. Specific surveillance expenditure depends on the spread rate, the damage caused by the weed in each geographical area and the interest rate and other uncertain factors.

[1] See Koopman (1946, 1980) for a deeper discussion of this exponential search function.
[2] See Breslow and Clayton (1993), Kery (2002) for a deeper discussion of the generalized linear mixed model.

1.2.4 CBA of vaccination in human and animal diseases

For human and animal diseases, vaccination may also be considered as a biosecurity measure. The cost of a vaccination program is the expenditure on the used vaccine, the human resources and in some cases, the reduced productivity of the vaccinated subjects. The benefit of vaccination is threefold: (i) reducing/eliminating the incursion probability, (ii) reducing the loss and (iii) slowing down the spread if an outbreak occurs. The benefits of vaccination are very much inter-related with other measures such as quarantine, eradication and surveillance. Thus, an accurate CBA of a vaccination program is challenging because it is hard to calculate the benefits exactly without analysing other existing measures.

All of the CBAs of vaccination we have examined so far assume 'business as usual' levels of the other measures when estimating the vaccination benefit. Examples of analyses on animal vaccination include Barasa et al. (2008), Bates et al. (2003), Elbakidze et al. (2009) for foot-and-mouth disease; Hinrichs et al. (2006), Fasina et al. (2007) for avian influenza and Shwiff et al. (2008) for dog rrabies. CBA on human vaccination includes Leelahavarong et al. (2011) for HIV, Meltzer et al. (2005) for human flu, Tseng et al. (2011) for tuberculosis, Thompson and Tebbens (2007) for poliovirus and Bishai et al. (2012) for measles. All of these vaccination programs are considered to be cost-effective with the benefit far exceeding the cost.

Chapter 2 Conceptual Framework

2.1 The differences between CBA and MCA

The term MCA covers a wide range of distinct approaches (UK Department for Communities and Local Government, 2009a). For the purpose of clarity, we focus on *a subset of MCA that is based on multi-attribute value theory* (Keeney and Raiffa, 1993; Raiffa, 1968) in this report, because our literature review shows that this is the most commonly applied approach in the biosecurity literature. CBA, by comparison, is a more unified body of techniques (UK Department for Communities and Local Government, 2009a).

CBA and MCA share common conceptual roots (Gregory et al., 2012). Some MCA practitioners claim that CBA is a specific type of MCA (Lahdelma et al., 2000), while for some economists, MCA is similar in many respects to cost effectiveness analysis (CEA) (Pearce et al., 2006). The ultimate reasons for applying CBA and MCA are also the same—maximising the net benefits or utility in decision-making processes (Gregory et al., 2012), yet the two frameworks also have very distinct features when comparing to each other. Table 1 lists five major differences in terms of their rationality, goal, role played by analysts, procedure and data requirements.

Table 1. The major differences between CBA and MCA.

	CBA	MCA
Rationality	Monetary commensurability	Incommensurability
Goal	Efficiency	Effectiveness
Role played by analysts	Normative	Positive
Procedure	Outcome-driven	Process-driven
Data requirements	Quantitative	Quantitative or qualitative

2.1.1 Rationality

Alternatives are *incommensurable* when they cannot be precisely measured along some common cardinal scale of units of value (Aldred, 2006). At the heart of CBA is the claim that benefits and costs can be expressed in terms of money and hence made comparable (Aldred, 2002).

MCA does not preclude the monetisation of some policy impacts, whenever monetising impacts can bring insight to decision makers (Gregory et al., 2012). However, all other potential impacts (e.g. environmental and social effects) are expressed in the relevant natural units when possible.

If natural units do not exist, proxies or constructed scales are the second best options (Keeney and Gregory, 2005).

2.1.2 Goal

A concept at the heart of mainstream economics, (Pareto) efficiency is achieved when a policy cannot make one or more members of society better off without making anybody else worse off (Bishop, 1993). CBA provides a model of rationality that is solely based on *economic efficiency*. It assesses whether the aggregated benefits, based on individuals' revealed or stated willingness to pay, of a policy exceed the costs. It offers a rule for *deciding if any policy option at all should be chosen*. CBA also has the capacity to determine the optimal scale of the policy at the point where net benefits are maximised (Pearce et al., 2006).

Because not all impacts in MCA are in the same units, MCA cannot define any optimum or address the issue of whether any option *should* be chosen. Instead, it provides guidance on how to choose between existing policy options or whether to create new options that reflect decision makers' multiple goals, whether they are economic, social or environmental. MCA seeks to find a better way to achieve these final targets by injecting greater transparency and accountability into decision-making processes (Beria et al., 2012). Instead of economic efficiency, MCA aims to achieve *effectiveness* in facilitating decision-makers to address their multiple and potentially incommensurable goals (called objectives or criteria in the language of MCA) (Soma, 2006).

2.1. 3 Role played by analysts

CBA is a *normative* procedure that prescribes whether a decision is good or bad (Pearce et al., 2006). Based on the efficiency criterion, CBA analysts will inform policymakers about the tradeoffs they should make and also pass judgement on the quality of their choices.

MCA practitioners normally restrict their role to *positive* tasks. Instead of making normative recommendations, they will only describe and inform decision makers on the nature of those tradeoffs. MCA practitioners can also design a menu of programs that reflect decision makers' social choices back to them to consider (Gregory et al., 2012).

2.1.4 Procedure

At the end-point of an evaluation, CBA seeks to provide decision makers with a single monetary estimate of the net costs or benefits of the policy options under consideration. MCA, on the other hand, aims to provide decision makers with clarity about what to do, based on a good understanding of the available options, the key trade-offs, uncertainties, and preferences of stakeholders that are explicitly discussed during the MCA process (Gregory et al., 2012). The former is an outcome-driven procedure that is characteristic of rational comprehensive planning, and the latter is process-driven. This is particularly true for *Participatory MCA* (PMCA).

PMCA combines the advantage of MCA in providing structured analysis with the benefits of public participation. Compared to MCA without a participatory component, PMCA offers an opportunity for making divergences in preferences explicit, for facilitating consensus-building and for initiating a dynamic process of social learning (Rauschmayer and Wittmer, 2006).

2.1.5 Data requirement

It is preferable to assess policy impacts in quantitative terms, but this is not an option for some "intangible" impacts such as public support. Natural resources management problems usually involve qualitative information, thus there is a clear need for methods to take it into account (Munda et al., 1994).

CBA cannot use qualitative data directly and the strong quantitative tradition in economics enables researchers to assign dollar values to non-market impacts (Freeman III, 2003). MCA has the capacity of incorporating information of mixed types (both quantitative and qualitative) (Gamper and Turcanu, 2007). For this reason, MCA is more amenable to addressing decision problems with high levels of scientific uncertainty (Liu et al., 2012).

2.2 The evolutionary views about CBA and MCA

A major debate about CBA and MCA originated in the 1960s, when a group of economists at the Harvard Water Program developed a multi-objective version of CBA to account for intangible goods (e.g. distributional effects of water projects). This new approach emphasises the incommensurability of different type of benefits and the importance of collective social choices arising from participatory decision-making (Banzhaf, 2009). Traditional CBA, by comparison, aggregates the multiple impacts of a water project into a single objective, using market or shadow prices. Instead of making the relevant trade-offs known to policy makers, practitioners of traditional CBA make normative recommendations and ultimately judge decisions on the account of economic efficiency.

The debate lasted for more than a decade, and substantial attention was drawn in both government and academic literature (for an overview of this history, see Banzhaf 2009). The stakes became very real when the Water Resources Council, the organisation in charge of setting planning standards and recommending water policies for the United States, proposed a version of multi-objective planning be instituted for water resources management (Cicchetti et al., 1973; Major et al., 1975).

A compromise position was adopted by the Water Resources Council in 1973 (Banzhaf, 2009), so the debate resulted in no clear winner. Nevertheless, it is probably the most important debate about CBA and MCA that has taken place. From an academic perspective, the debate disclosed a set of fundamental issues that differentiates the two decision-making frameworks, some of which, including incommensurability and the role of analysts in decision-making, are still topics of research today (Aldred, 2006; Pielke Jr, 2007). More importantly, the compromise that resulted from the debate opened the door for a "middle way" between CBA and MCA.

Indeed, over time the literature on CBA and MCA has seemed to shift its focus from a primarily dichotomous to a complementary perspective. Case studies that applied both CBA and MCA to the same natural resource management problem showed no clear conclusion about which is better. Both frameworks have been found to have strengths and weaknesses (Brouwer and van Ek, 2004; Gamper et al., 2006; Joubert et al., 1997; Strijker et al., 2000). As the authors of a recent project report funded by Australian Government's Commonwealth Environmental Research Facilities concluded, no single framework is "universally optimal, but rather that the optimal arrangement in any given context depends on the particulars of the context (Marshall et al., 2011)."

It follows that it is time to stop arguing "Which framework is more universally applicable," and to start asking the more constructive questions of "When is it better to use which" and "How the two frameworks can be jointly applied and mutually strengthened by each other?" We attempt to answer these questions by applying the frameworks in biosecurity management.

2.3 Choosing between CBA and MCA in biosecurity decision-making: When is it better to use which?

The existing literature offers no systematic answer to the question of when it is better to choose CBA or MCA as a support tool in natural resources management. However, researchers who have approached the question from different angles proposed a number of factors that will influence the choice. From a valuation perspective, the feasibility of conducting non-market valuation will dictate the choice between CBA and MCA (Hajkowicz, 2008). For institutional economists, CBA and MCA are institutions and three factors should determine which tool to use, including a) who should participate (in decision-making processes) and in what capacity, b) how participants interact, and c) the character of the natural resources to be managed (Vatn, 2009). Acknowledging the embedded value systems in decision-support tools, some researchers conclude that the choice should be consistent with the values of the stakeholders affected by the decision to be made (Gasparatos, 2010).

Different decision-making contexts also lead to different sets of *determining factors*. Within the context of Western Australia's salinity investment framework, the selection of decision tools was supposed to be guided by five questions such as whether the tool is able to prioritise between economic, social and environmental assets (Department of Environment Western Australia, 2003). For comparing two high-profile cases of environmental risk management, political context, underpinning sciences, and problem scale were proposed as the factors that explain the choice between CBA and MCA in the U.K. government (Dietz and Morton, 2011). In the case of community-based environmental governance, 17 non-mutually-exclusive criteria were listed as factors to be considered (Marshall et al., 2011).

For biosecurity management, we identified 10 determining factors that are most relevant and categorised them into four groups: the nature of the decision problem, the policy context, the underpinning science and resource availability (Figure 4). We believe that three out of the 10 are of prime importance and will dictate the choice between CBA and MCA. These factors are scientific uncertainty, public agreement, and political mandate. We name them *primary determining factors* and place them in the inner circle of Figure 4. The remaining seven factors in the outer circle will only influence downstream procedures *after* either CBA or MCA has been chosen, so we call them *secondary determining factors* (to be discussed) in section 2.4.

Political mandate is the primary determining factor that favours the choice of CBA. MCA can be applied in all other situations unless there is a political mandate for CBA, and such a mandate will be supported by two arguments. First, CBA offers a rule for deciding whether it is economically efficient to choose a certain policy option. It is the obvious choice if there is a mandate for efficiency, that is, the aggregated benefit of the policy is larger than its aggregated cost (Pearce et al., 2006). The second argument relates to the argument that CBA studies can canvas a broader and more representative sample of society. By comparison, MCA, especially PMCA, usually only works with smaller groups of people, so the process could be less representative (Gregory, 2000).

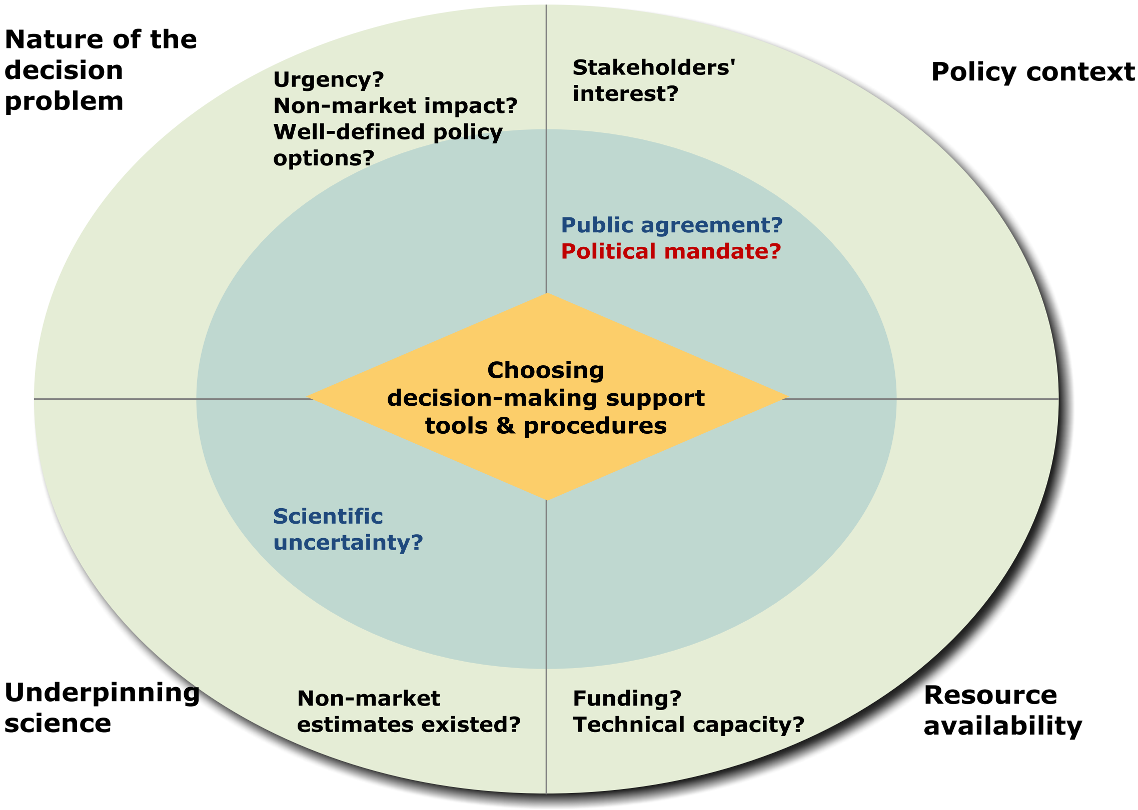


Figure 4. The four categories of factors that influence the choice of decision-making support tools and procedures for biosecurity management. The three factors on the inner cycle are the primary determining factors that will dictate the choice between CBA and MCA. The seven factors on the outer cycle are the secondary factors that will influence downstream procedures after either CBA or MCA has been chosen.

The other two primary determining factors, public (dis)agreement and scientific uncertainty, will work in favour of MCA. Figure 5 demonstrates that MCA, PMCA in particular, is a better fit when there is a combination of a low degree of public agreement and a high level of scientific uncertainty.

One recent example demonstrated the connection between public agreement and the selection of a decision support tool. Lack of public support explained the choice of MCA as the support tool for managing radioactive waste in the U.K. (Dietz and Morton, 2011). Radioactive waste treatment was on the policy agenda for several decades and the U.K. government repeatedly tried and failed to develop solutions. The history left a legacy of suspicion and distrust, and PMCA provided a more transparent and responsive approach to deal with these issues.

For biosecurity management, a decision based on incomplete consideration of diverse perspectives can be controversial and in some cases such decisions have been delayed or halted. In Europe, this was most clearly illustrated when animal rights groups initiated legal action to stop a trial eradication of grey squirrels (Bremner and Park, 2007). Therefore, the setting of objectives related to non-native species management must take into account the differing needs of a broad array of groups (D'Antonio et al., 2004). This is because invasive species simultaneously generate multiple impacts that are spread across many stakeholders (Lodge et al., 2006), who might have very different perceptions about the impacts and benefits (Garcia-Llorente et al., 2008), the equity of the policy outcomes, and whether we should assign monetary values to non-market impacts

Compared to CBA, MCA is more accommodating of high levels of scientific uncertainty for two reasons. MCA has the capacity to take qualitative data, and in the case of PMCA, the deliberation process can function as a forum for risk communication, where decision makers, stakeholders and scientists can interact and discuss the uncertainty (Liu et al., 2011b). The process is important because it encourages social learning and provides a sense of ownership of the evaluation model. By contrast, the relative opacity of the CBA calculations gives limited opportunity for questioning among those without formal training in economics. As a result, quantitative results (a net benefit or benefit-cost ratio) might inspire a degree of confidence that easily supports political decisions, even if they should not (Neves et al., 2008).

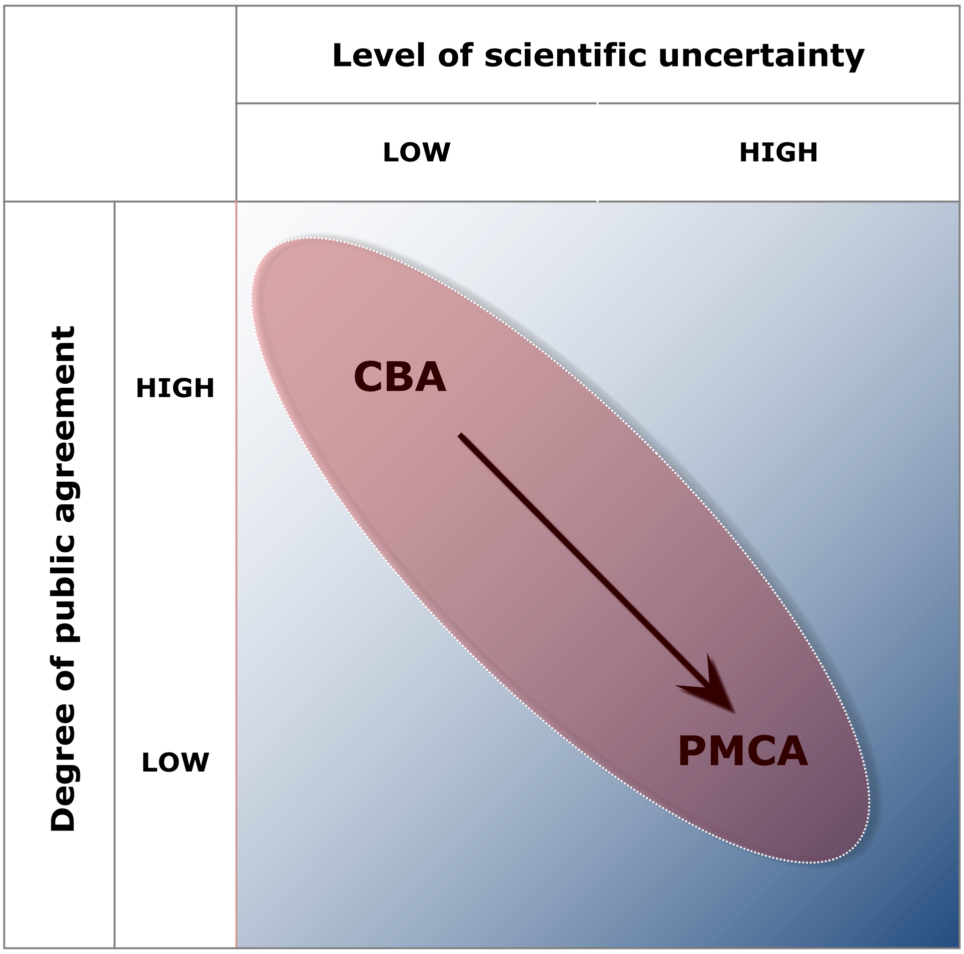


Figure 5. The joint influence of public agreement and scientific uncertainty on the choice of CBA and MCA (PMCA: participatory MCA).

Due to its tolerance of high levels of uncertainty, MCA might be more appropriate for developing countries and regions (e.g. aboriginal communities) where scientific information on potential impacts is not readily available. Another argument for applying MCA is that much of the population in these places is outside of formal market settings (Joubert et al., 1997), yet typically CBA would consider people as individual consumers in a market. By comparison, the deliberation component of PMCA may be more familiar to these communities by involving people as citizens or stakeholders, communicating in groups to find a common solution in the form either of a consensus or a compromise.

2.4 Integrating CBA and MCA in biosecurity decision-making

This section attempts to provide a conceptual framework to answer the question of how CBA and MCA can be jointly applied and mutually strengthened, after either is chosen as the major decision support tool for biosecurity management. We developed two decision trees, one with

CBA as the dominant tool and complemented by MCA elements (CBA orientation in Figure 6) and the other with MCA as the main decision-aid and complemented by CBA components (MCA orientation in Figure 7) to demonstrate the integrative framework.

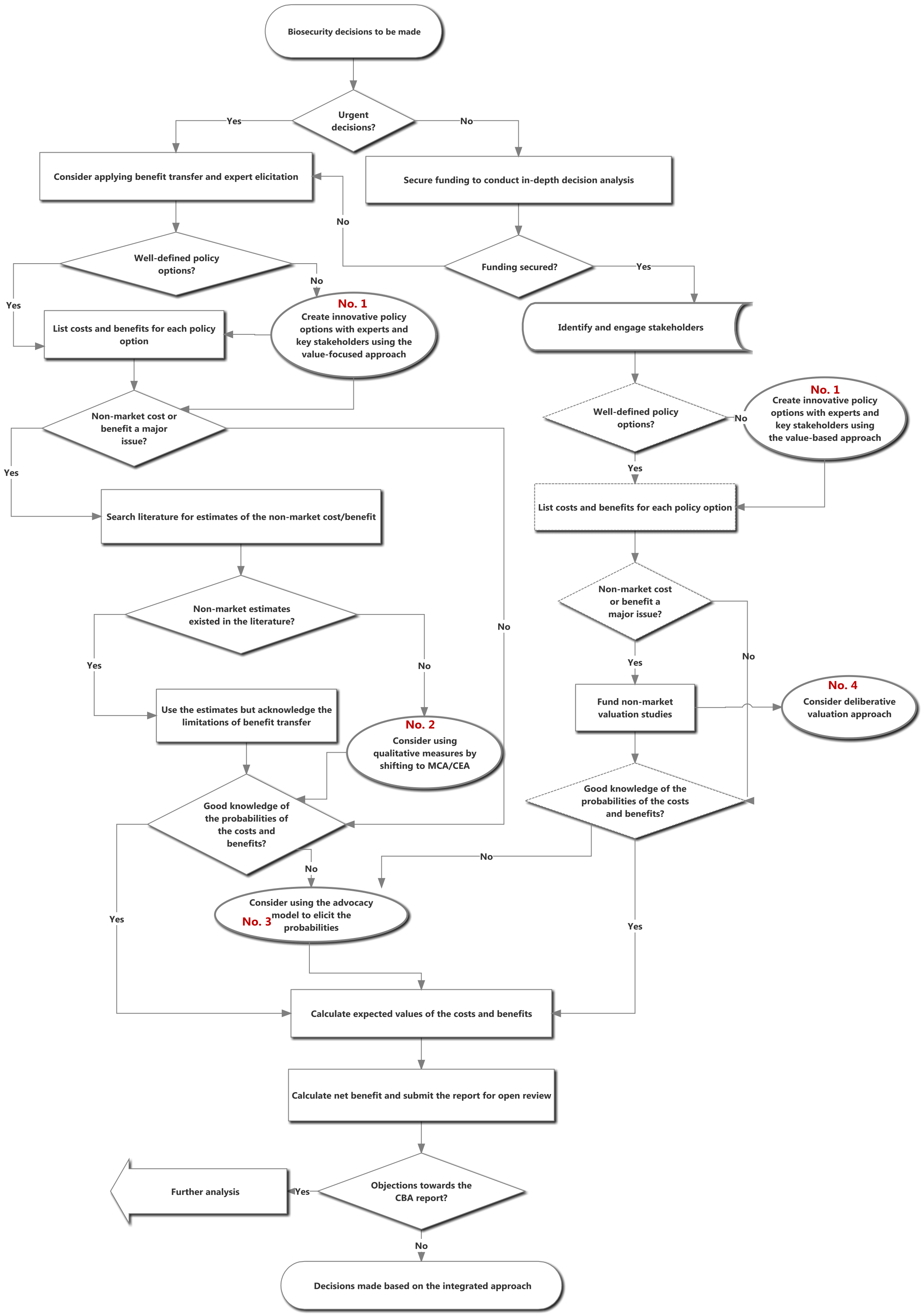

Biosecurity decisions to be made
Urgent decisions?
Yes
No
Consider applying benefit transfer and expert elicitation
Secure funding to conduct in-depth decision analysis
Well-defined policy options?
Funding secured?
No. 1
Create innovative policy options with experts and key stakeholders using the value-focused approach
List costs and benefits for each policy option
Identify and engage stakeholders
Non-market cost or benefit a major issue?
Well-defined policy options?
No. 1
Create innovative policy options with experts and key stakeholders using the value-based approach
List costs and benefits for each policy option
Search literature for estimates of the non-market cost/benefit
Non-market estimates existed in the literature?
Non-market cost or benefit a major issue?
Use the estimates but acknowledge the limitations of benefit transfer
No. 2
Consider using qualitative measures by shifting to MCA/CEA
Fund non-market valuation studies
No. 4
Consider deliberative valuation approach
Good knowledge of the probabilities of the costs and benefits?
Good knowledge of the probabilities of the costs and benefits?
No. 3
Consider using the advocacy model to elicit the probabilities
Calculate expected values of the costs and benefits
Calculate net benefit and submit the report for open review
Further analysis
Objections towards the CBA report?
Decisions made based on the integrated approach

Figure 6. The integrated CBA and MCA framework for biosecurity management (CBA orientation and potential MCA inputs are marked No.1 to No.4 in the ovals).

We present the decision trees in the format of a flowchart, and the seven secondary determining factors function as the burst nodes (in diamond shape) where decision paths split. For example, the first question decision makers typically have to answer is the urgency of the decision problem. Contingent on the answer of Yes or No to this question (the first diamond-shaped node where the decision paths split in both Figure 6 and 7), the decision makers will take different pathways and encounter more burst nodes with new questions to answer downstream. These questions are related to the secondary determining factors, such as whether the policy options pre-existor invasive species will create major non-market impacts. As mentioned before, even though such secondary factors do not dictate the choice of the main decision tool, they still influence the specific procedures or paths of a decision-making process after either CBA or MCA is selected.

We designed the decision trees to be mostly self-explanatory, so that people with knowledge about biosecurity management can follow the logic of the trees without referring to a lengthy text. These decision trees are also designed to be illustrative rather than exhaustive. With limited space (we sought simplicity such that the decision tree has to be able to fit into one printed page), we do not provide a step-by-step guide on how to conduct CBA or MCA. Instead, we focus on the typical issues that biosecurity decision makers will face and arrange them in the most likely sequential order. More importantly, these flowcharts mean to demonstrate how integration works, that is, how elements of a complementary MCA or CBA can be introduced into the dominant CBA or MCA process. We highlight such integrating procedures with an oval shape in both Figure 6 and Figure 7.

When CBA is the main decision tool (Figure 6), biosecurity decision makers might want to consider four MCA components that can be integrated with the CBA process to strengthen the integrated framework (marked No.1 to No.4 in the ovals). First, if there are no well-defined existing policy options, a value-focused approach can be applied to identify innovative alternatives. A focus on values means that policy options are explicitly designed to address the things that matter for the decision, and with the ends in mind, the value-focused thinking can guard against the common mistake of defining policy options too narrowly (Gregory et al., 2012). For example, after an invasive species (e.g. myrtle rust) has spread, defining policy options as “Live with it (i.e. major management efforts invested in mitigation of effects)” misses the fact that it is only a means for accomplishing something of value (e.g. protecting Australia’s native species and landscape amenity). As a result, other possible options such as “slow the spread (i.e. major efforts focused on reducing the rate of expansion of an invasive species from infected properties to surrounding areas with intensive surveillance and movement control)" are likely to be ignored.

Second, CBA reaches its limit and decision-makers have to shift to either MCA or CEA, if qualitative information has to be used. This happens when a policy or an invasive species has major non-market effects, yet, we cannot find any quantitative information from the literature to estimate the costs or benefits of such effects, and it is not an option to conduct research from scratch because the decision has to be made on the fly. MCA practitioners have developed a set of principles and procedures to deal with qualitative information, such as creating constructed or proxy attributes for qualitative measures, if there are no natural attributes available (Keeney and Gregory, 2005).

Third, when we do not have good knowledge about the probabilities of the potential effects of a biosecurity policy, it is recommended to adapt to an "advocacy model," a participatory process where a neutral panel of judges and potentially hostile stakeholders perform as an adversary to look for the weaknesses in an assessment (Franklin et al., 2008). This is because the risk posed by biological invasion belongs to a category of "extreme risks" characterised by low probability and potentially high impacts (Burgman et al., 2012). This low probability means there are limited observations of invasive species and their impacts, and this limited data is also not likely to be representative, so extra scrutiny has to be applied when we elicit information from experts (Franklin et al., 2008).

Fourth, when non-market impacts are of major concern and we do have time to fund studies to evaluate their costs or benefits, it is an option to conduct deliberative monetary valuation (DMV). DMV has at least three advantages over traditional non-market valuation methods: a) group discussion will improve survey respondents' knowledge base, especially if they are not familiar with the goods and services being valued (e.g. ecosystem services affected by myrtle rust), b) such a deliberative process will increase likelihood of stakeholder compliance and support, and c) it will also strengthen the democratic legitimacy of public policies (Spash, 2008).

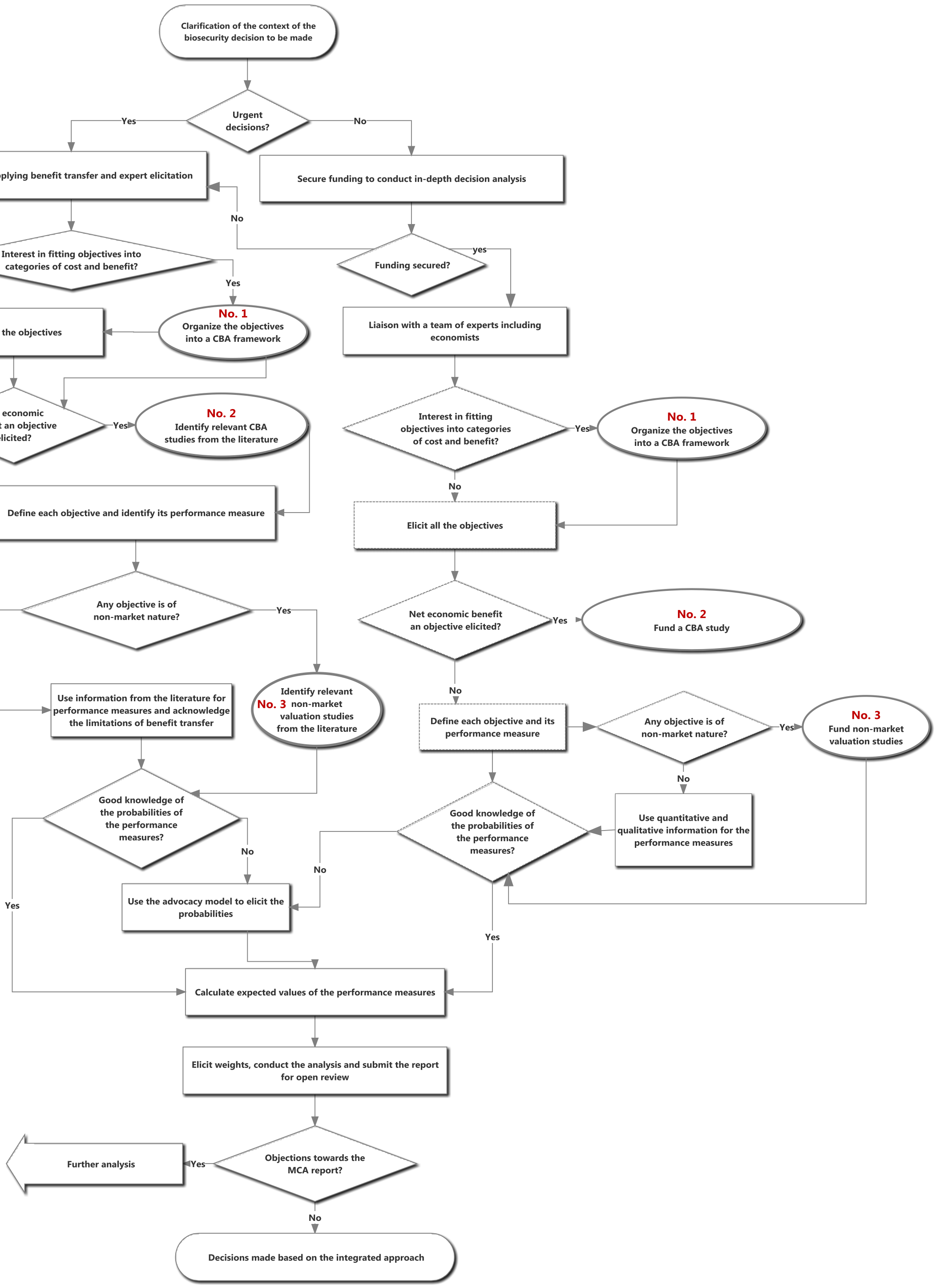
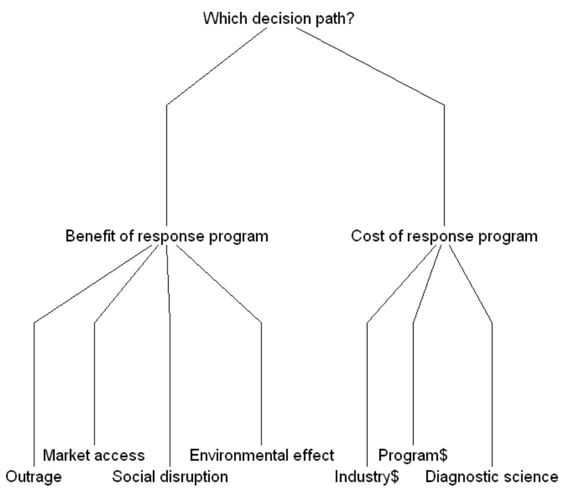


Figure 7. The integrated CBA and MCA framework for biosecurity management (MCA orientation and potential CBA inputs are marked No.1 to No.3 in the ovals).

When MCA is the dominant support tool (Figure 7), three CBA elements might potentially strengthen the joint framework (marked No. 1 to No.3 in the ovals). The first integrative point is to combine the straightforwardness of CBA with the flexibility of MCA by organising the objectives into the two categories of benefit and cost (other options include short-term versus long-term effects and opportunities versus risks). Stakeholders might be interested in looking at the decision problems via the lens of benefit versus cost and potentially comparing the results based on a parallel CBA exercise. For example, in a recent example of using PMCA to select a decision path among Eradication, Containment, and Live with it, if fire blight (*Erwinia amylovora*) becomes established in the Goulburn Valley, Vic (Figure 8a)(Cook et al., 2012), researchers arranged seven objectives into these two groups of cost and benefit. In addition to analysing the decision scores of each policy on each of the seven objectives, we can also look at the three options at an aggregated level. As Figure 8b shows, option 1 or Eradication provides full benefits with a fair amount of cost. By contrast, option 3 or Live with it is the least costly option with zero benefit.



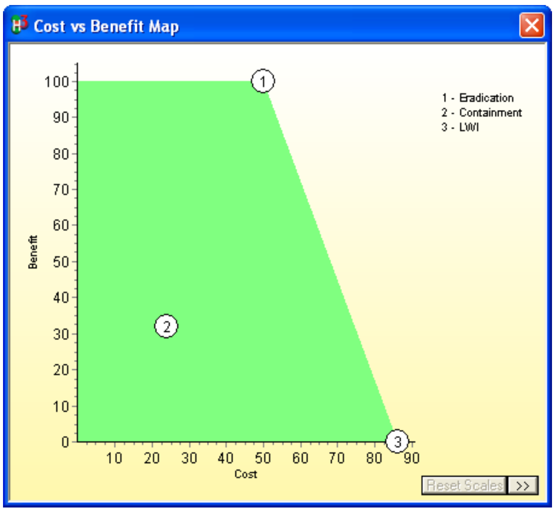


Figure 8a. Arranging PMCA objectives into the categories of cost and benefit for selecting a decision path among Eradication, Containment, and Live with it, if fire blight (*Erwinia amylovora*) becomes established in the Goulburn Valley, Vic.

Figure 8b. Cost vs. benefit map demonstrating the categorical scores of each policy option (1: Eradication, 2: Containment, 3: Live with it). The numbers on the axes are the total preference scores: A larger score on the Y axis means more benefits are provided by that policy option, and a larger score on the X axis means that option is the less costly.

The second opportunity to integrate CBA elements into the MCA is to use the cost-benefit ratio or net benefit/cost as one of the objectives. This is by far the most common way of integrating CBA and MCA in the literature (Barfod et al., 2011; Brouwer and van Ek, 2004; Diakoulaki and Karangelis, 2007; Gühnemann et al., 2012; Messner et al., 2006; Strijker et al., 2000; Sugden, 2005). MCA does not preclude monetisation and will use a combination of monetary and non-monetary estimates to bring insight to complex decisions. Ideally, a CBA study can be commissioned if CBA ratio or net benefit/cost is included as one objective. However, this is only feasible when there is sufficient time and resources to do so (the oval marked No.2 on the right-hand side). Alternatively, a “second best” strategy is to take results from existing CBA studies and apply them to the new context with little or no data, which is essentially a practice of *value transfer* (the oval marked No. 2 on the left-hand side) (Brookshire and Neill, 1992; Brouwer, 2000). However, value transfer is prone to introducing errors due to the differences between the original and new contexts (Rosenberger and Stanley, 2006), so the limitations of such a practice should be clearly addressed.

The third integrative point is to use dollar estimates from non-market valuation studies as impact scores in MCA. Similarly, such results can be derived from a commissioned primary study (the oval marked No. 3 on the right) or from value transfer (the oval marked No. 3 on the left). In the next chapter, we demonstrate how to integrate non-market valuation studies into an MCA with a hypothetical case of biosecurity management. This case study documents an extreme case of CBA and MCA integration where the entire impact matrix of MCA is derived from a choice-modelling study.

Chapter 3 Case study

3.1 Background

An argument that supports CBA is that it results in more democratic public policy making, because CBA canvases a broader and more representative sample of society than is typical of MCA (Gregory, 2000). On the other hand, without open discussion and scrutiny of expert assessment, CBA might conceal ethical dilemmas and subjective probability judgements (Schmidt et al., 2011). The group processes of PMCA offers opportunities for in-depth discussions and risk communication (Liu et al., 2011a) and it gives up some 'breadth' and representativeness in favour of 'depth' in analysis and deliberation (Gregory et al., 2012).

The challenge then is to design a hybrid instrument that combines features of pure CBA and MCA. In this case study, we attempt to achieve this goal by integrating a choice modelling study and a PMCA approach to select the most preferred biosecurity management option for SE Queensland.

Choice modelling (CM) has become increasingly popular in environmental economics due to its ability to estimate relative values for the different attributes for environmental resources (Hanley et al., 2001). In this study, we import an attribute table from a recent CM study that was based on an internet survey of more than 500 respondents (see Appendix 3 for details) into a PMCA process, where 19 biosecurity managers and experts participated in a one-day workshop. This hybrid CM-PMCA offers an innovative decision-making model for biosecurity management by combining the CM feature of broader representativeness and the PMCA characteristic of open and in-depth group process.

3.2 Methodology

PMCA breaks a complex decision-making process into 5 parts: (1) identification of the problem (i.e., the decision to be made), (2) formulation of objectives, (3) development of management options, (4) estimation of impacts associated with each alternative, and (5) evaluation of trade-offs and selection of alternatives (Hammond et al., 1999).

Importing the attribute table from the CM study as an impact matrix, in this study we truncated the typical PMCA process by focusing workshop participants' attention mostly on evaluating trade-offs and selecting preferred options. It is important, however, to ensure all participants share the same understanding about the methodology to be used, the decision context and the impact matrix. Thus we started the workshop with an introduction to PMCA and CM, followed by a group discussion on the decision problem and the details of the impact matrix (Figure 9).

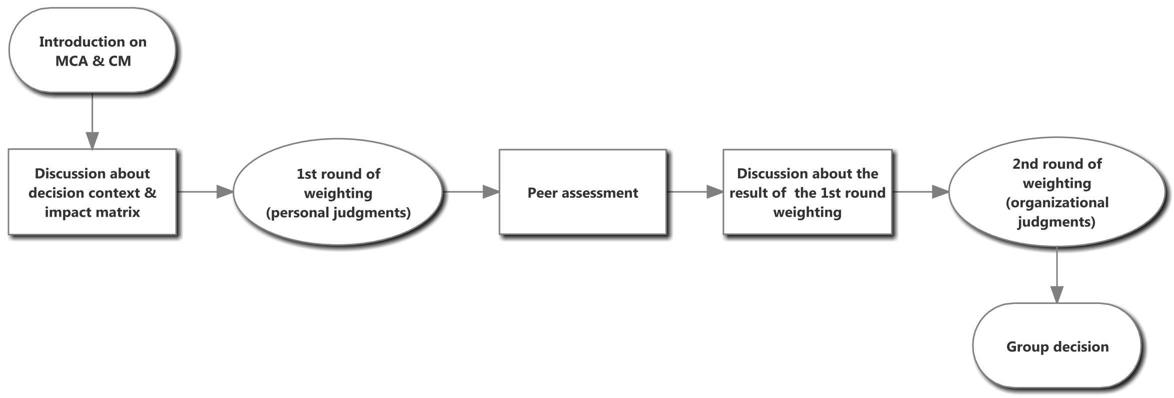


Figure 9. The process of the joint CM-PMCA procedure.

The decision to be made was to select the most preferred option among four alternatives, namely Policy 1, Policy 2, Policy 3 and *Status quo*. There were four objectives: weed cover, recreation, threatened species and cost. The impact matrix is a table, each element of which represents the performance measure for a particular policy option according to a particular objective (Table 2).

Table 2. The impact matrix used in the joint CM-PMCA study.

Objective	**Conditions Now**	**Conditions in 30 years**			
		Policy 1	**Policy 2**	**Policy 3**	***Status Quo***
Areas of landscape and water bodies covered by weeds	10% (17 mil ha)	5% (8 mil ha)	8 % (14 mil ha)	8 % (14 mil ha)	20% (35 mil ha)
Chances of invasive biting insects in the backyard and outdoor recreation areas	Medium (30%-50%)	Medium (30%-50%)	Low (10%-30%)	Medium (30%-50%)	High (50%-70%)
Number of threatened plant and animal species	10	8 (-2 threatened)	8 (-2 threatened)	3 (-7 threatened)	15 (+ 5 threatened)
Cost ($ per household per year)		75	100	150	0

Following group discussion about the impact matrix, the first round of weighting intended to elicit participants' *personal judgements* in terms of how important each objective was relative to the others. The group was instructed to consider their choices from a self-interested perspective, having in mind their income and family situation and the possibility of alternative uses of the resources, and what outcomes they imagined would be best for them. These instructions were identical to those used in the CM survey.

We applied the technique of swing weighting (Von Vinterfeldt and Edwards, 1986) for weight elicitations and the participants were asked to assign an integer between 1 to 100 to each objective, with a larger number indicating a more important criterion.

We used a simplified version of peer assessment (Regan et al., 2006) to aggregate the individual weights across the group. We multiplied each individual's weights by a "credibility factor" and then summed the results across all individuals to arrive at a credibility weighted average of the weights for the group. The basic idea is that the more credible a participant is, as assessed by their peers, the larger his credibility factor is. Each participant was invited to give a brief introduction about his/her professional experience and interest in the decision problem before his or her assessment by the group started, where the rest of the 18 participants each assigned an integer between 0 to 10 to him or her. Each person was also required to provide a self-assessment score. The credibility factor is then the ratio between the total scores a participant received from all members of the group (self-assessment included) and sum of the total scores all of the 19 participants received. These weights then have an expected value of 1/19 but are higher for participants who received above average assessments from the group and vice versa.

After a lunch break, we presented the result of the first round of weighting to the group. A debate between group members was prompted where they discussed questions and problems arising from the previous exercise. Participants raised any concerns over what they had been asked to do and requested clarifications from the project team.

For the second round, participants provided their *organizational judgement* when assigning weights to the four objectives. Participants were required to state and discuss what they thought was more important not only for themselves, but for the wider spectrum of their organisation. When assigning an organisational weight, respondents were reminded that their role in the decision-making process was to select the policy option on behalf of their organisation. Each individual still, though, made the weighting process separately and confidentially.

We used linear functions to transform raw performance values in Table 2 to normalized ones. The normalization process allows comparisons of value to be made between criteria with different units. Given the complexity of the impacts at the scale of our study, it should be noted that it is an extreme simplification to assume linear relationships. As a starting point, however, the linear function has the advantage of being easy to communicate to decision-makers. Using the peer-assessment-adjusted weights and the normalized values, we calculated the decision scores for the four candidate options and presented the results to the group, who thoroughly discussed the result before the final decision was delivered.

3.3 Result

3.3.1 Elicited weights based on individual and organisational judgements

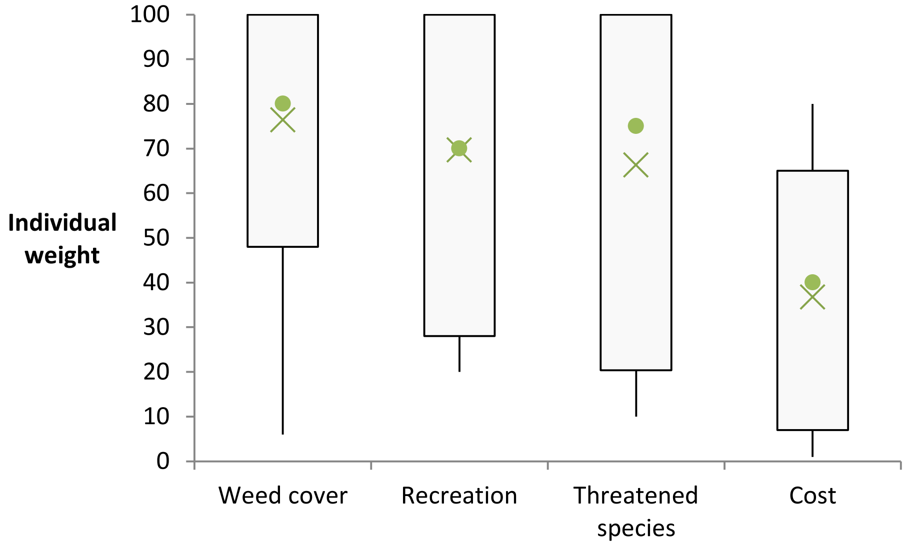


Figure 10a. The result of the first round of weighting based on individual judgements (Cross: mean, circle: median, each box plot symbol indicates the maximum, 90th and 10th percentiles, and minimum).

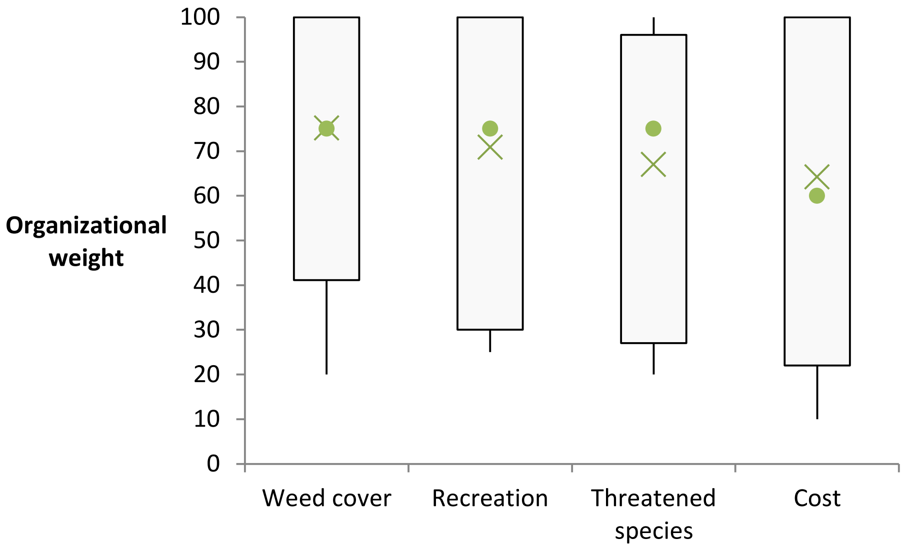


Figure 10b. The result of the second round of weighting based on organisational judgements (Cross: mean, circle: median, each box plot symbol indicates the maximum, 90th and 10th percentiles, and minimum).

Figures 10a and 10b show the results of weight elicitation of the two rounds. There was no major change when participants assigned weights based on organisational judgements, except for an increased weight for the objective of cost. This difference might be explained by the fact that $150 per household per year is probably not much for the participants of the workshop themselves, yet, when being asked to put on the hat of the representative of their organisations, they were forced to think about the cost figures in an aggregated sense and the opportunity cost of such a large biosecurity investment. There was also an increase in spread for the organisational weights for cost, which indicates that participants had more disagreement on the importance of cost in round two weighting.

3.3.2 Decision scores based on individual weights

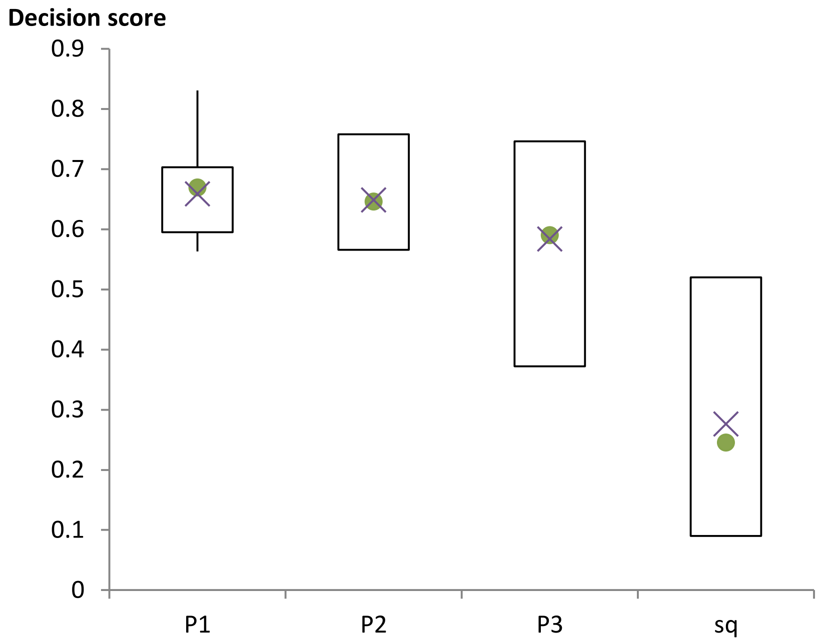


Figure 11. The result of decision scores for the four policy options based on individual weights (Cross: mean, circle: median, each box plot symbol indicates the maximum, 90th and 10th percentiles, and minimum, P1 to P3: Policy 1 to 3, sq: policy status quo).

Using the performance measures in Table 2 and the individual weights, we calculated the overall decision scores for each policy option (Figure 11). The decision score for each option is the weighted average of its performance measures in the consequence table on all the objectives, according to a commonly applied linear additive model (Prato and Herath, 2007; UK Department for Communities and Local Government, 2009b). Letting the decision score for option i and objective j be represented by S_{ij} and the weight for each objective by W_j, then if there are n objectives the overall score for each option, S_i, is given by Equation 1:

$$S_i = W_1 S_i 1 + W_2 S_i 2 + \ldots + W_n S_i n = \sum_{j=1}^{n} W_i S_{ij}$$

The *Status quo* policy was the least preferred option, judging by the group mean or median, however, it also had the largest spread, which suggested that the participants disagreed a great deal on its desirability. By comparison, Policy 1 and Policy 2 had smaller spreads and larger decision scores, thus they were the more preferred options. Our sensitivity analysis showed there was no significant difference between their decision scores. In other words, Policy 1 and Policy 2 were the most preferred options based on individual weights.

3.3.3 Scores of peer assessment

Figure 12 presents the mean, median and variation of the scores received from the whole group (self-assessment included) for each of the nineteen participants (A to T). The results show that there was no commonly agreed authoritative figure in this group, as indicated by the large spreads of every person's scores. For the same reason, there was no commonly agreed incapable person in the group either.

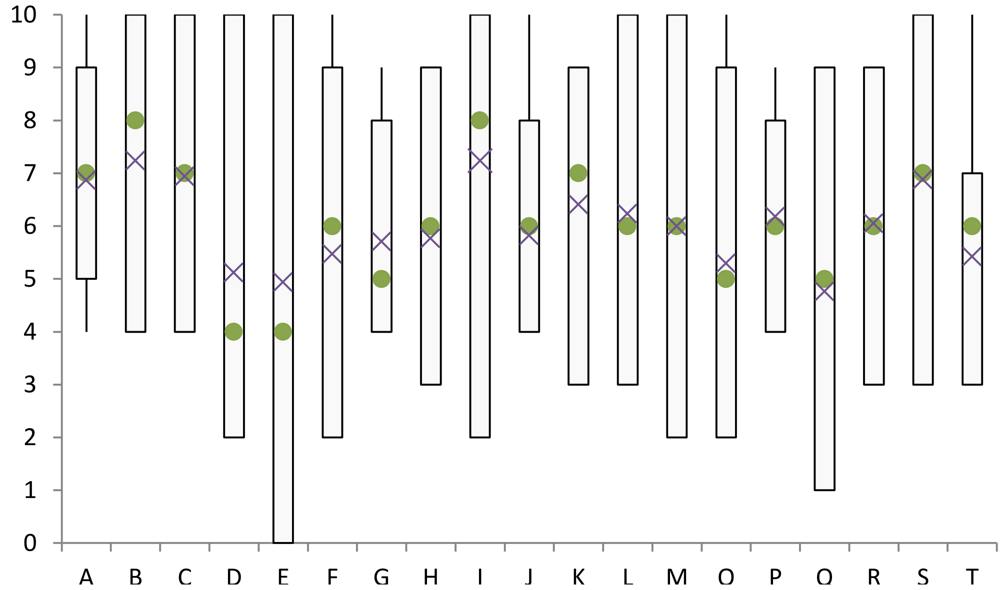
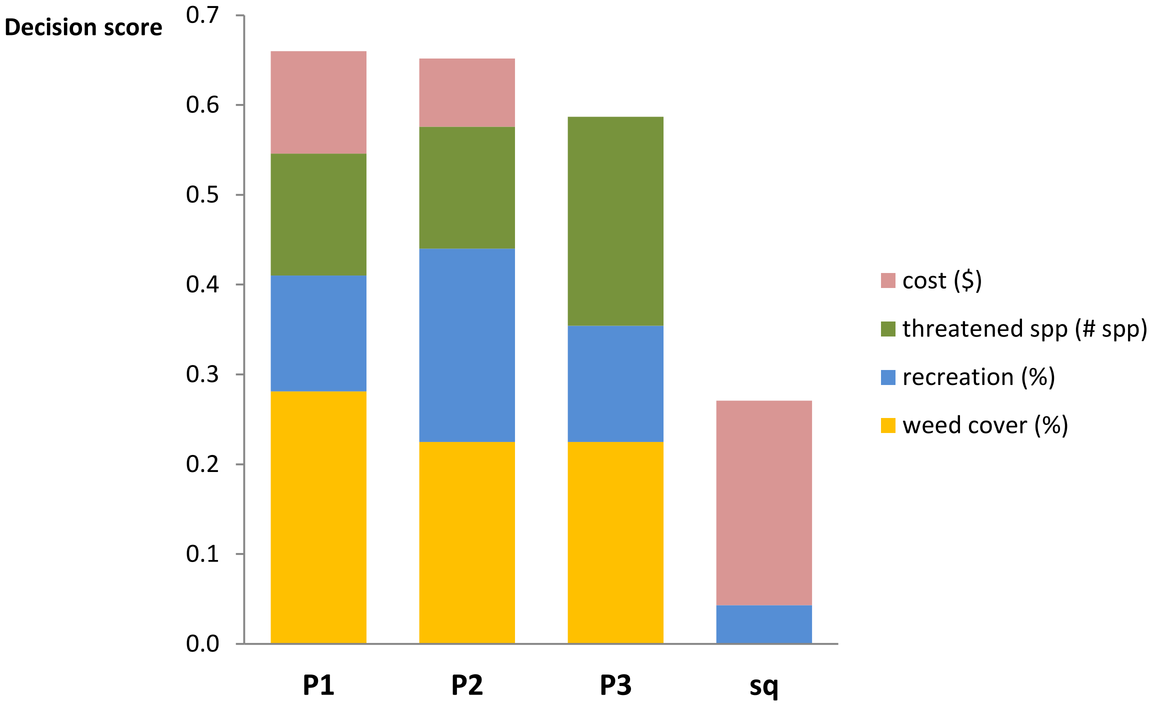


Figure 12. The result of peer assessment scores (0 to 10) assigned to the 19 participants of the workshop (Cross: mean, circle: median, each box plot symbol indicates the maximum, 90th and 10th percentiles, and minimum, A-T: each participant).

3.3.4 Selection of the most preferred option



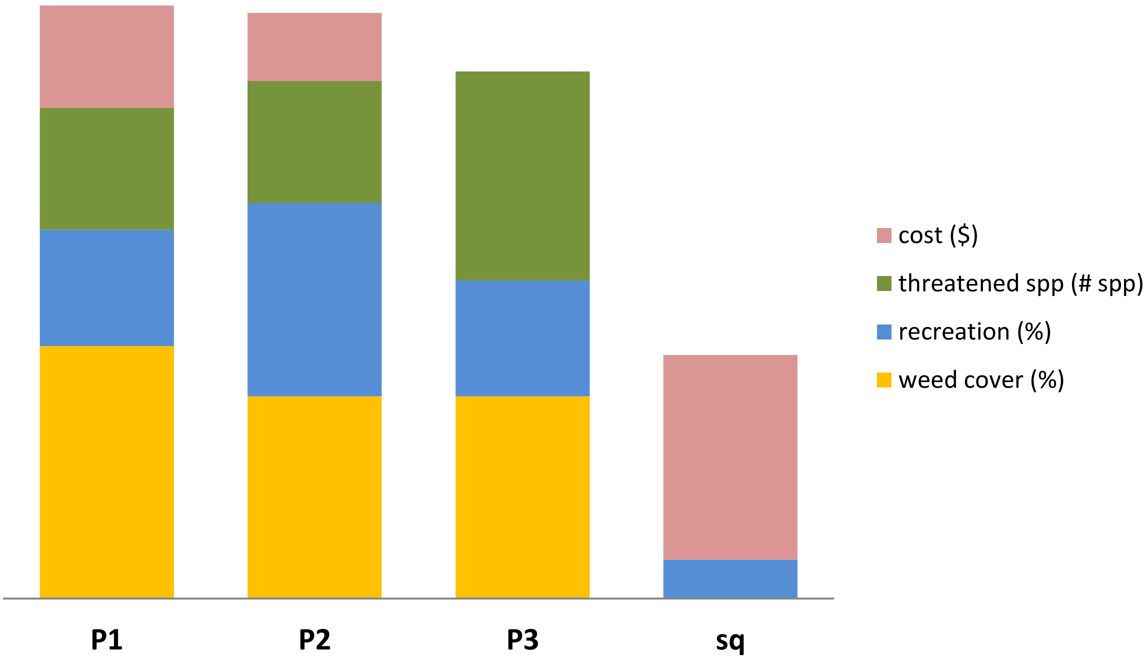


Figure 13. The result of overall decision scores based on peer-assessment-adjusted organisational weights (P1 to P3: Policy 1 to 3, sq: policy status quo).

The height of each bar in Figure 13 shows the total decision scores for each policy option, based on the performance measures (Table 2) and the peer-assessment-adjusted organisational weights. The higher the bar, the more preferred option it is. The resulting ranking was unchanged from round 1 (Figure 11). Our sensitivity analysis again showed that there was no significant difference between the scores for options 1 and 2. Figure 13 also shows the contribution of the decision scores of each objective to the overall score. Even though Policy 1 and Policy 2 were equally preferred, their high scores were explained by different factors. Policy 1 had the highest decision score on the account of weed cover, and Policy 2 was most preferred in terms of recreation. Although the policy *Status quo* was the most preferred on the basis of cost, it was the least preferred option when all four objectives were taken into consideration.

Conclusion

Application of MCA to biosecurity management is an emerging research field. We observed a steady increase in terms of the total number of peer-reviewed studies in the past 20 years, and the same trend was identified for the applications of MCA in other environmental management decision contexts (Huang et al., 2011; Kiker et al., 2005).

One unique feature of MCA studies in the biosecurity domain is the extensive informal use of MCA applications. Our review shows that about 60% of the studies fit into this category. For MCA researchers and practitioners, this is both welcome and worrying at the same time. On the one hand, the MCA framework is well rooted in the biosecurity community and there will be lower entry barriers for policy makers and managers to embrace it. Indeed, it might be seen that informal MCA is conducted each time a manager attempts to balance multiple needs and optimise resource allocation (Lippitt et al., 2008). On the other hand, there will be a higher chance for abuse and misuse of the methodology. We noticed, for example, that for AWRA and its derivative risk assessment systems, there were various ways to combine scores across the criteria and sub-criteria to obtain an overall risk score for each potential non-native species, with additive approaches being the most common. However, based on the high level of interdependence between the criteria, the practice of summing the scores for each criterion violates the assumption of preferential independence for additive models (UK Department for Communities and Local Government, 2009a).

Sharing common conceptual roots, the ultimate reasons for applying CBA and MCA are the same in both cases—to maximise net benefit or utility in decision making (Gregory et al., 2012). Yet, the two frameworks differ in terms of their rationality, goal, role played by analysts, procedures and data requirements. These fundamental differences triggered debate on CBA versus MCA as early as the 1960s. However, over time, literature on CBA and MCA has primarily shifted its focus to a complementary perspective. Instead of arguing "Which framework is more universally applicable," we attempted, in this report, to answer the more constructive questions of "When is it better to use which" and "How can the two frameworks be jointly applied and mutually strengthened for biosecurity management?"

We formed a list of factors from the literature that will influence the choice between CBA and MCA and identified 10 that are most relevant to biosecurity management. We classified them into primary and secondary determining factors. Primary factors include political mandate, public support and scientific uncertainty. These will dictate the choice between CBA and MCA. The remaining seven factors will only have effects on the specific procedures of biosecurity decision-making after either CBA or MCA is chosen.

Political mandate is the primary determining factor that favours the choice of CBA. Arguments about economic efficiency and democratic representation are the primary reasons that support such a mandate for CBA. The arguments for MCA include low degree of public support and high level of scientific uncertainty. The deliberation feature of PMCA will provide a more transparent and inclusive approach to deal with the issues of mistrust and disagreement (Dietz and Morton, 2011). MCA is more accommodating to high levels of scientific uncertainty due to its capability to include qualitative data, and in the case of PMCA, the deliberation process can function as a forum for risk communication, where decision makers, stakeholders, and scientists can interact and discuss the uncertainties (Liu et al., 2011b).

When integrating CBA and MCA, the challenge is to design hybrid instruments that combine features of pure CBA and MCA to create enhanced decision making tools. We developed a conceptual framework to answer the question of how CBA and MCA can be jointly applied and mutually strengthened, after either is chosen as the major decision support tool for biosecurity management. Two decision trees demonstrate the frameworks, one with CBA as the dominant tool complemented by four MCA elements and the other with MCA as the main decision-aid complemented by three CBA components. Using a joint-PMCA study, where the CM attribute table was imported to PMCA as the impact matrix, we showed how to integrate the 'breadth' and representativeness of CBA and the 'depth' in analysis and deliberation of MCA in biosecurity decision-making.

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Appendix 1 The MCA studies reviewed

Andreu, J., Vilà, M., 2010. Risk analysis of potential invasive plants in Spain. Journal for Nature Conservation 18, 34-44.

Barney, J.N., Ditomaso, J.M., 2008. Nonnative species and bioenergy: Are we cultivating the next invader? Bioscience 58, 64-70.

Benke, K.K., Steel, J.L., Weiss, J.E., 2011. Risk assessment models for invasive species: uncertainty in rankings from multi-criteria analysis. Biol. Invasions 13, 239-253.

Bomford, M., Hart, Q., 1999. Risk assessment for importing and keeping exotic vertebrates in Australia. Australian Journal of Environmental Management 6, 149-156.

Brunel, S., Branquart, E., Fried, G., Valkenburg, J.v., Brundu, G., Starfinger, U., Buholzer, S., Uludag, A., Joseffson, M., Baker, R., 2010. The EPPO prioritization process for invasive alien plants. Bulletin OEPP/EPPO Bulletin 40, 407-422.

Buddenhagen, C.E., Chimera, C., Clifford, P., 2009. Assessing Biofuel Crop Invasiveness: A Case Study. Plos One 4.

Caley, P., Kuhnert, P.M., 2006. Application and evaluation of classification trees for screening unwanted plants. Austral Ecology 31, 647-655.

Champion, P.D., Clayton, J.S., 2000. Border control for potential aquatic weeds. Stage 1. Weed risk model.

Cook, D.C., Proctor, W.L., 2007. Assessing the threat of exotic plant pests. Ecological Economics 63, 594-604.

Copp, G.H., Garthwaite, R., Gozlan, R.E., 2005. Risk identification and assessment of non-native freshwater fishes: a summary of concepts and perspectives on protocols for the UK. Journal of Applied Ichthyology 21, 371-373.

Copp, G.H., Vilizzi, L., Mumford, J., Fenwick, G.V., Godard, M.J., Gozlan, R.E., 2009. Calibration of FISK, an Invasiveness Screening Tool for Nonnative Freshwater Fishes. Risk Analysis 29, 457-467.

Cowie, R.H., Dillon, R.T., Jr., Robinson, D.G., Smith, J.W., 2009. Alien non-marine snails and slugs of priority quarantine importance in the United States: A preliminary risk assessment. American Malacological Bulletin 27, 113-132.

Crosti, R., Cascone, C., Cipollaro, S., 2010. Use of a weed risk assessment for the Mediterranean region of Central Italy to prevent loss of functionality and biodiversity in agro-ecosystems. Biol. Invasions 12, 1607-1616.

Cunningham, D.C., Barry, S.C., Woldendorp, G., Burgess, M.B., 2004. A framework for prioritizing sleeper weeds for eradication. Weed Technology 18, 1189-1193.

Daehler, C., Carino, D., 2000. Predicting Invasive Plants: Prospects for a General Screening System Based on Current Regional Models. Biol. Invasions 2, 93-102.

Daehler, C.C., Denslow, J.S., Ansari, S., Kuo, H.C., 2004. A risk-assessment system for screening out invasive pest plants from Hawaii and other Pacific Islands. Conservation Biology 18, 360-

368.

Dawson, W., Burslem, D.F.R.P., Hulme, P.E., 2009. Factors explaining alien plant invasion success in a tropical ecosystem differ at each stage of invasion. Journal of Ecology 97, 657-665.

De Lange, W.J., Stafford, W.H.L., Forsyth, G.G., Le Maitre, D.C., 2012. Incorporating stakeholder preferences in the selection of technologies for using invasive alien plants as a bio-energy feedstock: Applying the analytical hierarchy process. Journal of Environmental Management 99, 76-83.

Essl, F., Nehring, S., Klingenstein, F., Milasowszky, N., Nowack, C., Rabitsch, W., 2011. Review of risk assessment systems of IAS in Europe and introducing the German-Austrian Black List Information System (GABLIS). Journal for Nature Conservation 19, 339-350.

Fejzic, N., Haracic, S.S., Dargatz, D.A., McCluskey, B.J., Cornwell, S.M., Salman, M., Mumford, E.L., 2008. Development of an animal health surveillance infrastructure in Bosnia and Herzegovina - Case report. Slovenian Veterinary Research 45, 43-48.

Forsyth, G.G., Le Maitre, D.C., O'Farrell, P.J., van Wilgen, B.W., 2012. The prioritisation of invasive alien plant control projects using a multi-criteria decision model informed by stakeholder input and spatial data. Journal of Environmental Management 103, 51-57.

Gasso, N., Basnou, C., Vila, M., 2010. Predicting plant invaders in the Mediterranean through a weed risk assessment system. Biol. Invasions 12, 463-476.

Gordon, D.R., Gantz, C.A., 2008. Screening new plant introductions for potential invasiveness: a test of impacts for the United States. Conservation Letters 1, 227-235.

Hayes, K.R., Sliwa, C., 2003. Identifying potential marine pests - a deductive approach applied to Australia. Marine Pollution Bulletin 46, 91-98.

Hurley, M.V., Lowell, K.E., Cook, D.C., Liu, S.A., Siddique, A.B., Diggle, A., 2010. Prioritizing Biosecurity Risks Using a Participatory Decision-Making Tool. Hum. Ecol. Risk Assess. 16, 1379-1394.

Jefferson, L., Havens, K., Ault, J., 2004. Implementing invasive screening procedures: The Chicago Botanic Garden model. Weed Technology 18, 1434-1440.

Koop, A.L., Fowler, L., Newton, L.P., Caton, B.P., 2012. Development and validation of a weed screening tool for the United States. Biol. Invasions 14, 273-294.

Krivanek, M., Pysek, P., 2006. Predicting invasions by woody species in a temperate zone: a test of three risk assessment schemes in the Czech Republic (Central Europe). Diversity and Distributions 12, 319-327.

Lippitt, C.D., Rogan, J., Toledano, J., Saylgermano, F., Eastmana, J.R., Mastro, V., Sawyer, A., 2008. Incorporating anthropogenic variables into a species distribution model to map gypsy moth risk. Ecol. Model. 210, 339-350.

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. Ecological Economics 69, 2374-2382.

Liu, S., Walshe, T., Long, G., Cook, D., 2012. Evaluation of Potential Responses to Invasive Non-Native Species with Structured Decision Making. Conservation Biology 26, 539-546.

McClay, A., Sissons, A., Wilson, C., Davis, S., Evaluation of the Australian weed risk assessment system for the prediction of plant invasiveness in Canada. Biol. Invasions 12, 4085-4098.

Miller, T.K., Allen, C.R., Landis, W.G., Merchant, J.W., 2010. Risk assessment: Simultaneously prioritizing the control of invasive plant species and the conservation of rare plant species. Biological Conservation 143, 2070-2079.

Monterroso, I., Binimelis, R., Rodriguez-Labajos, B., 2011. New methods for the analysis of invasion processes: Multi-criteria evaluation of the invasion of Hydrilla verticillata in Guatemala. Journal of Environmental Management 92, 494-507.

More, S.J., McKenzie, K., O'Flaherty, J., Doherty, M.L., Cromie, A.R., Magan, M.J., 2010. Setting priorities for non-regulatory animal health in Ireland: Results from an expert Policy Delphi study and a farmer priority identification survey. Preventive Veterinary Medicine 95, 198-207.

Mourits, M.C.M., van Asseldonk, M.A.P.M., Huirne, R.B.M., 2010. Multi Criteria Decision Making to evaluate control strategies of contagious animal diseases. Preventive Veterinary Medicine 96, 201-210.

Nentwig, W., Kuhnel, E., Bacher, S., 2010. A Generic Impact-Scoring System Applied to Alien Mammals in Europe. Conservation Biology 24, 302-311.

Nishida, T., Yamashita, N., Asai, M., Kurokawa, S., Enomoto, T., Pheloung, P.C., Groves, R.H., 2009. Developing a pre-entry weed risk assessment system for use in Japan. Biol. Invasions 11, 1319-1333.

Olenin, S., Minchin, D., Daunys, D., 2007. Assessment of biopollution in aquatic ecosystems. Marine Pollution Bulletin 55, 379-394.

Ou, J., Lu, C., O'Toole, D.K., 2008. A risk assessment system for alien plant bio-invasion in Xiamen, China. Journal of environmental sciences (China) 20, 989-997.

Parker, C., Caton, B.P., Fowler, L., 2007. Ranking nonindigenous weed species by their potential to invade the United States. Weed Science 55, 386-397.

Pheloung, P.C., 2001. Weed risk assessment for plant introductions to Australia.

Pheloung, P.C., Williams, P.A., Halloy, S.R., 1999. A weed risk assessment model for use as a biosecurity tool evaluating plant introductions. Journal of Environmental Management 57, 239-251.

Randall, J.M., Morse, L.E., Benton, N., Hiebert, R., Lu, S., Killeffer, T., 2008. The Invasive Species Assessment Protocol: A Tool for Creating Regional and National Lists of Invasive Nonnative Plants that Negatively Impact Biodiversity. Invasive Plant Science and Management 1, 36-49.

Reichard, S.H., Hamilton, C.W., 1997. Predicting invasions of woody plants introduced into North America. Conservation Biology 11, 193-203.

Robinson, T.P., van Klinken, R.D., Metternicht, G., 2010. Comparison of alternative strategies for invasive species distribution modeling. Ecol. Model. 221, 2261-2269.

Roura-Pascual, N., Krug, R.M., Richardson, D.M., Hui, C., 2010. Spatially-explicit sensitivity analysis for conservation management: exploring the influence of decisions in invasive alien plant management. Diversity and Distributions 16, 426-438.

Roura-Pascual, N., Richardson, D.M., Krug, R.M., Brown, A., Chapman, R.A., Forsyth, G.G., Le Maitre, D.C., Robertson, M.P., Stafford, L., Van Wilgen, B.W., Wannenburgh, A., Wessels, N., 2009. Ecology and management of alien plant invasions in South African fynbos: Accommodating key complexities in objective decision making. Biological Conservation 142, 1595-1604.

Shartell, L.M., Nagel, L.M., Storer, A.J., 2011. Multi-Criteria Risk Model for Garlic Mustard (Alliaria petiolata) in Michigan's Upper Peninsula. American Midland Naturalist 165, 116-127.

Skurka Darin, G.M., Schoenig, S., Barney, J.N., Panetta, F.D., DiTomaso, J.M., 2011. WHIPPET: A novel tool for prioritizing invasive plant populations for regional eradication. Journal of Environmental Management 92, 131-139.

Smallwood, K.S., Salmon, T.P., 1992. A rating system for potential exotic bird and mammal pests. Biological Conservation 62, 149-159.

Tricarico, E., Vilizzi, L., Gherardi, F., Copp, G.H., 2010. Calibration of FI-ISK, an Invasiveness Screening Tool for Nonnative Freshwater Invertebrates. Risk Analysis 30, 285-292.

Tucker, K.C., Richardson, D.M., 1995. An expert-system for screening invasive alien plants in South Africa fynbos. Journal of Environmental Management 44, 309-338.

Vaclavik, T., Kanaskie, A., Hansen, E.M., Ohmann, J.L., Meentemeyer, R.K., 2010. Predicting potential and actual distribution of sudden oak death in Oregon: Prioritizing landscape contexts for early detection and eradication of disease outbreaks. For. Ecol. Manage. 260, 1026-1035.

Walshe, T., Burgman, M., 2010. A Framework for Assessing and Managing Risks Posed by Emerging Diseases. Risk Analysis 30, 236-249.

Ward, D.F., Stanley, M.C., Toft, R.J., Forgie, S.A., Harris, R.J., 2008. Assessing the risk of invasive ants: a simple and flexible scorecard approach. Insectes Sociaux 55, 360-363.

Weber, E., Gut, D., 2004. Assessing the risk of potentially invasive plant species in central Europe. Journal for Nature Conservation 12, 171-179.

Weber, E., Gut, D., 2005. A survey of weeds that are increasingly spreading in Europe. Agronomy for Sustainable Development 25, 109-121.

Weber, J., Panetta, F.D., Virtue, J., Pheloung, P., 2009. An analysis of assessment outcomes from eight years' operation of the Australian border weed risk assessment system. Journal of Environmental Management 90, 798-807.

Widrlechner, M.P., Thompson, J.R., Iles, J.K., Dixon, P.M., 2004. Models for predicting the risk of naturalization of non-native woody plants in Iowa. Journal of Environmental Horticulture 22, 23-31.

Xie, G., Chen, W., Lin, M., Zheng, Y., Zheng, Y., 2009a. Regional Ecological Risk Assessment for Invasive Alien Plants of Compositae in Poyang Lake Basin, in: Huang, C., Wiener, J.B., Ni, J. (Eds.), New Perspectives on Risk Analysis and Crisis Response, pp. 381-387.

Xie, G.W., Chen, W.G., Lin, M.Z., Zheng, Y.L., Guo, P.G., Zheng, Y.S., 2009b. Ecological Risk Assessment with MCDM of Some Invasive Alien Plants in China, in: Shi, Y., Wang, S., Peng, Y., Li, J., Zeng, Y. (Eds.), Cutting-Edge Research Topics on Multiple Criteria Decision Making, Proceedings. Springer-Verlag Berlin, Berlin, pp. 580-587.

Yemshanova, D., Koch, F.H., Ben-Haim, Y., Smith, W.D., 2010. Detection capacity, information gaps and the design of surveillance programs for invasive forest pests. Journal of Environmental Management 91, 2535-2546.

Appendix 2 The BCA studies reviewed

Adamson, D., Bilston, L., and Lynch, K., (2000). The potential economic benefits of the Siam weed (Chromolaena odorata) eradication program, Working paper, Queensland Department of Natural Resources and Mines.

Anderson, K. and James, S. (1998) On the need for more economic assessment of quarantine policies, The Australian Journal of Agricultural and Resource Economics, 42: 425-444.

Barasa, M., Catley, A., Machuchu, D., Laqua, H., Puot, E., Tap Kot, D. and Ikiror, D. (2008) Foot-and-Mouth Disease Vaccination in South Sudan: Benefit–Cost Analysis and Livelihoods Impact, Transboundary and Emerging Diseases 55(8): 1865-1682

Bates, W., Carpenter, T. and Thurmond, M. (2003) Benefit-Cost Analysis of Vaccination and Preemptive Slaughter as a Means or Eradicating Foot-and-Mouth Disease. American journal of veterinary research, 64 (7):805-12

Bishai, D., Johns, B., Lefevre, A., Nair, D., Simons, E. et al. (2012) Measles Eradication versus Measles Control: An Economic Analysis. Journal of Vaccines and Vaccination.

Bogich, T., Liebhold, A. and Shea, K., (2008). To sample or eradicate? A cost minimization model for monitoring and manage an invasive species, Journal of Applied Ecology, 45, 1134-42.

Breslow, N. and Clayton, D. (1993). Approximate Inference in Generalized Linear Mixed Models, Journal of the American Statistical Association, 88 (421), 9-25.

Breukers, A., Mourits, M., Werf, W., and Lansink, A. (2008) Costs and benefits of controlling quarantine diseases: a bio-economic modeling approach, Agricultural Economics, 38: 137-149

Cacho, O., Hester, S. and Spring, D. (2007). Applying search theory to determine the feasibility of eradicating an invasive population in natural environments, Australian Journal of Agricultural and Resource Economics, 51, 425-43.

Cacho, O., Wise, R., Hester, S. and Sinden, J. (2008) Bioeconomic modeling for control of weeds in natural environments. Ecological Economics, 65:559–568.

Chen, G., Kry, M., Zhang, J. and Ma, K. (2009). Factors affecting detection probability in plant distribution studies. Journal of Ecology, 97: 1383-1389.

Choi, A., Ritchie, B., Papandrea, F. and Bennett, J. (2010) Economic valuation of cultural heritage sites: A choice modeling approach. Tourism Management, 31(2), 213-220.

Cook, D., Carrasco, L, Paini, D and Fraser, R. (2011) Estimating the social welfare effects of New Zealand apple imports, Australian Journal of Agricultural and Resource Economics 55 (4):599–620

Cunningham, D., Woldendorp, G., Burgess, M. and Barry, S. (2003). Prioritising sleeper weeds for eradication: Selection of species based on potential impacts on agriculture and feasibility of eradication, Working Paper, Bureau of Rural Sciences, Canberra.

Elbakidze, L., Highfield, L., Ward, M., McCarl, B. and Norby, B. (2009) Economic Analysis of Mitigation Strategies for FMD Introduction in Highly Concentrated Animal Feeding Regions, Review of Agricultural Economics, 31(4):931-950.

Fasina, F., Meseko, A., Joannis, T., Shittu, A., Ularamu, H., Egbuji, N., Sulaiman, .K and Onyekonwu, N. (2007) 'Control versus no control: options for avian influenza H5N1 in Nigeria', Zoonoses Public Health. 2007;54(5):173-6

Garrard, G., Bekessy, S., McCarthy, M. and Wintle, B. (2008). When have we looked hard enough? A novel method for setting minimum survey effort protocols for flora surveys. Austral Ecology, 33: 986-998.

Gohin, A. and Rault, A. (2013) Assessing the economic costs of a foot and mouth disease outbreak on Brittany: A dynamic computable general equilibrium analysis, Food policy, 39:97-107.

Harris, S., Brown, J. and Timmins, S. (2001). Weed surveillance: how often to search? Science for Conservation 175. Department of Conservation, Wellington.

Harris, S., Brown, J. and Timmins, S. (2001). Weed surveillance: how often to search? Science for Conservation 175. Department of Conservation, Wellington.

Hauser, C. and McCarthy, M. (2009). Streamlining search and destroy: cost-effective surveillance for invasive species management, Ecology Letters, 12 (7), 683-92.

Hester, S., Odom, D., Cacho, O. and Sinden, J. (2004). Eradication of exotic weeds in Australia: comparing effort and expenditure. Working paper, University of New England.

Hinrichs, J., Sims, L., and McLeod, A. (2006) Some direct costs of control for avian influenza. Proceedings of the 11th International Society for Veterinary Epidemiology and Economics

Keogh-Brown, M., Smith, R., Edmunds, J. and Beutels, P. (2009) The macroeconomics impacts of pandemic influenza: estimates from models of the United Kingdom, France, Belgium and the Netherland', European Journal of Health Economics, 11:543-554.

Kery, M. (2002). Inferring the absence of a species - a case study of snakes. Journal of Wildlife Management, 66, 330-38.

Koopman, B. (1946). Search and Screening (OEG Report No. 56). Alexandria, VA: Center for Naval Analyses.

Koopman, B. (1980). Search and Screening: General Principles with Historical Applications. New York, Pergamon Press.

Kotani, K., Kakinaka, M. and Matsuda, H. (2009). Dynamic economic analysis on invasive species management: Some policy implications of catchability, Mathematical biosciences, 220, 1-14.

Leelahavarong, P., Yot, T., Pitsaphun, W., Chutima, A., Nakorn, P., Chawetsan, N., Wiwat, P. and Viroj. T. (2011) Is a HIV vaccine a viable option and at what price? An economic evaluation of adding HIV vaccination into existing prevention programs in Thailand.' BMC public health, 11 (1): 534

Leroux, A. and Maclaren, D. (2011) The Optimal Time to Remove Quarantine Bans Under Uncertainty: The Case of Australian Bananas', Economic Record 87(276):140–152

Meltzer, M., Neuzil, K., Griffin, M. and Fukuda, K. (2005). An economic analysis of annual influenza vaccination of children. Vaccine, 23(8):1004-14

Moore, J., Hauser, C., Bear, J., Williams, N. and McCarthy, M., (2011). Estimating detection effort curves for plants using search experiments, Ecological Applications, 21 (2), 601-07.

Mumford, J. (2002) Economic issues related to quarantine in international trade, European Review of Agricultural Economics, 29, 329-48.

Nord, E., Pinto, J., Richardson, J., Menzel, P. and Ubel, P. (1999) Incorporating societal concerns for fairness in numerical valuations of health programmes. Health Economics 8 (1): 25–39

Panetta, F., and Lawes, R. (2007) Evaluation of the Australian branched broomrape (Orobanche ramosa) eradication program. Weed Science 55: 644-651.

Panetta, F., Cacho, O., Hester, S. and Sims-Chilton, N. (2011a) Estimating the duration and cost of weed eradication programmes. In: Island invasives: eradication and management: proceedings of the International Conference on Island Invasives. IUCN and The Centre for Biodiversity and Biosecurity, Gland, Switzerland and Auckland, New Zealand, pp. 472-476.

Panetta, F., Csurhes, S., Markula, A. and Hannan-Jones, M. (2011b) Predicting the cost of eradication for 41 Class 1 declared weeds in Queensland, Plant Protection Quarterly 26(2): 42-6.

Prieto, L. and Sacristan, J. (2003) Problems and solutions in calculating quality-adjusted life years (QALYs). Health and Quality of Life Outcomes 1: 80

Rejmanek, M. and Pitcairn, M. (2002). When is eradication of exotic pest plants a realistic goal? Turning the Tide: the Eradication of Invasive Species (eds. Veitch, CR and Clout, MN), pp. 249-53. IUCN SSC Invasive Species Specialist Group. IUCN, Gland (CH) and Cambridge (GB).

Schoenbaum, M. and Disney, T. (2003) Modeling alternative mitigation strategies for a hypothetical outbreak of foot-and-mouth disease in the United States, Preventive Veterinary Medicine 58: 25–52

Sharov, A. and Liebhold, A. (1998). Bioeconomics of managing the spread of exotic pests species with barrier zones, Ecological Applications, 8, 833-45.

Shwiff, S., Kirkpatrick, K. and Sterner, R. (2008) Economic evaluation of an oral rabies vaccination program for control of a domestic dog-coyote rabies epizootic: 1995-2006. Journal of the American Veterinary Medical Association,233(11): 1736-41

Smith, R., Keogh-Brown, M., Barnett, T., (2011) Estimating the economic impact of pandemic influenza: An application of the computable general equilibrium model to the U.K, Social Science and Medicine 73(2):235-44

Soliman. T, Mourits. M., Oude-Lansink A. and Van der Werf, W. (2013) Economic justification for quarantine status – the case study of Candidatus Liberibacter solanacearum' in the European Union.

Thompson, K. and Tebbens, R. (2007) Eradication versus control for poliomyelitis: an economic analysis, Lancet 369: 1363–1371

Tseng, C., Oxlade, O., Menzies, D., Aspler, A. and Schwartzman, K. (2011) Cost-effectiveness of novel vaccines for tuberculosis control: a decision analysis study. BMC public health, 11(1): 55

Wang, X., Bennett, J., Xie, C., Zhang, Z., and Liang, D. (2007) Estimating non-market environmental benefits of the Conversion of Cropland to Forest and Grassland Program: A choice modeling approach. Ecological Economics, 63(1), 114-125.

Wittwer,G., McKirdy, S. and Wilson, R. (2005) Regional economic impacts of a plant disease incursion using a general equilibrium approach, The Australian Journal of Agricultural and Resource Economics, 49: 75–89

Woldendorp, G. and Bomford, M., (2004) Weed eradication: strategies, timeframes and costs, Bureau of Rural Sciences, Canberra

Appendix 3 Report 1

Economic consequences of biological invasions: valuing impacts of invasive species threats by biosecurity attributes in Queensland

Appendix 4 Report 2

Eradication of Invasive Species after Long Distance Spread: Strategy, Benefits and Costs for Red Imported Fire Ants

Appendix 5 Report 3

The Economics of Surveillance for Invasive Weeds: An Application of Cost-Benefit Analysis

References

Aldred, J., 2002. Cost-benefit analysis, incommensurability and rough equality. Environmental Values 11, 27-47.

Aldred, J., 2006. Incommensurability and monetary valuation. Land Economics 82, 141-161.

Banzhaf, H.S., 2009. Objective or Multi-Objective? Two Historically Competing Visions for Benefit-Cost Analysis. Land Economics 85, 3-23.

Barfod, M.B., Salling, K.B., Leleur, S., 2011. Composite decision support by combining cost-benefit and multi-criteria decision analysis. Decision Support Systems 51, 167-175.

Bax, N., Carlton, J.T., Mathews-Amos, A., Haedrich, R.L., Howarth, F.G., Purcell, J.E., Rieser, A., Gray, A., 2001. The control of biological invasions in the world's oceans. Conservation Biology 15, 1234-1246.

Beria, P., Maltese, I., Mariotti, I., 2012. Multicriteria versus Cost Benefit Analysis: a comparative perspective in the assessment of sustainable mobility. Eur. Transp. Res. Rev. 4, 137-152.

Bishop, R.C., 1993. ECONOMIC-EFFICIENCY, SUSTAINABILITY, AND BIODIVERSITY. Ambio 22, 69-73.

Bomford, M., Hart, Q., 1999. Risk assessment for importing and keeping exotic vertebrates in Australia. Australian Journal of Environmental Management 6, 149-156.

Bremner, A., Park, K., 2007. Public attitudes to the management of invasive non-native species in Scotland. Biological Conservation 139, 306-314.

Brookshire, D.S., Neill, H.R., 1992. Benefit Transfers - Conceptual and Empirical Issues. Water Resources Research 28, 651-655.

Brouwer, R., 2000. Environmental value transfer: state of the art and future prospects. Ecological Economics 32, 137-152.

Brouwer, R., van Ek, R., 2004. Integrated ecological, economic and social impact assessment of alternative flood control policies in the Netherlands. Ecological Economics 50, 1-21.

Burgman, M., Franklin, J., Hayes, K.R., Hosack, G.R., Peters, G.W., Sisson, S.A., 2012. Modeling Extreme Risks in Ecology. Risk Analysis 32, 1956-1966.

Cicchetti, C.J., Davis, R.K., Hanke, S.H., Haveman, R.H., 1973. Evaluating Federal Water Projects: A Critique of Proposed Standards. Science 181, 723-728.

Cook, D., Liu, S., Aurambout, J.P., Villalta, O., Kriticos, D., Edwards, J., 2012. Final project report of "Communicating Uncertainty in Biosecurity Adaption". Plant Biosecurity CRC, Caberra, Australia.

Cook, D.C., Proctor, W.L., 2007. Assessing the threat of exotic plant pests. Ecological Economics 63, 594-604.

Copp, G.H., Vilizzi, L., Mumford, J., Fenwick, G.V., Godard, M.J., Gozlan, R.E., 2009. Calibration of FISK, an Invasiveness Screening Tool for Nonnative Freshwater Fishes. Risk Analysis 29, 457-467.

Crosti, R., Cascone, C., Cipollaro, S., 2010. Use of a weed risk assessment for the Mediterranean region of Central Italy to prevent loss of functionality and biodiversity in agro-ecosystems. Biological Invasions 12, 1607-1616.

D'Antonio, C.M., Jackson, N.E., Horvitz, C.C., Hedberg, R., 2004. Invasive plants in wildland ecosystems: merging the study of invasion processes with management needs. Frontiers in Ecology and the Environment 2, 513-521.

Department of Environment Western Australia, 2003. Salitnity investement framework Interim Report -- Phase I, Department of Environment, Salinity and Land Use Impacts Series No. SLUI 32.

Diakoulaki, D., Karangelis, F., 2007. Multi-criteria decision analysis and cost-benefit analysis of alternative scenarios for the power generation sector in Greece. Renew. Sust. Energ. Rev. 11, 716-727.

Dietz, S., Morton, A., 2011. Strategic Appraisal of Environmental Risks: A Contrast Between the United Kingdom's Stern Review on the Economics of Climate Change and its Committee on Radioactive Waste Management. Risk Analysis 31, 129-142.

Figueira, J., Greco, S., Ehrgott, M., 2004. Multiple Criteria Decision Analysis:State of the Art Surveys (International Series in Operations Research & Management Science). Springer.

Franklin, J., Sisson, S.A., Burgman, M.A., Martin, J.K., 2008. Evaluating extreme risks in invasion ecology: learning from banking compliance. Diversity and Distributions 14, 581-591.

Freeman III, A.K., 2003. The measurement of environmental and resources values. Resource for the Future, Washington, D,C,.

Gamper, C.D., Thoni, M., Weck-Hannemann, H., 2006. A conceptual approach to the use of Cost Benefit and Multi Criteria Analysis in natural hazard management. Natural Hazards and Earth System Sciences 6, 293-302.

Gamper, C.D., Turcanu, C., 2007. On the governmental use of multi-criteria analysis. Ecological Economics 62, 298-307.

Garcia-Llorente, M., Martin-Lopez, B., Gonzalez, J.A., Alcorlo, P., Montes, C., 2008. Social perceptions of the impacts and benefits of invasive alien species: Implications for management. Biological Conservation 141, 2969-2983.

Gasparatos, A., 2010. Embedded value systems in sustainability assessment tools and their implications. Journal of Environmental Management 91, 1613-1622.

Gordon, D.R., Gantz, C.A., 2008. Screening new plant introductions for potential invasiveness: a test of impacts for the United States. Conserv. Lett. 1, 227-235.

Gregory, R., Failing, L., Harstone, M., Long, G., McDaniels, T., Ohlson, D., 2012. Structured decision making: A practical guide to environmental managment choices. Wiley-Blackwell, West Sussex, UK.

Gregory, R.S., 2000. Valuing environmental policy options: A case study comparison of multiattribute and contingent valuation survey methods. Land Economics 76, 151-173.

Gühnemann, A., Laird, J.J., Pearman, A.D., 2012. Combining cost-benefit and multi-criteria analysis to prioritise a national road infrastructure programme. Transport Policy 23, 15-24.

Hajkowicz, S., 2008. Rethinking the economist's evaluation toolkit in light of sustainability policy. Sustainability: Science, Practice, & Policy 4, 1-8.

Hajkowicz, S., Collins, K., 2007. A review of multiple criteria analysis for water resource planning and management. Water Resources Management 21, 1553-1566.

Hammond, J.S., Keeney, R.L., Raiffa, H., 1999. Smart choices: a practical guide to making better decisions. Harvard Business School Press, Boston.

Hanley, N., Mourato, S., Wright, R.E., 2001. Choice modelling approaches: A superior alternative for environmental valuation? Journal of Economic Surveys 15, 435-462.

Hayashi, K., 2000. Multicriteria analysis for agricultural resource management: A critical survey and future perspectives. European Journal of Operational Research 122, 486-500.

Huang, I.B., Keisler, J., Linkov, I., 2011. Multi-criteria decision analysis in environmental sciences: Ten years of applications and trends. The Science of the Total Environment 409, 3578-3594.

Joubert, A.R., Leiman, A., deKlerk, H.M., Katua, S., Aggenbach, L.C., 1997. Fynbos (fine bush) vegetation and the supply of water: a comparison of multi-criteria decision analysis and cost-benefit analysis. Ecological Economics 22, 123-140.

Keeney, R., Raiffa, H., 1993. Decisions with multiple objectives: preferences and value tradeoffs. Cambridge University Press, UK (Original work published 1976).

Keeney, R.L., Gregory, R.S., 2005. Selecting attributes to measure the achievement of objectives. Operations Research 53, 1-11.

Kiker, G.A., Bridges, T.S., Varghese, A., Seager, P.T.P., Linkov, I., 2005. Application of multicriteria decision analysis in environmental decision making. Integrated environmental assessment and management 1, 95-108.

Kim, J.B., Hobbs, B.F., Koonce, J.F., 2003. Multicriteria Bayesian analysis of lower trophic level uncertainties and value of research in Lake Erie. Hum. Ecol. Risk Assess. 9, 1023-1057.

Kjærsgaard, J., 2007. Multiple criteria decision making in fisheries management: Experiences and implications. Food Economics – Acta Agriculturae Scandinavica, Section C 4, 49-64.

Lahdelma, R., Salminen, P., Hokkanen, J., 2000. Using multicriteria methods in environmental planning and management. Environmental Management 26, 595-605.

Lippitt, C.D., Rogan, J., Toledano, J., Saylgermano, F., Eastmana, J.R., Mastro, V., Sawyer, A., 2008. Incorporating anthropogenic variables into a species distribution model to map gypsy moth risk. Ecological Modelling 210, 339-350.

Liu, S., Hurley, M., Lowell, K.E., Siddique, A.-B.M., Diggle, A., Cook, D.C., 2011a. An integrated decision-support approach in prioritizing risks of non-indigenous species in the face of high uncertainty. Ecological Economics 70, 1924-1930.

Liu, S., Sheppard, A., Kriticos, D., Cook, D., Liu, S., 2011b. Incorporating uncertainty and social values in managing invasive alien species: a deliberative multi-criteria evaluation approach. Biological Invasions 13, 2323-2337.

Liu, S., Walshe, T., Long, G., Cook, D., 2012. Evaluation of Potential Responses to Invasive Non-Native Species with Structured Decision Making. Conservation Biology 26, 539-546.

Lodge, D.M., Williams, S., MacIsaac, H.J., Hayes, K.R., Leung, B., Reichard, S., Mack, R.N., Moyle, P.B., Smith, M., Andow, D.A., Carlton, J.T., McMichael, A., 2006. Biological invasions: Recommendations for US policy and management. Ecological Applications 16, 2035-2054.

Maguire, L.A., 2004. What can decision analysis do for invasive species management? Risk Analysis 24, 859-868.

Major, D.C., Cicchetti, C., Davis, R., Hanke, S., Haveman, R., 1975. Appraising Proposed Federal Standards for Water Resources Investment. Science 187, 79-81.

Marshall, G.R., McNeill, J.M., Reeve, I.J., 2011. Economics for accountability in commnity-based envrionmental governance. Institute for Rural Futures, University of New England, Armidale, NSW.

Messner, F., Zwirner, O., Karkuschke, M., 2006. Participation in multi-criteria decision support for the resolution of a water allocation problem in the Spree River basin. Land Use Policy 23, 63-75.

Munda, G., Nijkamp, P., Rietveld, P., 1994. Qualitative Multicriteria Evaluation for Environmental-Management. Ecological Economics 10, 97-112.

Neves, L.P., Martins, A.G., Antunes, C.H., Dias, L.C., 2008. A multi-criteria decision approach to sorting actions for promoting energy efficiency. Energy Policy 36, 2351-2363.

Pearce, D., Atkinson, G., Mourato, S., 2006. Cost-benefit analysis and the environment: recent developments. OECD.

Pheloung, P.C., Williams, P.A., Halloy, S.R., 1999. A weed risk assessment model for use as a biosecurity tool evaluating plant introductions. Journal of Environmental Management 57, 239-251.

Pielke Jr, R.A., 2007. The hnest broker: Making sense of science in policy and poltics. Cambridge University Press, Cambridge, UK.

Pohekar, S.D., Ramachandran, M., 2004. Application of multi-criteria decision making to sustainable energy planning - A review. Renew. Sust. Energ. Rev. 8, 365-381.

Prato, T., Herath, G., 2007. Multiple-criteria decision analysis for integrated catchment management. Ecological Economics 63, 627-632.

Raiffa, H., 1968. Decision analysis: Introducing lectures on choices under uncertainty. Addison-Wesley, Reading.

Rauschmayer, F., Wittmer, H., 2006. Evaluating deliberative and analytical methods for the resolution of environmental conflicts. Land Use Policy 23, 108-122.

Regan, H.M., Colyvan, M., Markovchick-Nicholls, L., 2006. A formal model for consensus and negotiation in environmental management. Journal of Environmental Management 80, 167-176.

Rosenberger, R.S., Stanley, T.D., 2006. Measurement, generalization, and publication: Sources of error in benefit transfers and their management. Ecological Economics 60, 372-378.

Schmidt, M.G.W., Lorenz, A., Held, H., Kriegler, E., 2011. Climate targets under uncertainty: challenges and remedies. Climatic Change 104, 783-791.

Smallwood, K.S., Salmon, T.P., 1992. A RATING SYSTEM FOR POTENTIAL EXOTIC BIRD AND MAMMAL PESTS. Biological Conservation 62, 149-159.

Soma, K., 2006. Natura economica in environmental valuation. Environmental Values 15, 31-50.

Spash, C.L., 2008. Deliberative monetary valuation and the evidence for a new value theory. Land Economics 84, 469-488.

Strijker, D., Sijtsma, F.J., Wiersma, D., 2000. Evaluation of nature conservation - An application to the Dutch Ecological Network. Environmental & Resource Economics 16, 363-378.

Sugden, R., 2005. Integrating cost-benefit analysis and mult-criteria analysis of flood and coastal erosion risk management projects. British Department for Environment, Food and Rural Affairs.

UK Department for Communities and Local Government, 2009a. Multi-criteria analysis: a manual, London.

UK Department for Communities and Local Government, 2009b. Multi-criteria analysis: a manuel, London.

Vatn, A., 2009. An institutional analysis of methods for environmental appraisal. Ecological Economics 68, 2207-2215.

Von Vinterfeldt, D., Edwards, W., 1986. Decision analysis and behavioral research. Cambridge University Press, Cambridge, UK.