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Deliberative Methods for Assessing Utilities
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Summary
The aim of this project was to review literature on biosecurity economics and the integration of economic analysis in biosecurity policy-making, and to investigate novel approaches to improve the uptake of quantitative information in invasive alien species risk management decisions. The project identifies deliberative multi-criteria evaluation as a useful tool with potential to generate substantial improvements in complex decision making involving the management of established species. This report provides a comprehensive literature review, an introduction to DMCE, an Australian case study in which this technique is trialled with a diverse decision-making group, and suggestions regarding the future use of DMCE as a decision facilitation device.

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Deliberative Methods for Assessing Utilities (0803)

David Cook, CSIRO

Final Project Report

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1. Executive Summary

1.1 Biosecurity economics

Uncertainty is a pervasive feature of biosecurity. Failure to deal effectively with uncertainty has tended toward ad-hoc reactive management responses. In order to move towards proactive management it is critical to understand the components of uncertainty associated with Invasive Alien Species (IAS) and communicate them effectively to the stakeholders before a collective decision can be made.

It is widely asserted that economic forces are the main driver of the worsening IAS problem, and that we therefore require economic solutions (Perrings, Williamson et al. 2002). However, economic analyses of IAS issues (i.e. biosecurity economics) is still very much in its infancy. Of the work economists have thus far carried out concerning IAS, one of three problems tend to restrict their use in risk management decisions:

(1) Past efforts focus mostly on partial damage estimation using one component of what is essentially a complex system (i.e. estimated agricultural impact, rather than social, environmental and agricultural impacts);

(2) While market and direct costs are well understood, non-market and indirect costs are not;

(3) Ex-post (after a species has invaded a country or region), rather than ex-ante (before invasion) evaluations have been favoured in the literature.

1.2 Inter-disciplinarity, public good and uncertainty

The invasive species problem is characterised by inter-disciplinarity, public good and uncertainty, and that these characteristics have in turn led to the three aforementioned problems. After reviewing existing literature within both Cost-Benefit-Analysis (CBA)/Cost-Effective-Analysis (CEA) and Multi-criteria Decision Analysis (MCDA) frameworks, we draw the following conclusions with regards to these issues:

- Invasive species should be regarded as part of human-ecosystem dynamics;
- Biosecurity policy-making would be more effective if it involves the public when making decision on public goods;
- Deliberative Multi-Criteria Evaluation (DMCE) techniques may be used to facilitate highly-complex policy decisions regarding invasive species, particularly those species with both market (e.g. agricultural, industrial) and non-market (e.g. environmental, social) impacts.

1.3 Case study: European house borer

A combined fuzzy set-DMCE method was applied to determine the appropriate regulatory response from the building industry in WA to the threat posed by the European House Borer (EHB). The fuzzy set approach reduces DMCE participants' cognitive burden in the evaluation process by allowing them to use linguistic terms to weight criteria. The DMCE offers a platform for scientists and other stakeholders to interact and to make a decision based on deliberation. Via open discussion the DMCE also successfully eliminated some linguistic uncertainties and ensured that the group as a whole shared the same understanding of each specific term.

The hypothetical challenge posed to the case study jury revealed some interesting insights into perceptions of EHB risk. The workshop participants felt that the process served to raise awareness, generate new ways of thinking, and give the group a common language. Based on these results the application of the combined technique in a biosecurity resource-allocation context warrants further investigation.

2. Introduction

Invasive Alien Species (IAS) and their associated damages impose significant financial costs to society. A recent US study showed that invading alien species cause losses adding up to almost $120 billion per year nationwide (Pimentel, Zuniga et al. 2005). In Australia loss to agriculture due to weed invasion alone was estimated as $3.9 billion per year (Sinden 2004). Moreover, the spread of invasive plants is now ranked second, behind species extinction, as the greatest threat to ecosystem functions worldwide (Millennium Ecosystem Assessment, MEA 2005).

Numerous terms have been used around biological invasions, including “non-indigenous”, “non-native”, ‘”alien”, “exotic”, “invasive”, “noxious”, “nuisance”, and “weed”. This proliferation of terms has caused considerable confusion and misuse of existing terminology[1]. The term ‘invasive’ in particular has been problematic as ecologists typically use it in reference to species which spread quickly and/or widely beyond the location of initial establishment, whereas in policy and legal documents it tends to imply negative effects caused to human beings even though invasiveness of a species does not necessarily predict its impact (Ricciardi and Cohen 2007).

For the purpose of this report an IAS is defined as a species that does not naturally occur in a specific area and whose introduction does or is likely to cause a net loss in social (including human health), economic or environmental welfare. Throughout the review we use the words “impacts” or “effects” without necessarily suggesting a negative connotation. We note in passing that most existing economic analyses focus on negative impacts of invasive species.

While the use of economic analyses to inform policies directed towards IAS management is relatively rare, the rate of invasion or introduction of species into new ranges globally continues to accelerate with growing trade and faster commercial transportation. It is widely asserted that economic forces are the main driver of this worsening invasive species problem, and that we therefore require economic solutions (Perrings, Williamson et al. 2002). However, the complexity of impact determination, particularly where environmental and social impacts must be considered, often limits the use of economics as a decision aid to agricultural impacts. Without the help of analytical tools, decision-making tends to be problematic, failing to countenance key stakeholder values and miscalculating event uncertainties (Lahdelma et al., 2000).

In this report we explore the possible use of Multi-Criteria Decision Analysis (MCDA) designed to deal with mixed sets of data (both qualitative and quantitative) and take explicit account of both trade-offs and uncertainty (Mendoza and Martins, 2006; Wittmer et al., 2006). Deliberative Multi-Criteria Evaluation (DMCE) seeks to combine these facets of MCDA in providing analytical structure together with the benefits of citizen/stakeholder participation (Proctor and Drechsler, 2006). Compared to MCDA without a participatory component, DMCE offers an opportunity for diverse views to enter the process, for facilitating consensus-building and for initiating a dynamic process of social learning (Rauschmayer and Wittmer, 2006).

A DMCE applied to management options for European house borer in Western Australia is used as a test case for the technique. It is used to help a diverse group of industry and government stakeholders decide between a range of regulatory options that could be put in place to mitigate the spread and impact of the insect. We combine a fuzzy logic approach with the interactive decision aid used with the decision making group. The results we achieved we mixed. While DMCE showed considerable promise in this example, we experiences several problems related to workshop design. Most of these related to the use of fuzzy sets instead of natural units in scoring management alternatives, and imprecise definitions of several criteria.

[1] Discussions of terminology and related issues are available in Richardson et al. (2000), Mark (2000), Lodge and Shrader-Frechette (2003), Colautti and MacIsaac (2004), and Lodge and Williams et al. (2006).

On the basis of the literature review and case study, this report concludes with a number of recommendations relating to future use of DMCE as a decision facilitation technique. The process appears well suited to group decision making related to the management of established IAS, but care needs to be exercised in forming the management alternatives, criteria and the information supplementing scoring and weighting decisions. Rather than DMCE offering an alternative to traditional economic analysis (i.e. involving benefit cost analyses or cost effectiveness analyses), DMCE offers a complementary decision-making method capable of enhancing the communication of complex information to decision makers from diverse backgrounds.

3. Methodological Review

3.1 Complexities of biosecurity economics

3.1.1 Interdisciplinary nature and lack of systematic analysis

When considering biosecurity economics and the role economics plays in biosecurity decision-making, calculations of the costs of invasions readily spring to mind as *the* fundamental contribution of the discipline. But, economics is much more than just a method for calculating costs (or control benefit). It is *"a framework for understanding the complex causal interactions between human behaviour and natural processes…"* (Perrings, Williamson et al. 2002). Indeed many biosecurity economic papers have co-authors from ecology/biology (to take care of the natural process part) and economics (to deal with the human behaviour component).

However, there are problems to overcome before people from different backgrounds can work effectively together. Sometimes different academic disciplines lack a common language for communication (Bingham, Bishop et al. 1995). The word "value", for instance, has very different meaning for ecologists and economists (Farber, Costanza et al. 2002). The list of differences between these two groups is a long one. At an operational level, steady-state equilibrium and relatively large temporal scale data (often yearly) are the norms for economists. Ecologists are often more interested in abrupt changes beyond thresholds and their data are often at shorter temporal scale (daily or monthly) (Bockstael 1996). But, sometimes the time scales they are interested in are considerably longer (i.e. millions of years). In addition, spatial components are at least as important for ecologist as temporal ones but only recently have economists begun to focus on spatial part of the story (e.g. Wilen 2007).

With these differences in mind it is perhaps easier to understand why there has been little systematic economic analysis of species invasion, reflecting a *supply* side problem if we regard the interdisciplinary research team as the suppliers of a biosecurity economic analysis. However, problems may also exist on the *demand* side. Typically, policy-makers who demand biosecurity economic analyses do not consider invasive species as a part of an ecological system, but instead are concerned with their effects on one system component (e.g. cultivated crops) (Foxcroft 2004). In other words, the market for more comprehensive analyses would be a limited one, even if they were widely available. However, the complex nature of control programs focused on single species (particularly at low population levels) and the increasing pressure from new invasions is forcing policy makers to adapt new approaches where invasive species are accepted as part of the human-ecosystem dynamics.

3.1.2 Public goods and non-market valuation issues

A public good is defined as "*A commodity or service whose benefits are not depleted by an additional user and from which it is generally difficult or impossible to exclude people, even if people are unwilling to pay for the benefits* (p. 256) (Baumol and Blinder 2000)". Examples of public good include the national defence system, public roads, street lighting and biodiversity. One common concern about the provision of public goods is who provides them since they normally don't have a market price (i.e. providers can not exclude users from consuming the good). Therefore their provision can not be financed by private parties (Doering 2007), and government must pay for public goods if they are to be provided to a socially desirable level.

The management of invasive species is an international and frequently global public good (Perrings, Williamson et al. 2002). If we classify impacts of invasive species into economic, environmental, or social in nature[2], the last two types of impacts often touch the public good domain so it is difficult to quantify them in dollar terms. Consequently environmental (e.g. biodiversity loss) and social (e.g. quality of life) impacts are often labelled as indirect and non-market and then neglected in a CBA or CEA. Of course these impacts are sometimes intertwined. For instance, biodiversity loss also indirectly erodes social welfare though the lost of genetic information with potential pharmaceutical value. While most-policy makers, indeed anyone with a social conscience would regard such indirect and non-market effects as significant, a lack of quantifiable evidence frequently prevents their inclusion in economic analyses.

3.1.3 Uncertainty and lack of ex-ante, or pre-invasion analysis

Uncertainty is a pervasive feature of invasive management issues (Perrings 2005; Caley, Lonsdale et al. 2006; Touza, Dehnen-Schmutz et al. 2007), where either the probability distributions have not been assessed or they change over time (Ewel, O'Dowd et al. 1999). Uncertain parameters include arrival (Batabyal and Nijkamp 2007), demography and dispersal (Buckley, Brockerhoff et al. 2005) of invasive species, on-site plant biomass data (Rinella and Luschei 2007), rates of industry growth (de Wit, Crookes et al. 2001), discounting rate (Settle and Shogren 2004), and impacts of invasive species (Horan, Perrings et al. 2002) on the existing biosecurity economic analyses.
There are different types of uncertainty. *Risky* is a term often used to describe a situation in which possible outcomes and their probabilities are both known (e.g. throwing a dice or tossing a coin). Pure *uncertainty* occurs when we only know the possible outcomes but not the probabilities of these outcomes (e.g. estimating wildlife reproductive rates where we can not accurately predict the multitude of factors that affect the rates but we do know the range over which reproduction is possible). *Ignorance* or absolute uncertainty occurs when we do not even know the range of possible outcomes. Predicting the alternate state into which an ecosystem might flip when it passes an ecological threshold (e.g. global warming), and how humans will adapt, may be cases of absolute ignorance (Farley and Daly 2003).

In the case of invasive species we are often faced with a situation of ignorance (Williamson 1999)[3]. We have great difficulty predicting whether any human actions will result in introduction, naturalization and spread of an invasive species or whether a successful invader will have economically significant effects. For instance, the red fire ant *Solenopsis invicta* is an invasive species in the southern USA, and was deemed a nuisance to humans, an agricultural IAS and a threat to wildlife upon its arrival. Yet 12 years after fire ants invaded Texas, they became a "benign presence" (Strayer, Eviner et al. 2006).

Uncertainty and ignorance are likely to become more prominent in future in association with a wider range of global changes. Indeed, a major uncertainty in assessing patterns of invasion will be in predicting the "time bombs" or sudden non-linearity of invasions that occur in the context of global environmental change (Naylor 2000).

Given this situation it is difficult to predict things with an *ex-ante* (i.e. before an incursion event) study when so much uncertainty and ignorance is involved. Furthermore we usually become motivated to study invasions after a species has spread extensively (Parker, Simberloff et al. 1999). For these reasons there have been more *ex-ante* economic analyses conducted than *ex post* (i.e. after invasion) studies (Born, Rauschmayer et al. 2005). Exceptions include Higgins, Richardson et al. (1996), Sharov and Liebhold (1998) and Cook, Thomas et al. (2007).

[2] Economic impacts are those of direct consequences to humans, typically leading to monetary losses. Environmental impacts are those that affect ecosystem structure and function. Social impacts include human health, quality of life, cultural heritage, etc. (Charles and Dukes 2007).
[3] A recent paper found that high-impact invaders (i.e. those that displace native species) are more likely to belong to genera not already present in the system (Ricciardi and Atkinson 2004).

3.2 Solutions in biosecurity economics research

3.2.1 Ecosystem Service as a bridging framework for interdisciplinary research

The lack of systematic analysis in biosecurity economics is easily demonstrated by looking briefly at research into leafy spurge (*Euphorbia esula*) impacts conducted in North Dakota, USA. Globally, leafy spurge is perhaps one of the most intensively studied invasive plant species in terms of its economic impacts. Using a bioeconomic model developed in Leitch, Leistritz et al. (1994) and Leitch, Leistritz et al. (1996) to estimate the economic impacts of leafy spurge on grazing land and wildland[4] in a four-state region (Montana, North Dakota, South Dakota and Wyoming), Leistritz, Bangsund et al. (2004) estimated the effect of changing levels of leafy spurge infestation on land output (e.g. carrying capacity for grazing livestock and wildlife numbers supported). Changes in biophysical outputs were then used to estimate direct/primary economic impacts, and changes in grazing land output were used to estimate effects on livestock producers (reduced income) and local agribusiness firms (reduced sales or receipts). Similarly, reductions in wild land output were used to estimate changes in a subset of outdoor recreation expenditures and outlays necessary to mitigate damages from runoff and soil erosion. The secondary economic impacts[5] were estimated using input–output analysis. The total (direct plus secondary) economic impacts measure the effects of leafy spurge infestations on the economy of the four states in the northern Great Plains region.

Leafy spurge infestations on grazing land were estimated to result in a loss in regional grazing capacity sufficient to support a herd of 90,000 cows. Direct economic impacts on stock growers, landowners, and agribusiness firms were estimated to exceed $37 million annually, whereas secondary impacts totalled almost $83 million. Losses on wild land were $3.4 million and $6.4 million per year for the primary and secondary impacts, respectively. Total impacts (primary and secondary) for the four state region were calculated to be $129.5 million annually (in 1993 USD, see Figure 1 for summary).

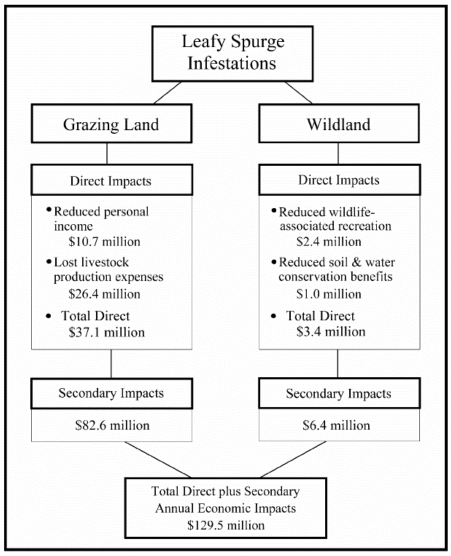


Figure 1. Annual economic impacts of leafy spurge in Montana, North Dakota, South Dakota and Wyoming 1993 (Leistritz, Bangsund et al. 2004).

[4] Wildland was defined as land not classified as urban or build-up, industrial or agricultural land. Wildland include forest, range, or recreation areas.
[5] Those resulted from the direct/primary effects through the multiplier process.

Using the same bioeconomic model, Bangsund, Leistritz et al. (1997), Bangsund, Leistritz et al. (1999) and Bangsund, Nudell et al. (1999) predicted future economic impacts of biological control of Leafy Spurge for the four states in year 2025. After biological control, rangeland output of leafy spurge was assumed to be 75% of its pre-infestation output and in the case of wild land, the output assumed a 100% return of pre-infestation outputs. In addition, 65% of total future leafy spurge was assumed to be controlled with biological agents in year 2025. Direct economic impacts from control were estimated to total about $19.1 million annually, and secondary impacts were estimated at $39.3 million, for a total annual economic impact of $58.4 million (in 1997 USD).

Bangsund, Nudell et al. (1999) extended the economic analysis using sheep as a biological control agent to improve grazing output for cattle in leafy purged infested ranchland. A bioeconomic model incorporating relationships between sheep grazing, leafy spurge control, grass recovery and forage use (by cattle) was developed to evaluate the viability of using sheep to control leafy spurge (see Figure 2). Costs and benefits of using sheep control were discounted over 5-year, 10-year and 15-year periods[6]. A number of scenarios were used to evaluate the returns of adding a sheep enterprise to existing ranches to control leafy spurge. Infestation cover scenarios included 5, 15, and 30 percent, which represented low (17%), moderate (50%) and high (100%) per unit area grazing losses for cattle.

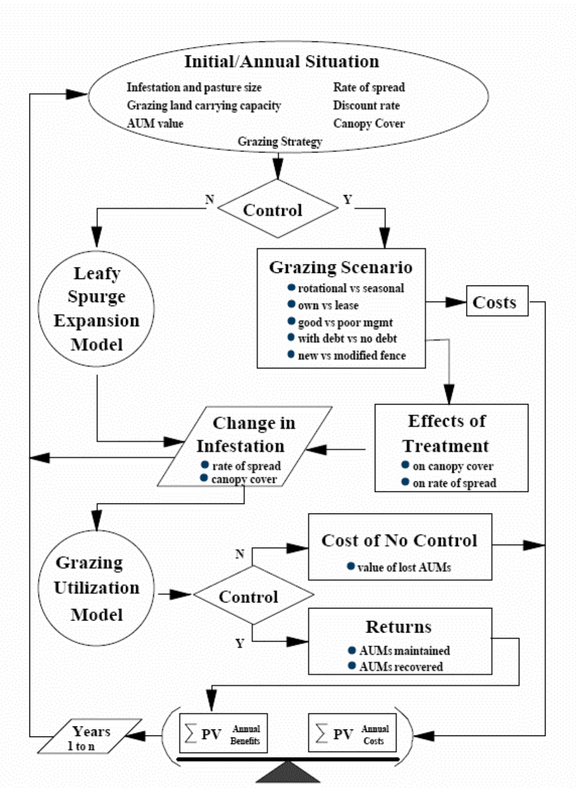


Figure 2. Bioeconomic model of the control of leafy spurge using sheep grazing (Bangsund, Nudell et al. 1999)

This series of state-of-the-art studies demonstrate the tendency for existing CBA/CEA efforts to focus on impacts to specific industries by a single species. The loss related to agricultural production was studied most often although secondary impacts to conservation and recreation were also estimated through an Input-Output analysis. But, little attention was paid to the interaction between leafy spurge

[6] A discount rate of 4% was used in this study.

and native species[7] and the impacts of invasion to the environment. This is perhaps attributable (at least in part) to the interdisciplinary nature of invasive species issues, which makes it difficult to integrate information between natural and social sciences. As a result few systematic analyses exist.

A bridging framework is required to structure the connection between the information flows, and *Ecosystem Services* have been proposed to fill this role (Binimelis, Born et al. 2007). The term "ecosystem services" first appeared in Ehrlich and Ehrlich's work (1981). It was popularized by Daily (1997) and Costanza and Folke (1997). Recently it was also employed by Millenium Ecosystem Assessment (MEA) as its main conceptual framework (MEA 2003). *Ecosystem services* are the benefits people obtain from ecosystems (Costanza and Folke 1997; Daily 1997; MEA 2003). These include provisioning services such as food and water; regulating services such as regulation of floods, drought, land degradation, and disease; supporting services such as soil formation and nutrient cycling; and cultural services such as recreational, spiritual, religious and other nonmaterial benefits (Figure 3, from MEA 2003).

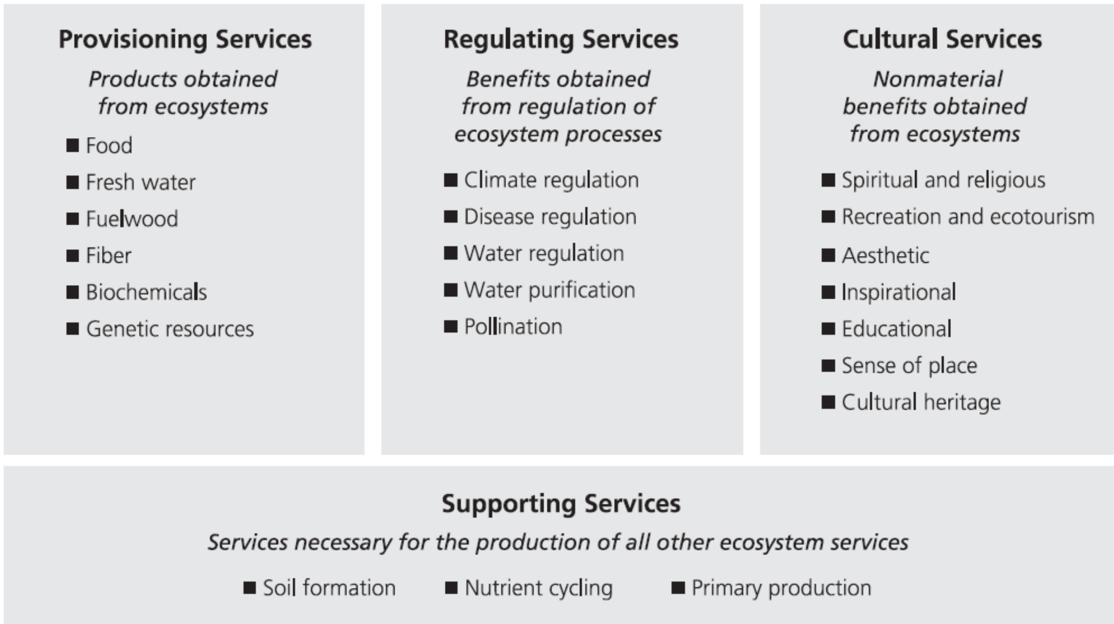


Figure 3. Four categories of ecosystem services (MEA 2003).

The concept of ecosystem services has been proven useful for at least two reasons. First, it helps synthesize essential ecological and economic concepts, allowing researchers and managers to link human and ecological systems in a viable and policy relevant manner. Second, scientists and policy makers can use the concepts to evaluate economic and political tradeoffs in a CBA/CEA framework (Costanza, Wilson et al. 2007).

Although CBA/CEA results can be embedded within an *ex post* ecosystem service framework, few studies have attempted this. Zavaleta (2000) is one example. Here the author firstly identified three major ecosystem services affected by *Tamarix* invasion of the riparian ecosystems: water provision, flood control and wildlife. Secondly, the annual monetary benefits of replacing *Tamarix* with native vegetation to each service was estimated using a benefit transfer approach (See Box 1 for more about benefit transfer). Finally, Net Present Values (NPV) of the eradication program was derived (0% discount rate applied). Results showed that the presence of *Tamarix* will cost an estimated $7~16 billion in lost ecosystem services over the next fifty-five years.

[7] Recently there have been several papers on the interaction between invaders and native species (Barbier 2001; Knowler 2005; Finnoff 2005; Gutierrez 2005).

There are many pathways by which invasive species can impact ecosystem services, but the most frequently estimated are market, or direct impacts (Charles and Dukes 2007). This is perhaps not surprising due to the difficulty of assigning values to the other types of ecosystem services (which mostly are public good, and hence fall into the indirect and/or non-market categories). A common practice in the literature is to include non-market and/or indirect values in a theoretical model but neglect them in the following quantification (e.g. Barbier 2001). But, the research question remains: how do we value ecosystem services that don't have a direct price signal in the market?

BOX 1: BENEFIT TRANSFER

Benefit transfer is defined as the adaptation of existing ecosystem service value information or data to new policy contexts which have little or no data. The transfer method involves obtaining an estimate for the value of ecosystem services through the analysis of a single study, or group of studies, that have been previously carried out to value "similar" goods or services in "similar" locations. The transfer itself refers to the application of derived values and other information from the original 'study site' to a 'policy site' which can vary across geographic space and/or time (Brookshire and Neill 1992, Desvousges, Naughton et al. 1992). For example, an estimate of the benefit obtained by tourists viewing wildlife in one park (study site) might be used to estimate the benefit obtained from viewing wildlife in a different park (policy site).
Over time, the transfer method has become a practical way of making decisions when primary data collection is not feasible due to budget and time constraints (Moran 1999). Primary valuation research is always a "first-best" strategy in which information is gathered that is specific to the location and action being evaluated. However, when primary research is not possible or plausible, then benefit transfer, as a "second-best" strategy, may inform evaluation of management and policy impacts. For instance, the US Environmental Protection Agency (EPA) regulation development process almost always involves benefit transfer. Although it is explicitly recognized in the EPA's *Guidelines for Preparing Economic Analyses* (US Environmental Protection Agency 2000) that this is not the optimal situation, conducting an original study for anything but the most significant policies is typically cost prohibitive (Griffiths 2002).

3.2.2 Non-market valuation tools for estimating impacts on ecosystem services

Because there are no markets for most ecosystem services, there are no observable prices. Consequently, a suite of valuation techniques have been developed to value them (Freeman 2003; Champ, Boyle et al. 2003; US National Research Council, 2005). These include both non-monetizing valuation methods within the multi-criteria decision analysis framework as well as conventional economic techniques within the CBA/CEA framework (Box 2).

The traditional economic tools used for non-market valuation include stated and revealed preference techniques. The critical distinction among these economic valuation methods is based on the data source, that is, whether they come from observations of people's behaviour in the real-world (i.e. *revealed-preference approaches*) or from people's responses to hypothetical questions (*stated-preference approaches*) such as "How much would you be willing to pay for a reduction in invasive species damage?"

When an ecosystem service is difficult to value using any of the above methods, researchers (mainly ecologists) have resorted to using the method of *replacement/avoided cost* (e.g. de Wit, Crookes et al. 2001). However, economists believe these cost-based approaches should be used with great caution, if at all (Shabman and Batie 1978; Bockstael 2000; US National Research Council 2005). This is because any value estimates derived from them should be on the cost side of the benefit-cost ledger, not counted as a benefit, and the conditions under which these cost estimates can serve as a last resort proxy are often too rigid to be met.

Box 2 illustrates some valuation tools are more appropriate for an ecosystem service than for others. For example, Travel Cost (TC) is primarily used for estimating recreation values while Hedonic Pricing (HP) is used for estimating property values associated with aesthetic qualities of natural ecosystems. Contingent Valuation (CV) and Conjoint Analysis (CA) are the only methods available that can measure non-use values like existence value of wildlife[8]. In many applications, multiple ecosystem valuation techniques involving multiple respondents will be required to account for total value of ecosystem services affected by invasive species. Only then CBA/CEA calculations will provide guidance for determining a broadly acceptable strategy for controlling invasions (Naylor 2000). Unfortunately, this is not possible in most cases due to time and budget limitations and lack of know-how.

Brown, Lynch et al. (2002) highlighted the significance of omitted non-market (specifically environmental) ecosystem service values. Here, two possible biological control methods for controlling Pierce’s disease in wine grapes were examined. Growers can increase profits either by planting barriers next to a source area to block insect movement into the vineyard or by clear-cutting the source of the disease. The research found the clear-cut policy is optimal only if the value of the environmental benefit of barrier vegetation is more than $5,500. But, since little is known about society’s willingness to pay for riparian vegetation the “optimal” strategy might not lead to a social optimum.

The difference the inclusion of social values can make to an invasive species control assessment is further demonstrated in Milon and Welsh (1989). Using a CV approach control of the aquatic plant Hydrilla (*Hydrilla verticillata*) was shown to be economically valuable to both interstate people and local anglers sport fishing on Lakes Harris and Griffin in Lake County, Florida, USA. This is reflected by fishers’ willingness to pay (WTP) for different levels of control. Average WTP for Hydrilla control ranged from $19.13 to $27.67 per person for a comprehensive control plan (A) and from $13.56 to $18.11 per person for a partial control plan (B)[9] (1989 USD). The higher bound values are associated with Lake County residents’ value and lower bound with out-of-state anglers. Aggregate WTP of all anglers was $175,840 for plan A and $119,362 for plan B. Interestingly, the WTP of anglers from Lake County was approximately 50% of the total. This indicates that non-residents (from out-side Lake County and outside Florida) had a significant interest in, and place a high value on, Hydrilla controls in the lakes.

[8] The concept of economic value is much more inclusive than many people realise. Much of what is typically considered non-economic value, like moral and bequest values, are in fact to some degree captured by “existence value”.

[9] For Plan A, Hydrilla would be controlled so that only a few small isolated patches are present in areas of the lakes with water depths less than 5 feet. No Hydrilla in boat ramp areas. For plan B Hydrilla would be allowed to cover many areas of the lakes less than 5 feet deep. Hydirlla would be mixed with other aquatic plants in these shallow areas. Some Hydrilla would be grown in boat ramp areas.

BOX 2: NON-MARKET VALUATION TOOLS

CBA/CEA framework
Revealed preference approaches

- Market methods: Valuations are directly obtained from what people must be willing to pay for the service or good (e.g., timber harvest).
- Travel cost: Valuations of site-based amenities are implied by the costs people incur to enjoy them (e.g., cleaner recreational lakes).
- Hedonic methods: The value of a service is implied by what people will be willing to pay for the service through purchases in related markets, such as housing markets (e.g., open-space amenities).
- Production approaches: Service values are assigned from the impacts of those services on economic outputs (e.g., increased shrimp yields from increased area of wetlands).

Stated-preference approaches

- Contingent valuation: People are directly asked their willingness to pay or accept compensation for some change in ecological service (e.g., willingness to pay for cleaner air).
- Conjoint analysis: People are asked to choose or rank different service scenarios or ecological conditions that differ in the mix of those conditions (e.g., choosing between wetlands scenarios with differing levels of flood protection and fishery yields).

Cost-based approaches

- Replacement cost: The loss of a natural system service is evaluated in terms of what it would cost to replace that service (e.g., tertiary treatment values of wetlands if the cost of replacement is less than the value society places on tertiary treatment).
- Avoidance cost: A service is valued on the basis of costs avoided, or of the extent to which it allows the avoidance of costly averting behaviours, including mitigation (e.g., clean water reduces costly incidents of diarrhoea).

MCDA framework
Individual index-based method, including rating or ranking choice models, expert opinion.
Group-based methods, including voting mechanisms, numerical aggregation, focus groups, citizen juries, and stakeholder analysis.

Nunes (2004) also looked at the private and social willingness to pay for the control of Harmful Algal-Bloom species (HABs). HABs are invasive exotic species that are primarily introduced in North European waters through ballast water of ships. The economic value of a marine protection program, including non-market benefits associated with beach recreation, human health and marine ecosystem impacts, was estimated with a joint Travel Cost-Contingent Valuation (TC-CV) survey undertaken at Zandvoort, a famous beach resort in the Netherlands. According to the TC model estimates, if the beach was closed to visitors for an entire year due to HABs the total recreational welfare loss equalled €55 per individual per year. The contingent valuation estimates indicated that the annual WTP amounts to €76 per respondent. The comparison of the TC and CV estimates implied the importance of marine ecosystem non-market benefits because the CV result mainly referred to non-market impacts caused by HABs. The economic value of the marine protection program was estimated between €225-326 million per year (Nunes 2004).

While techniques like CV can be used to measure differences in private and social WTP, it remains difficult to interpret this information. Generally, people tend to be more averse to a loss than they are attracted to an equivalent gain (Coursey, Hovis et al., 1987; Knetsch and Sinden, 1987; Kahneman, Knetsch et al., 1990). So, there tends to be a disparity between an individual's *willingness to pay* to prevent environmental damage and their *willingness to accept* compensation for that damage. The disparity between the two can be reduced with repeated experimentation, but this makes the process of revealing environmental values extremely costly (Portney, 1994).

Nevertheless, growth in the environmental valuation literature in the advent of the Exxon Valdez disaster in 1989 has been unprecedented (Adamowicz, 2004)[10]. But, significant though this body of work is, it is of very little use in terms of quantifying invasive species impacts. A number of problems with stated preference techniques have been identified and discussed, many relating to the tendency of respondents to act strategically when expressing their preferences. The willingness of respondents to pay for specific environmental or social goods may be embedded within answers expressed to surveyors if those answers reflect a broader set of goods. For instance, when asked to express a willingness to pay for the avoidance of damage to a specific environment amenity, a person may give answers reflecting their general desire to protect the environment. They are therefore biasing responses in a conscious or unconscious attempt to ensure the results lead to more environmental protection, rather than the significance of a particular environmental component.

There are several additional reasons why results should be treated with caution when working in invasive species space. Firstly, environmental effects attributable to invasive species often involve changes in the population or health of an environmental resource, rather than its complete destruction. Eliciting values for these marginal changes is yet to be attempted, and simply taking an aggregate value and extrapolating ignores changes at the margin resulting from scarcity. Secondly, the WTP to protect an environmental good (or to guard against changes in its wellbeing) can not be explained without understanding the sociological elements involved in that agent's decision-making process[11]. Factors such as age, income and background can have a dramatic influence on willingness to pay. A related issue involves the non-use values associated with environmental amenities[12]. While an agent may not receive tangible benefits from knowing these amenities are in a state of 'health', they may respond to questions to enjoy the "warm glow" of contributing towards environmental welfare (Kahneman and Knetsch, 1992). This becomes particularly complex when the concepts of irreversibility and irreplacability are considered. Finally, the resources required to accurately calculate the true value of environmental externalities are often prohibitive.

3.2.3 System modelling and incorporating uncertainty

Traditionally, the uncertainty described in 3.3 has not been considered in a comprehensive manner in CBA/CEA studies. In a survey of 27 economic assessments of biological control programs, Hill and Greathead (2000) found that although the vast majority of studies had a benefit-cost ratio larger than

[10] The *Exxon Valdes* was an oil tanker which ran aground in Prince William Sound, Alaska in March 1989. The resulting 30 million US gallons of crude oil that poured into the Sound affected over 1,900km of coastline, and had a devastating impact on resident wildlife. Exxon spent US$2 billion on the clean-up, and a further US$1 billion on penalties. The clean up operation, which involved the use of high temperature and high pressure spray equipment also caused additional environmental damage. Contingent valuation was used to derive an estimate of total damage resulting from the spill of US$287 million, and punitive damages of US$5 billion. In the process, a lively debate ensued concerning the reliability of these estimates and of the survey approach in general. The appeals process continues. Non-market valuation has remained one of the most subscribed areas of economics.

[11] The income elasticities for environmental goods are thought to be large and positive. Comprehensive empirical evidence for such a pattern of income elasticity is currently lacking (Whitby, 2000; Waage, Fraser et al., 2005).

[12] Values can be derived for environmental amenities from the cost of 'using' them (e.g. recreation, sport and tourism), but there are also 'non-use' values to consider. These include existence, moral and bequest values (mentioned above) that depend on the continued existence of the amenity and extend over generations in time. These non-use values make valuation extremely difficult.

1[13], very few had attempted to estimate the variability surrounding point-estimates of a benefit:cost ratio [14]. It is therefore impossible for decision-makers to make an informed judgement about the explanatory power of the analyses, or the appropriate level of confidence that should be placed in the results. This situation is gradually being changed with formal economic frameworks for risk management being put forward in the literature to address uncertainty issues. Shogren (2000), for instance, developed an optimal control model for reducing risks from invasive species by characterizing uncertainty through probabilities (i.e. treating (pure) uncertainty as essentially the same as risk and then risks could be reduced by either mitigation or adaptation). But, a practical limitation of these risk-based models is that it may be difficult to assign a probability to a one-time event such as the entry, establishment, spread and impact of invasive species, without historical precedent (Gren 2008).

Several studies attempt to address this problem in different ways. Eiswerth and van Kooten (2002) use an expert judgment questionnaire to assign invasive yellow starthistle (*Centaurea solstitialis*) infestation rates as one of the four possibilities, minimal, moderate, high and very high. Using stochastic dynamic programming they then analyse the control of the weed in California and compare the efficiency of five management options. Cook, Thomas et al. (2007) develop a stochastic bioeconomic model to predict the economic impact of the varroa bee mite (*Varroa destructor*) to the ecosystem service of pollination, and apply a combined probability of entry and establishment using a uniform distribution. Ten thousand iterations are then run with values randomly sampled across the range of each distribution using Monte Carlo simulations to represent uncertainty in the arrival process. Rinella (2007) adapt hierarchical Bayesian statistics to quantify uncertainty related to local and regional plant abundances and impacts of leafy spurge (*Euphorbia esula* L.). Without such a hierarchical approach 19 non-hierarchical models for each local site would have been constructed and sample-to-sample variation within each site would have been ignored. In contrast the hierarchical model employed a probability distribution of each site mean.

However, Horan, Perrings et al. (2002) argued that in the face of ignorance, where neither the range of possible outcomes or the possibility of these outcomes are known, decision models based on standard expected utility theory or Bayesian methods have limited value. They developed a model where policy makers were assumed to be risk averse. Their result showed that under ignorance it is optimal to devote more resources to confronting high-damage events that are considered possible even if the probability is considered to be low (i.e. low potential surprise), and to allocate few or no resources to confronting events that are considered less possible (high potential surprise). Addressing this issue, Moffitt and Osteen (2006) developed a model based on the minimax criteria. According to the model loss-aversive policy makers seek to minimize their maximum possible loss. Therefore, a policy option with the greatest difference between estimated damages and costs of action would be selected. The minimax/relative cost approach has an advantage over risk management based evaluations if decisions have to be made under ignorance[15].

Uncertainty research is critical to biosecurity issues since decisions often have to be made under risk, uncertainty, and even ignorance (Horan and Lupi 2005), and ex ante research is in great demand (Perrings, Williamson et al. 2000; Raghua, Dhileepan et al. 2007). As Born, Rauschmayer et al. (2005) reveal in a recent review of the biosecurity literature, the existing small set of ex-ante studies generally employ system models. This modelling approach offers at least three advantages:

1. It is not restricted by the *status quo*. In biosecurity economic analysis there is often empirical difficulty in collecting the necessary information. In contrast, system models permit the calculation and comparison of an essentially unlimited range of measures, because they are not subject to the logistic constrains of collecting empirical data (Parker, Simberloff et al. 1999), and

[13] Only 1 out of the 27 studies has a ratio of 0.99.
[14] This is not to say control programs themselves have a high success rate. To the contrary, most attempts at classic biological control are failures or have adverse side effects (Hill and Greathead 2000).
[15] Moffitt, Stranlunc et al. (2006) propose a more general model designed for solving the uncertainty problem.

can instead use data from other systems and situations. By running scenario specific analysis, for instance, results of different management options could be compared and then the most effective strategy could be selected.

2. It has the flexibility to incorporate the entire invasion process, including both ecological and economic components (i.e. results from non-market valuation) (Leung, Lodge et al. 2002). Furthermore, it also can incorporate human action towards bio-invasion which could be an important feedback in such a model (Finnoff, Shogren et al. 2005).
3. It permits uncertainty to be included in the analysis either by running sensitivity analysis[16] for parameters associated with uncertainty (Pimentel, McNair et al. 2001; Cook, Thomas et al. 2007) or by incorporating results from other techniques designed for tackling uncertainty issues, such as Bayesian (Rinella and Luschei 2007) and neural network analysis (Worner and Gevrey 2006).

An example of an *ex ante* (i.e. pre-invasion) invasive species analyses involving a system modelling approach appears in Stansbury (2002), in which the probability of Karnal bunt (*Tilletia indica*) entering and establishing in Western Australia is explored. The likely spread, containment and consequent economic impact of the pathogen to the State's grains industries are simulated, and the benefits of different risk mitigation strategies estimated. A sensitivity analysis shows that increase in quarantine funding can reduce the entry probability from one entry per 25 years to one in every 50 years, and the establishment probability from one every 67 years to one every 100 years. The economic impact ranges from 8% to 24% of the total value of wheat production depending on the resources allocated for detection and the spread rate of the pathogen.

Another example appears in Raghua (2007) where a life-cycle model for chrysomelid beetle (*Charidotic auroguttata*) is developed within the STELLA software environment to predict the risks and benefits of introducing the beetle to control the invasive liana *Macfadyena unguis-cati* in Australia. Using an environmental standard (which in itself is difficult to establish), the model predicts that risk to the non-target plant becomes unacceptable when the ratio of target to non-target species in a given patch ranged from 1:1 to 3:2. This simulation result was used to identify regions where the biocontrol agent might pose an unacceptable risk.

3.2.4 Cost Benefit Analysis (CBA) and Cost Effectiveness Analysis (CEA)

The traditional and most widely used method in biosecurity economics is cost-benefit analysis (CBA), which requires that the expected present value of the benefits (of any control program) be no less than the expected present value of the costs. Cost effectiveness analysis (CEA) has a similar structure, although only the costs of alternative means of achieving a previously defined set of objectives are analysed, and generally the lowest cost method(s) preferred. So, essentially CBA provides an answer to the question "should we take action?", and CEA answers the question "what action should we take?"

If the information on invasion impact is presented solely as a list of consequences in physical terms, then we encounter the classic problem of comparing apples and oranges. The purpose of CBA/CEA is to make the economic, environmental and social impacts comparable to each other, using a common metric. A strength of CBA and CEA analyses is that they break down the multidimensionality of invasive species' impacts into one-dimensional estimate in dollar terms (Born, Rauschmayer et al. 2005).

However, this strength can also be a weakness, especially at large spatial and temporal scales because large-scale studies may be confounded by spatial gradients or temporal trends in the environment such as climate change. For example, the costs of invaders to the American economy from two studies, Office of Technology Assessment (1993) and Pimentel, Zuniga et al. (2005), have a difference of two

[16] Sensitivity analysis rarely applies to ecological behaviour (Born 2005).

orders of magnitude. Perhaps the numbers generated in these studies do not mean much by themselves but do offer a general indication of both the scale of the problem (Perrings, Dalmazzone et al. 2005) as well as the level of difficulty encountered in biosecurity economic analyses

CBAs have also been used to assess the net gains or costs that may occur as a result of commencing trade with international sources in various commodities, but they have not followed a consistent format. Those studies that have been completed tended to follow narrow terms of reference set by high-profile sources concerning specific quarantine decisions as opposed to the broader social welfare implications of policy options (Nunn 2001; Roberts 2001). The way in which the economic implications of imports have been estimated appears to have been done on a case by case basis, rather than using a standardised method. Case studies have used a variety of economic analyses, including those that simply assume an outbreak scenario only affecting producers, those that seek to put a probability on this occurrence, those considering both consumer and producer impacts, or combinations of these (Cook and Fraser 2008).

Hinchy and Low (1990) addressed a New Zealand request made in 1989 to export apples to Australia, where the major disease transference concern was Fireblight, a disease caused by the bacteria *Erwinia amylovora* that affects apples and pears. Australia's detailed response to this request, coordinated by the Australian Quarantine and Inspection Service (AQIS), included an economic component (Hinchy and Low (1990)) which took the form of a benefit cost analysis comparing expected consumer and producer welfare changes resulting from relaxing quarantine laws protecting the apple industry. In 1995 New Zealand made another request to access the Australian apple market. This time the economic analysis by Bhati and Rees (1996) was quite different in approach. Expected consumer welfare change is not discussed. The analysis only considers possible producer surplus losses to pome fruit growers if a fireblight outbreak were to occur (Cook and Fraser 2008).

A market access application concerning salmon products from New Zealand, potentially forming an exposure pathway for Whirling Disease of salmon, also prompted an analysis of economic consequences by McKelvie (1991), which uses a deterministic model. This analysis builds an entry scenario involving the introduction of whirling disease to three prominent Tasmanian fisheries and derives possible damage estimates. Neither the likelihood of disease arrival, the effect on domestic salmon consumers, nor the likelihood of scenario occurrence is discussed. Following a similar market access request from Canada a second economic analysis was prepared, McKelvie, Reid et al. (1994). This analysis dealt with two salmon diseases considered an importation risk, Furunculosis and Infectious Haematopoietic Necrosis (IHN). Again, the analysis includes a gross estimate of producer welfare loss in the event of a disease incursion, rather than a net welfare assessment (Cook and Fraser 2008).

Applications by the U.S.A., Denmark, Thailand and New Zealand to export chicken meat to Australia were the topic of another economic impact assessment. The potential economic implications of importing from these countries were examined in Hafi, Reynolds, et al. (1994), which used one potentially imported disease (Newcastle disease) to illustrate the possible consequences of relaxing quarantine protocols. The method used in this analysis is similar to that of Hinchy and Low (1990) in that a critical probability of disease arrival is determined which brings the benefits and probable costs of trade into balance (Cook and Fraser 2008). Trade benefits were calculated as the change in consumer welfare resulting from lower domestic prices for chicken products, while the costs calculations were based on a severe Newcastle disease outbreak scenario causing a contraction in domestic supply of close to 20 per cent (Cook and Fraser 2008).

The analysis presented in James and Anderson (1998) focused on Australia's ban on international banana imports. It compared consumer surplus losses resulting from import protection to a hypothetical producer surplus loss induced by a relaxing of trade restrictions. Here, the consumer gains are shown to outweigh production losses, casting doubt over the validity of the ban in terms of net social welfare (Cook and Fraser 2008). This analysis was not prompted by a market access request, rather it was designed to highlight possible problems in the application of sanitary and phytosanitary measures in accordance with the World Trade Organization's SPS Agreement.

3.3 Setting the Scene for Multi-Criteria Decision Analysis

3.3.1 On the Need for Decision Facilitation

Given the different methodologies, models and techniques economics has presented in the literature to help us make resource allocation decisions for invasive species, how do we actually go about using them in the decision-making process? This is not a straightforward question when we consider that decision-making groups, be they in government, NGOs or the private sector, seldom represent a single discipline or perspective. More often these groups house a diversity of opinions, expertise, knowledge and experience, not necessarily including economics or social science. This can make it difficult for technical analysts to communicate their results to the group in a way they can understand, and in a form easily used in the decision-making process.

In the most comprehensive review of the Australian biosecurity system to date, Nairn, Allen et al. (1996, the so called Nairn review), it is clearly stated that communication forms a critical part of risk analysis. Nairn, Allen et al. (1996) defined the process of risk analysis as comprising of three parts:

(a) Risk Assessment – the process of identifying and estimating risks associated with a policy option and evaluating the likely consequences of taking those risks;

(b) Risk Management – the process of identifying, documenting and implementing measures to reduce these risks and their consequences; and

(c) Risk Communication – the process of interactive exchange of information and views concerning risk between analysts and stakeholders (Nairn, Allen et al. 1996; Nunn 1997).

This asserts that a successful risk assessment should exhibit each of these components if it is to facilitate a socially acceptable allocation of relatively scarce resources. The Nairn review went on to list several fundamental principles to be included in the analytical process, which included: stakeholder/industry consultation; objectivity and robustness in scientific methodology and political independence; transparency; consistency and harmonisation; subject to appeal on process, and; subject to periodic external review (Cook 2002).

The task of resource allocation is particularly complex in cases where regulatory measures such as quarantine or invasion responses protect non-market (e.g. environmental) as well as market (e.g. agricultural) goods. Environmental decisions are particularly complex, multi-faceted, and involve a variety of stakeholders with different priorities or objectives (Linkov, Varghese et al. 2004). In these cases, economic analyses using a narrow single commodity method of assessing risk must be supplemented by other information. Generally, the difficulties involved in quantifying the non-market impact of IAS (described above) prevent their inclusion in classical economic analyses of quarantine strategies. However, if policies directed by such analyses are to reflect social welfare preferences, a more formal recognition of potential non-market damage is needed.

In addition to environmental consequences of invasion, other non-market goods that receive little attention in the literature but often need to be considered by policy-makers involve the socio-economic disposition of rural communities. But, as with environmental amenities, quantifying these effects is difficult. In the same way an environmental resource may have an existence or moral value, so too might a rural community. As such, a majority of the community may be willing to pay to preserve it even if they spend most of their time in urban areas and have little social or economic ties to rural communities. Bennett, van Bueren et al. (2004) presents evidence to this affect in three very different regions of rural Australia[17].

Animal welfare is also emerging as a non-market good requiring greater attention, particularly in the wake of the 2001 foot and mouth disease outbreak response in the United Kingdom. Here, the rules of the Office International des Epizooties (OIE) (or the World Organisation for Animal Health) necessitated a mass culling as a disease response. This distressful situation was made worse by an inflated compensation schedule which led to over-application for payments and competition between legitimate claimants and those reacting to financial incentives (Whiting, 2003). The non-market values associated with animal welfare were not used to influence the response policy. Evidence presented in Frank (2008) suggests positive income elasticities for animal welfare (i.e. the wealthier we are the more animal welfare we demand), possibly attributable to scientific, philosophical and theological advances over the past 30 years, as well as increased numbers of companion animals in the developed world.

Given the complications of taking into account all market and non-market impacts of invasive species in a single measure of impact required by CBA or CEA, Multi-Criteria Decision Analysis (MCDA) techniques may offer a practical solution to the dilemma facing biosecurity policy makers. Rather than striving for definitive proof of the right decision, MCDA can be used to stimulate discussion amongst the decision-making group about possible resource allocation choices, trade-offs and uncertainties. Instead of the exclusive use of quantitative estimates of non-market policy implications, semi-quantitative estimates can be used to make decision-makers aware of the full consequences of their decisions.

In the following sections we provide background information on the growth of MCDA as a decision aid, and work towards a technique allowing group participation in the resource allocation process.

3.3.2 Aiding complex decision-making and biosecurity

Perhaps the most important and challenging aspect of biosecurity is the need for rapid response and change. Standards applied to the environment change with time as we learn more about it and as social values change. Biosecurity institutions therefore need to be able to keep pace. However, the sheer quantity of biophysical and socio-economic data can quickly overwhelm stakeholders who are trying to make sense of natural resource-related issues (Hajkowicz, Young et al. 2000). In responding to invasive species outbreaks, complexity and uncertainty lead to more difficult decision-making and justification of selecting a course of action. Contrary to the reductionism view, systems thinking aids in the understanding of the linkages and interactions between the elements that comprise the whole system (Sposito, Faggian et al. 2007).

To help guide decision-makers in effectively resolving complex biosecurity issues, an *evaluation method* is required to transform broad policy goals into conclusions or agreements (Munda, Nijkamp et al. 1994). Evaluation methods fall under the broad category of systems methodologies which encompasses a variety of methods that include rational and ordered steps grouped in stages, and take a range of alternative perspectives into account (Sposito, Faggian et al. 2007). Emerging as extremely useful evaluation methods over the past half a century is a suite of tools and techniques collectively termed *Multi-criteria Decision Analysis*.

Multi-criteria Decision Analysis (MCDA) has been developed and refined within many different disciplines including operations research, decision theory, management science, regional planning, economics, policy analysis, psychology and marketing research. MCDA had its origins as a structured

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

decision-making aid in the late 1950s in the area of Operations Research. During the late 1960s, multiple criteria evaluation methods became popular and emerged as a distinct class of decision-making techniques which moved away from the optimisation of single argument objectives to that of multiple argument objectives (Nijkamp, Rietveld et al.1990). During the 1970s there was a huge growth in the number of works in both the theory and application of multiple criteria methods (Rietveld 1980, Nijkamp, Rietveld et al. 1990).

Within a historical context, the use of MCDA within an environmental policy context has been relatively new. In Australia it was trialled in application to forest policy by the Resource Assessment Commission (RAC 1992d) and there have been other applications to water catchment issues (Llewellyn 1985) and natural resource planning (Proctor 2001; Hajkowicz 2000; Gomez 2000; Robinson 1998). An IAS categorisation process is carried out within Biosecurity Australia's Import Risk Assessment methodology to identify species associated with a region and product that require in-depth examination in the risk analysis (Australian Government Department of Agriculture, Fisheries and Forestry 2007). The use of deliberative decision-making processes in biosecurity or invasive species management strategies had not been explored prior to Cook and Proctor (2007).

3.3.3 Form and extent of multi-criteria decision analysis

Terms such as *multi-criteria decision analysis* and *multi-attribute decision-making* are used interchangeably (see for example: Resource Assessment Commission 1992d; Yoon and Hwang 1995). The terms *multiple objective decision methods* and *multi-objective decision-making* have also been used interchangeably to refer to those techniques which involve assessment of continuous alternatives and incorporate mathematical optimisation methods (Kazana 1999, Rietveld 1980, Yoon and Hwang 1995). Here, we use the term Multi-Criteria Decision Analysis (MCDA) to refer to a suite of approaches designed for situations where more than one objective or criterion must be considered within a decision-making context, and where there are a finite number of pre-determined and discrete options from which to choose.

MCDA is based on the premise that decision making can be improved by making the criteria for decision-making explicit and ranking each option according to how well it satisfies each of the criteria. 'Options' can be almost anything among which we wish to distinguish, be it management options, investment portfolios, points in time or space, job candidates, refrigerator models, or in this case invasive species management strategies. Distinguishing among options can take many forms. For example, it can involve simply selecting the best, ranking all from 'best' to 'worst' or determining if a decision will be 'acceptable'.

Before embarking on an MCDA, it is important to clearly define objectives and outcomes and to carefully articulate the questions to be addressed and the nature of the options being distinguished. More subtle but crucial questions about whose problem is being addressed, the number and type of decision-maker/s and the skills and experience of the decision analyst (running the MCDA) are equally important to address before embarking upon an MCDA. In general, basic steps described in section 6 (below) are followed by the decision-maker and the analyst in all MCDA techniques. Once a specific form of MCDA has been chosen by the analyst, the first step involves identifying feasible options involved in the decision problem. The choice of options (or scenarios) and overall objectives can be developed by various sources including an expert or lay jury, expert advice, computer simulation models, and/or political processes (Proctor and Drechsler 2006). Next, the decision-maker seeks to identify the overall objective that is to be achieved in the process and then identifies the criteria by which to judge the selected options. An important part of the process is then to apply appropriate weights to each of the criteria. These reflect the particular preferences of the decision-maker/s in how important each criterion is in relation to the overall objective. The next step is to assess each of the options. This is done by examining how each option performs under the different criteria and the weightings. The sensitivity of the ranking of options can then be estimated with respect to the chosen weights and method of aggregation. The whole process may be repeated with ‘fine tuning’ by the decision-maker to aspects related to his or her input. This part also allows for any tradeoffs to be identified in the overall decision process.

Section 3.4 provides more detail of the individual steps to be followed in an MCDA.

3.4 Key steps involved in a multi-criteria decision analysis

3.4.1 Choosing the options and the objectives

The choice of options and of the overall objective or objectives are important and closely related steps in any decision-making process. The objectives and options are chosen by the decision-maker/s, with potential input from expert advice, computer simulation models or political and/or legislative prerequisites. The objective can be as broad as necessary, but in the case of multiple decision-makers, overall agreement should be reached. The options could reflect each of the preferred scenarios of the decision-makers or could be based on an amalgamation of plans of the decision-makers. Massam (1988) suggests a benchmarking approach as a framework for the options to reflect a brad range of possible solutions which should include:

- the status quo (e.g. do nothing)
- an ideal best plan (spend whatever is required to prevent/eradicate invasive species)
- an hypothetical worst plan (e.g. remove current biosecurity practices & protocols)
- a plan of minimum satisfaction (e.g. targeted eradication in some areas)

The options should be sufficient in number to represent a realistic and varied selection, as a starting point, for the decision-maker but should not be too numerous to make the analysis unwieldy or unnecessarily complex. It should be kept in mind that the options may be altered and ‘fine-tuned’ as part of the iterative process that follows.

3.4.2 Selecting the criteria

The decision-maker(s) usually select the criteria. The criteria are designed to compare and assess each of the options and therefore must relate to the overall objective of the decision-making task. Initially, criteria can be very broad and then broken down into components or sub-criteria and even lower level sub-criteria. Ideally, the lowest level of the criteria structure are those which are measurable (quantitatively or qualitatively) and are commonly referred to as indicators.
In general, the criteria should:

- be *complete and exhaustive* in that they cover all possible aspects of the decision-making problem and make the analysis complete. Basically, the criteria reflect the ‘trade-offs’ involved in the decision making process.
- *contain mutually exclusive* (non-redundant) items so as to prevent ‘double counting’ of aspects of the decision-making problem and to better allow ‘trade-offs’ to take place. This essentially means that the preferences associated with the impacts of options can be assigned independently from one criterion to the next. A simple check for this is to assign preference scores for the options on one criterion without knowing what the options’ preference scores are on any other criteria? If the answer is yes, then this criterion is preference independent of the others. This condition must be met for any MCDA if a weighted average summation approach to combine preferences across criteria is to be used.
- be *clearly defined* and directly relevant to the defined problem. Because it is often necessary to break criteria down into sub-criteria in order to make meaningful measurements,
- be *decomposable* into smaller measurable units. For example, a criterion such as ‘quality of life’ may be measured as an index based on the sub-criteria of level of income, access to health care and level of education.
- be *minimal* so that no other smaller set of criteria can be measured.

- be *restricted* so that weighting the criteria does not become unmanageable or difficult.

To allow adequate understanding of the decision problem and to assist in achieving a smoothly run process, most practitioners regard 7 to 12 criteria as the maximum for a MCDA process (Yoon and Hwang 1995). In contrast, the Analytic Hierarchy Process (AHP) facilitates the use of more criteria through imposing a hierarchical structure (i.e. the criteria can be broken down into smaller components of sub-criteria).

3.4.3 Weighting the criteria

In Multi-criteria analyses, the preferences of the decision-maker/s are accounted for by the weighting or scoring placed on each of the criteria and sub-criteria. These weightings may range from equal importance of all criteria, to a ranking of most to least important or to a relative weighting of all criteria. The weights may be qualitatively expressed, quantitatively expressed or a mixture of both. In analyses which involve many different decision-makers, this step is critical because it allows stakeholders to express differing views explicitly and it helps identify those areas which are of most personal importance and which warrant careful investigation and possible trade-off solutions. Different multi-criteria techniques employ different methods for extracting weightings from the decision-maker. All techniques should, however, be sympathetic to the comparative abilities of the human brain:

> "When devising methods for formulating and assessing preferences, one has to take into account the limitations in human capabilities for undertaking such endeavors. Psychological research on decision-making reveals that people have limited capacities concerning the number of conceptual units that can be handled at a certain point in time…" (Nijkamp, Rietveld et al. 1990, p. 40).

Weighting techniques can be divided into direct and indirect estimation methods. *Direct estimation* is designed to estimate the exact worth of one item compared to another item in the weightings (for example, the Rating Method, the Ranking Method and Paired Comparisons). *Indirect estimation* techniques may include information on weights used in past studies (see for example Nijkamp, Rietveld et al. (1990)).

The particular weighting method chosen should be appropriate for the particular decision makers involved. For example, some people may be more relaxed and comfortable with using a ranking method rather than putting weights directly on each of the criteria. Different cultural considerations should also be taken into account.

3.4.4 Evaluation of the options

The options are assessed in two stages: first, by how important each of the criteria and sub-criteria are to the decision-maker (the weight vector $\boldsymbol{w}$) and second, by how well each option rates in terms of each of the criteria and sub-criteria of assessment.

The second stage is displayed by means of an Impact Matrix where each of the components represent the evaluation or impact of the options according to the individual criteria (e.g. Table 1). In general, if one option performs better than another for all of the criteria then that option will be ranked highest. If the performance varies for different criteria, i.e. one option performs better for some but not other criteria in comparison to the second option, then its ranking will depend on how highly the superior performing criteria are weighted by the decision makers.

The final ranking of each of the options is then calculated by a mathematical operation using the Impact Matrix and the criteria/sub-criteria weights. The form of this mathematical operation (often referred to as the 'aggregation procedure', Munda, Nijkamp et al. 1994) often describes the particular type of MCDA employed.

Table 1. Example impact matrix for an IAS management problem

Criteria	Alternative			
	1 No logging in any area	**2** Reduced areas available for logging	**3** Current area available for logging	**4** Increased areas available for logging
Area of native habitat retained (%)	100	86	65	42
Quality of water (index)	95	77	68	60
Change in wage levels ($)	-234 000	-45 000	0	137 000
Change in tourism revenue ($m)	37	15	0	-10
Change in jobs (number)	-34	-6	0	7
Access to bushwalking (ha)	124 000	67 000	54 000	23 000

3.4.5 Sensitivity analysis

Although not always undertaken, particularly in MCDAs that pursue a single optimal outcome, sensitivity analysis of the results is an extremely useful part of the MCDA process. It may be conducted in order to take into account uncertainty in estimation of values or weightings and may provide a range that can be statistically analysed. It can also consider the effects of different techniques used in the weighting procedure. A sensitivity analysis adds greatly to the overall understanding of complex decision problems and feeds directly into the iterative process that allows for refinement and improvement of the finally chosen option. The impacts of the various options under different criteria may fall within a statistically estimated range that can be incorporated into the sensitivity analysis.

It is very important that the sensitivity of outcomes can be tested for different values of the most crucial and contentious criteria, weightings and impacts. For example, in a group decision-making situation, if it were found that there was a great disparity in preferences for a certain criterion that reflects a crucial trade-off, then it may be enlightening to find out how the overall results change with the changes in preference levels for this criterion. If the results are not greatly affected, then the criterion can take less importance in the overall process and the decision-makers can concentrate on other criteria and trade-offs. If the results are extremely sensitive to this criterion, then closer scrutiny should be given to it by confirming values and measurements.

The sensitivity analysis is given a dominant role in a technique incorporating multiple decision-makers and risk analysis called Multi-criteria Mapping (Stirling and Mayer 1999). ‘Mapping’ refers to that part of the analysis where the results are expressed in terms of various, systematically applied, sensitivities with ‘… prescriptive conclusions being drawn only conditionally, by reference to the clearly-defined perspectives taken by different participants’ (Stirling and Mayer 1999, p. 69). Another example of sensitivity analyses accounting for risk and uncertainty involved with the data uses Monte Carlo simulation to estimate probability distributions for underpinning the ranking of options (Van Delft and Nijkamp 1977).

3.4.6 Iterating and fine-tuning

The analyst can achieve greater understanding of the decision-making problem by interacting with the decision-maker to allow further iterations in the analysis if necessary, to identify where trade-offs can be made to concentrate on the important issues in the process and finally to re-define criteria and options to take into account what has been learnt from the process. As mentioned previously, the initial list of options should be viewed as a starting point with the subsequent MCDA process contributing to the development of the most suitable option. In group decision-making situations, this step can be crucial if the ultimate aim of the analysis is to reach some compromise or agreement on the outcome. Often, interaction and further iterations can be facilitated by the use of computer software models that allow for faster manipulation of the data.

3.5 Risk Communication and Deliberative Multi-Criteria Evaluation (DMCE)

Uncertainty is a pervasive feature of IAS management (Perrings, 2005; Caley, Lonsdale et al., 2006; Touza et al., 2007). Components of epistemic uncertainty include the likelihood of arrival (Batabyal and Nijkamp, 2007), demography and dispersal of IAS (Buckley, Brockerhoff et al., 2005), biomass potential (Rinella and Luschei, 2007), rates of industry growth (de Wit, Crookes et al., 2001), and impacts of IAS (Horan, Perrings et al., 2002). Traditional biosecurity economic analyses require application of a discounting rate (Settle and Shogren, 2004), the specification of which entails uncertain social judgments. In addition, language-based uncertainty is a common feature in scientists' attempts to communicate analyses to the public.

These uncertainties are likely to become more prominent in future in association with a wider range of global changes. Indeed, a major uncertainty in assessing patterns of invasion will be in predicting the "time bombs" or sudden non-linearity of invasions that occur in the context of global environmental change (Naylor, 2000). The importance of explicit consideration of uncertainty in environmental decision-making is being increasingly recognised (Morgan and Henrion, 1990); Halpern, Regan et al. 2005; Georgiou 2008). Burgman's proposal of 'honest and complete' risk assessment, for instance, emphasized a science-based, stakeholder-involved, transparent and holistic approach as part of decision-making processes (Burgman, 2005).

Few studies to date have taken an integrated approach that identifies and incorporates all sources of uncertainty into the decision-making process, though many studies have focussed on the identification and quantification of specific components of uncertainty (Ascough II, Maier et al, 2008). One of the major shortcomings of existing efforts is they mainly focus on either quantifying or communicating uncertainties and this dichotomy leads to a separation between risk analysis and risk management. There is no precedent from which decision-making groups can draw inspiration when given the task of making IAS management decisions using complex analytical information. Moreover, there are no guidelines to follow that ensure the uptake and retention of risk assessment information throughout the process of making management decisions.

Group discussions that lead to the resolution of a complex management issue can be effective exercises. Policy emerges from identifiable patterns of interdependence between key social actors (Considine 1994). There are advantages of group decisions over individual processes as more perspectives may be considered, there is a higher chance of having systematic thinkers involved, as well as deliberative and well-informed members (Linkov, Varghese et al. 2004). On the other hand, the involvement of different groups conveying a range of priorities and outlooks can make for a convoluted criteria selection and evaluation process (Dragan, Feoli et al. 2003). Groups are susceptible to the tendency of establishing ingrained positions or to prematurely adopt a status quo perspective that excludes contrary and often relevant information (McDaniels, Gregory et al. 1999). Due to this potential discrepancy in group opinion, effort must be committed to providing as much background knowledge and social context as logistically possible to the groups involved in the process (Dragan, Feoli et al. 2003). Participants should distinguish between interests which are their underlying concerns, and positions which are their stands on the issue being negotiated. By focusing on interests rather than positions, parties can engage in integrative bargaining and find creative ways to benefit all parties and help work toward consensus (Fisher and Ury 1991).

Stakeholder and decision-maker characteristics can be vast and varied to include experts, stakeholders, and the general public. There is a growing trend toward the use of participatory approaches, particularly in the public sector, to create a more democratic and open process (Gilmour and Beilin 2006). The rationale for stakeholder involvement in decision-making processes can be classified as being substantive, instrumental, or normative (Gilmour and Beilin 2006). Substantive reasoning to include stakeholders is simply that these players combine to form an otherwise absent multidisciplinary local knowledge base incorporating natural, physical, and social sciences, medicine, politics, and ethics (McDaniels, Gregory et al. 1999). The instrumental argument sees the involved parties being more likely to accept the decision outcome due to the transparency and inclusion of their respective voices and opinions within the negotiation process (Gilmour and Beilin 2006). The normative aspect is present due to the tendency for the issues to involve common resources, meaning group decision processes requiring a diverse mix of local people and knowledge are called for (Linkov, Varghese et al. 2004).

3.5.1 What is DMCE?

The DMCE method (Proctor and Drechsler 2006) combines the facilitation, interaction, and consensus-building features of citizens' jury processes (Crosby 1999, Dienel and Renn 1995), with the structuring and integration features of Multi-Criteria Decision Analysis (Massam 1988, Munda, Varghese et al. 1994, Proctor 2005). It has been developed for more effective engagement of multiple stakeholders in the decision making process, which overcomes some of the problems associated with MCDA which has been essentially developed for a single decision maker.

The citizens' jury involves around ten participants being charged with the responsibility of constituent representation and decision making. The group is presided over by an independent facilitator who ensures that participants have equal opportunity to express their views and the process is able to follow a suitable course to achieve outcomes. The group is encouraged to use expert witnesses, technical analyses and anecdotal information as required to form individual opinions. Time is then devoted to group discussion, information clarification, questioning of expert witnesses and debate in which group opinions are revealed and often modified. Although desirable, it is not necessary that a group consensus be reached for a preferred outcome to be forthcoming (Cook and Proctor 2007).
A jury is traditionally composed of a random sample of the population that will be affected by the decision, but often is represented by stakeholders, decision-makers, and/or experts in the case of environmental issues. Proctor and Drechsler (2006) selected natural resource managers who represented the decision-makers in the issue at hand, rather than randomly chosen community members, terming the process a `stakeholder jury'. Often, jury members are made up of "experts" who can be defined as having detailed knowledge, reliability, credibility, communication ability, and experience in the topic at hand (Burgman, Fidler et al. 2006).

Key features of MCE embedded in the DMCE approach include the simplification and truncation of complex decision problems using interactive decision support software. Using this software, the opportunity costs of different options can be explored and communicated to the group, helping them to consider the trade-offs associated with decisions. Sensitivity analysis of results can also be performed to indicate the most critical parameters in the decision-making process, and therefore, where supplementary information may be required to facilitate a desirable/defensible decision (Cook and Proctor 2007).

The objectives of the procedure as well as the decision criteria used in the deliberations on an optimum management scenario are developed by the group. Subsequently, an Impact Matrix is created that contains estimated impacts or scores on criteria to which each participant assigns priority weights. Once the criteria weights and Impact Matrix have been determined, a deliberative process is carried out with the aid of the facilitator and MCDA software. The software is used interactively during the process and the results of each iteration displayed to the participants. The objective of the deliberations is for the participants to come to an agreement on a set of weights for the decision criteria that would be used to determine an optimum management scenario (Cook and Proctor 2007).

3.5.2 Past applications of DMCE

3.5.2.1 Invasive Alien Species prioritisation in Western Australia

The use of MCDA in IAS prioritization was explored in a workshop in Perth, Western Australia (WA) in November 2005 (Cook and Proctor 2007). This involved a hypothetical resource allocation exercise in which decision-makers were asked to establish ten priority species according to a set of agreed criteria and criteria weights. The list that jury members were asked to consider comprised of species with a wide variety of impacts, ranging from species that are predominantly of agricultural significance, to those with substantial environmental or human health and wellbeing implications. The decision-making group comprised of representatives from a variety of government, industry and community groups that might be affected in the event of an IAS incursion (see Cook and Proctor (2007) for details).

In the weeks preceding the workshop participants were provided information about the purpose of the prioritisation exercise, DMCE methodology and format of the workshop. Scientific information about the nature and biology of the ten IAS was also provided and economic impact assessments for each. Questions and queries of a technical nature regarding this information were referred to specialist within the CSIRO and Department of Agriculture and Food Western Australia (DAFWA) for clarification, and explanatory/supplementary materials circulated to all the decision makers.
The initial phase of the workshop was used to outline objectives and give purpose to the DMCE process. The following set of environmental, economic and social criteria were agreed upon and an Impact Matrix constructed to assess the various species:

- Environmental
  - Likelihood of Arrival
  - Flora & Fauna
  - Ecological Linkages
  - Extinctions & Irreversibilities
- Economic
  - Local Economies
  - Production Costs
  - Yield Loss
- Social
  - Human Health
  - Cultural Loss
  - Political Imperative

With the assistance of an independent facilitator who promoted constructive debate through interactive software manipulation, the decision-makers scored species according to each criterion based on testimony provided by DAFWA entomologists, plant pathologists, ecologists and economists. To supplement their testimony, anecdotal information from areas where species are known to occur was also used to inform scores. Time was then devoted to discussion, information clarification, debate and consensus building within the group. The support software was used to facilitate discussion, providing a medium by which the opportunity costs of investment decisions could be explored, and sensitivities of different criteria communicated within the group. This helped participants to consider the trade-offs associated with decisions, and to identify where supplementary information may be required to facilitate a desirable/defensible decision (Cook and Proctor, 2007).

After reaching agreement on the impact matrix, participants were asked to indicate the relative importance they thought each criterion should carry. They were each asked to distribute 100 weighting units among the 10 criteria, which were then averaged and combined with the impact matrix to form a set of rankings for the EPPs. Whilst being mindful of a wide variation in weights, the initial preferences included a mix of environmental and agricultural IAS ranked highest. The next stage in the DMCE process involved participants being asked to try to reach a consensus on criteria weights in an effort to reduce ranking uncertainty, and more clearly identify priority species. Those criteria for which weights differed most significantly were discussed first, with jury members who had expressed the most extreme maximum and minimum weights for each criterion asked to defend their choices. During this review process, jurors could reflect on their choices and those of other jury members and to adjust their weights if they felt it was necessary (Cook and Proctor 2007). This revision process continued until participants were no longer willing to adjust their weightings. Priorities still contain a high level of uncertainty, but generally species of high social and environmental significance were ranked higher than others of a predominately agricultural significance.

Although an experimental study, Cook and Proctor (2007) proved effective enough to act as a catalyst for continued research into the use of DMCE in IAS prioritisation. The National Plant Biosecurity Cooperative Research Centre, Horticulture Australia Ltd. and the Rural Industries Research and Development Corporation initiated a joint project in January 2007 to further investigate the role of DMCE in plant biosecurity resource allocation decisions (Cook and Proctor 2007).

3.5.2.2 South Australian lead and zinc smelter improvement project

Another important application of DMCE involved the lead and zinc smelter in Port Pirie, South Australia, which has been operating for over 100 years and was facing a number of health and environmental related improvement challenges involving air, water, soils and wastes (Proctor, McQuade et al. 2006). In particular it had come under increasing surveillance from the South Australian Environmental Protection Authority (EPA), the South Australian Department of Health and the media due to increasing levels of lead being discovered in blood samples taken from children in the region and these high lead levels having potentially damaging effects on childrens' learning abilities.

A Deliberative Multi-criteria Evaluation was conducted with stakeholders in the region including residents, employees, smelter management, local council and other interest groups. First, researchers reviewed background material on the problems facing the smelter as well as the issues concerning local residents and the EPA and the availability of relevant data to assist in the deliberative process. Smelter staff were involved throughout the research process and particularly in the early stages to establish their exact needs and to determine the stakeholders and experts that they believe should be involved (in consultation also with the EPA). Then a set of discrete scenarios or options that they believed would represent a full suite of possibilities for the company over a period of time into the future was developed. The next step was to develop and pre-test a deliberative process with the assistance of the smelter employees and an outside facilitator was chosen to carry out the facilitator role. Once the participants were chosen, a mail-out questionnaire was posted to them to determine their views on the objectives of the process as well as the relevant decision criteria that could be used in their deliberations on a favoured scenario. Agreement was established and a second mail-out questionnaire was sent out to establish priority weightings for each of the criteria by each of the participants.

Next, an Impact Matrix, showing the estimated impact of each of the criteria under each of the chosen scenarios, was developed using the following criteria:

- Environmental
    - Ground level sulphur dioxide (SO_2) in air concentrations
    - Ground level lead (Pb) in air concentrations
    - Soil contamination by petroleum products
    - Soil contamination by heavy metals
    - Discharge of specific heavy metals from the site into the marine environment
    - Nutrient discharge into a marine environment
    - The management of mosquitoes
- Social/Cultural
    - Children's lead in blood

    - Odour resulting from SO_2
    - Noise resulting from the activities of the smelter
    - Visual fugitive emissions

- Economic
    - Financial costs
    - Benefits

The result was an agreement between the company, the community, the EPA and other government departments for a strategy that the company could follow to lesson the impact of emissions on the health of children without affecting employment at the smelter. Some suggestions included changing smelter operations depending on the wind direction as well as changing the location of school playgrounds. A community review committee that will meet every two years and use the DMCE process, was established to monitor the impacts and change the recommendations if necessary.

4. Case Study

4.1 Methodology

4.1.1 Purpose of using a practical example

DMCE has been applied in the natural resource management arena as a decision-aid tool (Bojorquez-Tapia et al., 2005; Hajkowicz and Collins, 2007). Only very recently have researchers used DMCE in Invasive Alien Species (IAS) risk management (Cook and Proctor, 2007). Here we present another case applying the DMCE technique in facilitating decision-making regarding IAS using a fuzzy set approach. Fuzzy set theory is based on a gradual transition from one class to another. Items can have partial membership in multiple sets. This method can be particularly powerful in handling uncertainty inherent in MCDA problems and it is the most commonly used ranking method in MCDA for water resource planning and management (Hajkowicz and Collins 2007).

Our study attempts to integrate quantification and communication of uncertainties. The combined DMCE and fuzzy system approach offers a platform 1) to demonstrate and inform the best science, including the uncertainties associated with scientific results to the stakeholders, 2) to get feedback from the stakeholders who may have better/more knowledge, and 3) to make a collective decision through interaction and consensus-building.

4.1.2 Fuzzy logic

The IAS management strategy performance evaluation problem involves a number (n) of discrete management options M_i ($i = 1, 2, \ldots, n$). These options are to be scored according to a set of m criteria C_j ($j = 1, 2, \ldots, m$), each of which is separated into p_j sub-criteria C_{ik} ($k = 1, 2, \ldots, p_j$) (Yeh et al., 2000). Since some of the relevant criteria (i.e. those dealing with non-market IAS impacts) can not be scored with the aid of quantitative information, qualitative assessments are to be given according to a set of linguistic terms with corresponding fuzzy membership functions. Separate linguistic terms are used to determine (a) appropriate scores for each management option against each sub-criterion, and (b) the relative importance of each sub-criterion in choosing the most appropriate management strategy (Yeh et al., 2000).

The linguistic terms defined in Tables 2 and 3 are used, each of which corresponds to a triangular fuzzy number representing their approximate value range between 1 and 9 (Juang and Lee, 1991; Yeh et al., 2000). The range is defined as (a_1, a_2, a_3), where $1 \le a_1$, $1 \le a_2 \le a_3 \le 9$. The values of a_1 and a_3 represent the lower and upper bounds of the fuzzy number, respectively, while a_2 is the most likely value of a linguistic term (Yeh et al., 2000).

The impact matrix can be expressed as:

$$X = \begin{bmatrix} x_{11} & x_{12} & \cdots & x_{1m} \\ x_{21} & x_{22} & \cdots & x_{2m} \\ \cdots & \cdots & \cdots & \cdots \\ x_{n1} & x_{n2} & \cdots & x_{nm} \end{bmatrix}. \qquad (1)$$

Here, x_{ij} indicate the linguistic scores for IAS management option M_i ($i = 1, 2, \ldots, n$) with respect to criterion C_j ($j = 1, 2, \ldots, m$) (Yeh et al., 2000; Chang and Yeh, 2002). Since sub-criteria C_{ik} ($k = 1, 2, \ldots, p_j$) are used for each criterion a more detailed impact matrix can be expressed as:

$$Y_{C_j} = \begin{bmatrix} y_{11} & y_{12} & \cdots & y_{n1} \\ y_{12} & y_{22} & \cdots & y_{n2} \\ \cdots & \cdots & \cdots & \cdots \\ y_{1p_j} & y_{2p_j} & \cdots & y_{np_j} \end{bmatrix}, \tag{2}$$

where, y_{ik} are the stakeholder jury's linguistic scores for the performance of management option M_i (i = 1, 2, …, n) with respect to sub-criterion C_{ik} (k = 1, 2, …, p_j) (Yeh et al., 2000). To truncate the prioritisation process a single consensus impact matrix was used in this study.

Table 2. Liguistic terms used to seed the impact matrix

Qualitative term	Very Poor (VP)	Poor (P)	Fair (F)	Good (G)	Very Good (VG)
Membership function (a_1, a_2, a_3)	(1, 1, 3)	(1, 3, 5)	(3, 5, 7)	(5, 7, 9)	(7, 9, 9)

(Yeh et al., 2000)

Table 3. Liguistic terms used to weight sub-criteria

Qualitative term	Least	Less	Fair	More	Most
Membership function (a_1, a_2, a_3)	(1, 1, 3)	(1, 3, 5)	(3, 5, 7)	(5, 7, 9)	(7, 9, 9)

(Yeh et al., 2000)

A weighting vector W_j (j = 1, 2, …, m) for the perceived importance of sub-criteria to a decision-maker in making an IAS management decision is revealed using the linguistic terms in Table 3, and is expressed:

$$W_j = \left(w_{j1}, w_{j2}, \ldots, w_{jk}, \ldots w_{jp_j}\right). \tag{3}$$

Here, w_{jk} are the fuzzy weights for sub-criteria C_{ik} (k = 1, 2, …, p_j).

Sub-criteria weighting vectors were elicited individually for each jury member. By combining the scored for each management alternative against each sub-criterion with the sub-criteria weights from one or more rounds of weighting by a stakeholder jury, each alternative can be ranked in order of preference to the jury. We employ the widely-used concept of the *degree of optimality* to establish clear and defined preferences. The optimal management alternative is the one that is both closest to the ideal solution and farthest from the negative ideal solution (Zeleny, 1982; Benıtez et al., 2007).

The first step in the ranking procedure involves the formation of a weighted fuzzy impact matrix through the multiplication of the criteria impact matrix X (i.e. (1)) with a consensus weighting vector W_j (i.e. (3)) (Yeh et al., 2000). A normalised preference function $\left(x_{1j}, x_{2j}, \ldots, x_{nj}\right)$ for criterion C_j with sub-criteria C_{jk} is given by:

$$\left(x_{1j}, x_{2j}, \ldots, x_{nj}\right) = \frac{W_j, Y_{C_j}}{\sum_{k=1}^{p_j} w_{j,k}}. \tag{4}$$

From the scores and sub-criteria weights expressed by a stakeholder jury, we can identify the IAS management alternatives with the maximum fuzzy preference value ($M_{\max}^k$) and the minimum fuzzy preference value ($M_{\min}^k$) with respect to each sub-criterion. The degree of preference for a single alternative with respect to each sub-criterion can then be calculated by comparing its weighted fuzzy performance with both $M^k_{\max}$ and $M^k_{\min}$, i.e.:

$$\mu_{M^k_{\max}}(x) = \begin{cases} \dfrac{x - x^j_{\min}}{x^j_{\max} - x^j_{\min}}, & x^j_{\min} \le x \le x^j_{\max} \\ 0, & \text{otherwise} \end{cases}, \quad (5)$$

$$\mu_{M^k_{\min}}(x) = \begin{cases} \dfrac{x^j_{\max} - x}{x^j_{\max} - x^j_{\min}}, & x^j_{\min} \le x \le x^j_{\max} \\ 0, & \text{otherwise} \end{cases} \quad (6)$$

where $j = 1, 2, \ldots, m$ and $k = 1, 2, \ldots, p_j$.

Expression (5) gives a performance rating for each management option for each criterion based on how far it is from the criterion-specific most preferable option. This reflects the status of any one management option relative to the management option with the highest combined weight and score. Yeh et al. (2000) describes this measure of preference as being appropriate where the attitudes of decision-makers to risk and uncertainty are "optimistic". In other words, the jury prefers the management option that most consistently exhibits a combined weight and score near the highest total for each criterion.

Conversely, expression (6) gives a pessimistic preference weighting in which the jury seeks the option furthest from the worst option (Chang and Yeh, 2002). It is appropriate where the stakeholder jury seeks the management option closest to the best according to each of the sub-criteria. This is the concept we will use in the case study that follows, but we note it is also possible to form a pessimistic preference rating in which a jury seeks the option furthest from the worst option (Chang and Yeh, 2002).

Equations (5) and (6) represent the preference limits for a jury. We represent this range of possible risk attitudes using an index (λ) with a value 0 or 1. $\lambda = 0$ and $\lambda = 1$ represent the optimistic or pessimistic view of jury members, respectively (Yeh et al., 2000). An optimistic jury member prefers higher values of the fuzzy sets, while a pessimistic member is more conscious of low values. So, in the context of the case study presented in this paper, an optimistic jury member pays more attention management alternatives with higher scores, while a pessimistic jury member is more concerned about lower scores (Chang and Yeh, 2002).

By summing performance ratings $\mu_{M^k_{\max}}(x)$ and $\mu_{M^k_{\min}}(x)$ across sub-criteria for each IAS management option we can form an index of the degree of similarity S_j between each option M_i and the fuzzy maximum and minimum, $M^j_{\max}$ and $M^j_{\min}$, for each criterion C_j, i.e.:

$$S_j = \lambda \sum_{k=1}^{p_j} \mu_{M^j_{\max}}(x) + (1-\lambda) \sum_{k=1}^{p_j} \mu_{M^j_{\min}}(x), \quad (7)$$

where $j = 1, 2, \ldots, m$. The higher the value of S_j the more preferable the management alternative is to the stakeholder jury with a mixture of optimistic and pessimistic attitudes with respect to the IAS. The positive and negative ideal management strategy for the jury (i.e. the positive and negative extremes of risk preference) can be stated as:

$$z^+ = \max. \sum_{k=1}^{p_j} \mu_{M^j_{\max}}(x) \quad (8)$$

$$z^{-} = \max.\sum_{k=1}^{p_j} \mu_{M_{\min}^{j}}(x) \quad (9)$$

In order to determine the overall performance of each management option the index S_j (from (7)) must be compared to both the ideal minimum and maximum. The difference between S_j, z^+ and z^- is referred to as the Hamming distance (Yeh et al., 2000; Chang and Yeh, 2002), and can be calculated as:

$$h^{+} = \sum_{j=1}^{m} (z^{+} - S_{M_j}) \quad (10)$$

$$h^{-} = \sum_{j=1}^{m} (S_{M_j} - z^{-}). \quad (11)$$

A crisp overall index of the relative preference for any management strategy M_i is then calculated as:

$$P_i = \frac{h_i^{-}}{h_i^{+} + h_i^{-}} \quad (12)$$

In the case study which follows P_i is expressed in percentage form (see Table 9).

4.2 European house borer in Western Australia

We conducted a hypothetical DMCE exercise to determine an appropriate regulatory response from the building industry in WA to the threat posed by the recently-introduced European House Borer (EHB). Although our case study involved a real IAS, a real set of alternative regulatory options and real community stakeholders it was not directly tied to a real response. The exercise we undertook was designed to trial the effectiveness of the DMCE technique as a means of resolving IAS management issues, and as a means of showcasing the methodology to a group of interested parties. Due to a tight workshop budget and the fact that the workshop was purely illustrative, it was necessary to take steps to condense the DMCE process into a single day.

The EHB was discovered on the outskirts of Perth in January 2004, but may have been present for ten or more years prior to detection. It is a destructive IAS of seasoned coniferous timber, including pine, fir and spruce, and is therefore capable of causing structural damage to buildings. Residences with untreated radiata pine, southern pine, Douglas fir, hoop pine or bunya pine frames are particularly at risk. Despite extensive surveying, EHB has only been found in several outer suburbs of Perth. A national cost-shared control program formally commenced in January 2007 under the Primary Industries Standing Committee (PISC) arrangement in which the Commonwealth paid 50 per cent of the costs of eradication while the remainder was funded by the WA State government.

To supplement this eradication campaign the WA Department of Housing and Works (DHW) commissioned a report investigating possible regulatory actions that could be taken to mitigate the impacts of the insect on the housing industry (The Allen Consulting Group, 2006). This report was intended to stimulate public consultation on possible regulatory options, but by November 2008 no regulatory actions had been decided on. The three management alternatives the report put forward for consideration were as follows:

(a) Do nothing

Under this scenario there are no additional building regulations put in place to guard against possible EHB damage. Hence, private home and business owners deal with effects of the insect in their own way;

(b) State-Wide Building Restrictions

All use of untreated softwood building materials in new homes and businesses will be restricted. Regulations would be put in place banning the use of untreated softwoods for structural purposes ensuring that houses are structurally protected from EHB infestation. The proposed regulations would be monitored and enforced by local government authorities as part of the process which they currently undertake in granting approval to building applications;

(c) Delimited Building Restrictions

The use of untreated softwood building materials in new homes and businesses will be restricted in areas where the Borer has been detected. Hence, the structural quality of new homes in affected areas will remain at a safe level while limiting compliance costs faced by the community relative to a state-wide approach. This alternative relies heavily on the assumption that EHB will not spread significantly beyond its current distribution.

The objective of our DMCE study was to evaluate these three regulatory options using a list of criteria developed in consultation with a stakeholder jury, and to decide on the most desirable option according to these criteria.

The jury was made up representatives from various community and business groups potentially affected by EHB. Approaches were made to local shire councils who would be in a position to enforce any building regulations agreed to; the State department of Housing and Works who commissioned The Allen Consulting Group report; the Department of Agriculture and Food - Western Australia who is currently involved in the control/eradication of EHB; building industry associations; State and local environmental conservation groups concerned with insecticide usage. However, an unfortunate set of unforeseen circumstances prevented the attendance of several groups who had indicated an intention to participate. Consequently, from a list of 15 agreed attendees, the jury on the day of the workshop totalled 10 people. The DMCE exercise was presided over by a professional facilitator, Ms. Christine Moro of Christine Moro & Associates Corporate Communications. The names and affiliations of attendees appear in Table 4. In the results to follow the identity of individuals and their corresponding preferences is kept confidential.

Table 4. List of Workshop Participants

Name	Organisation	Email
Moro, Christine (facilitator)	Christine Moro & Associates	christine@christinemoro.com.au
John Van Schagen	DAFWA	jvanschagen@agric.wa.gov.au
Mike Grimm	DAFWA	mgrimm@agric.wa.gov.au
Art Diggle	DAFWA	adiggle@agric.wa.gov.au
Nabil Yazdani	DHW	nabil.yazdani@dhw.wa.gov.au
Richard Olsen	City of Swan	richard.olsen@swan.wa.gov.au
Garry Fisher	City of Swan	garry.fisher@swan.wa.gov.au
Duncan Wilson	Shire of Kalamunda	duncan.wilson@kalamunda.wa.gov.au
Jonathan Smith	Shire of Kalamunda	jonathan.smith@kalamunda.wa.gov.au
Ted Skouros	City of Wanneroo	ted.skouros@wanneroo.wa.gov.au

Through our investigations and consultation with different stakeholder groups potentially affected by EHB regulations for the building industry three criteria (economic, social and environmental) were identified as being vital to a prioritisation process incorporating a range of sub-criteria. These included three economic, four social and one environmental sub-criterion. The authors provided jury members with suggested scores for each regulatory action alternative with respect to each of the criteria in order to shorten the workshop process. The jury then adjusted these suggested scores up or down depending on the view of one or more jury members who disagreed. These eight sub-criteria are discussed below:

Economic criteria

(1) *Compliance costs*
The costs of adhering to the proposed regulations. Compliance costs will initially fall onto home builders as the costs of building materials will increase. However, these costs will ultimately be passed on to new home buyers through increased prices (The Allen Consulting Group 2006).

(2) *Expected damage costs*
The damage costs of infested houses given the continuation of current EHB Program monitoring and containment activities.

(3) *Administrative costs*
Administrative costs include items associated with the design and implementation of regulations. In the case of the proposed regulations, administrative costs will fall on the State government in terms of the initial implementation of the regulations (i.e. amendments to existing legislation) and local government in terms of monitoring, enforcement and the assurance of compliance.
Table 5 shows quantitative data from The Allen Consulting Group (2006) relevant to each of the economic criteria, and the stakeholder jury's corresponding score for each management alternative using the scale defined in Table 2.

Table 5. Quantitative assessment data and corresponding qualitative assessment results for the *Economic* Criterion

Sub-Criterion	Management Alternative		
	Do Nothing	**State-Wide Building Restrictions**	**Delimited Building Restrictions**
Compliance Costs	$ - VG	$ 697,000,000 VP	$ 37,000,000 F
Expected Damage Costs	$ 120,000,000 VP	$ - VG	$ 1,000,000 F
Administration Cost	$ - VG	$ 52,000 G	$ 52,000 G

Social criteria

(4) *Reduction in infested houses*
This sub-criterion viewed in a social context takes into consideration the health and safety aspects associated with EHB-infestations. Banning the use of untreated softwoods that are used for structural purposes will ensure that houses built after the regulations are implemented are structurally protected from Borer infestation. The regulations would protect the health and safety of: (i) occupiers of new houses (i.e. the regulations will protect against structural roof collapse due to infestation and also ensure that new houses maintain structurally strong enough to withstand extreme climatic conditions such as wind gusts); and (ii) those working on new houses (i.e. workers such as electricians or roof tilers that are commonly required to work within roof spaces or on top or roofs will be protected from suffering injuries due to structural collapse caused by infestation when working on homes built with treated structural timbers).

It is difficult to accurately quantify the value of the health and safety benefits that would arise from mandating the use of treated softwoods as such a task relies on assumptions being made about the incidence of injury or harm attributable to Borer damaged timbers. However, the modelling undertaken by the Department of Agriculture indicates that the introduction of state wide building restrictions could decrease the number of infested houses from around 198,000 (over the course of 30 years) to 77 (Blanchard et al. 2005; The Allen Consulting Group 2006). Thus, the regulations could be seen to lower the risks of health and safety incidents occurring by a substantial degree. To this end, the health and safety benefits arising from the regulations on building materials are deemed to be significant.

(5) *Peace of mind*
The treatment of timbers to a level that would prevent EHB infestations may be a source of reassurance against attacks from other IAS. For instance, the treatments recommended for EHB also provide protection against termite infestations. This criterion captures the positive externality (or flow-on effect) from this additional protection.

(6) *Damage to iconic structures*
The structural quality and longevity of houses does pose significant public interest concerns, particularly as houses are significant cultural or personal assets. Modern churches, public buildings and works of art and other structures of social significance may also be put at risk by EHB infestation.

(7) *Unknown catastrophe*
This sub-criterion captures systemic risks brought about forces yet to be determined. The growing complexity of regulatory structures, corporations and institutions may make social systems less resilient to the impacts of a slow-spreading IAS like EHB in ways we are not yet aware of, but which could impose a significant cost on future generations.

Table 6 shows the jury's score for each management alternative using the scale defined in Table 2. Again, it should be noted that the authors provided jury members with suggested scores for each regulatory action alternative with respect to this criterion in order to shorten the workshop process.

Table 6. Qualitative assessment results for the *Social* Criterion

Sub-Criterion	Management Alternative		
	Do Nothing	**State-Wide Building Restrictions**	**Delimited Building Restrictions**
Reduction in Infested Houses	VP	VG	VG
Peace of Mind	P	VG	G
Damage to Iconic Structures	VG	VG	VG
Unknown Catastrophe	VP	G	G

Environmental criteria

(8) *On and off-site damage*
This criterion relates to chemical residue issues brought about through the increased use of insecticidal timber treatments. Chromated Copper-Arsenate (CCA) in particular leaches out of the treated timber over time so there can be residues of arsenic, copper and chromium on the surfaces of the wood. Timber treatment plants can be particularly contaminated (i.e. referred to by the jury as *on-site* contamination), but are covered by other environmental and chemical regulations. Nevertheless, an audit undertaken by the NSW EPA of five timber treatment plants found contamination through inadequate storage of materials and wastes at 5 plants, failure to maintain drains, dams or treatment facility at 4 plants, and inadequate surface water controls at 4 plants (NSW EPA, 2003).

The Scientific Committee on Toxicity, Ecotoxicity and the Environment (CSTEE) of the European Commission noted: 'There is extensive documentation of past substantial soil and groundwater contamination at wood treatment sites…There is also evidence in the published literature… that contamination of the soil and vegetation can extend to the area beyond the immediate boundaries of such sites, something that has been attributed to wind erosion, percolation, surface drainage as well as on-site incineration of wood waste' (CSTEE, 1998). Residues of arsenic, copper and chromium on the surfaces of the wood can be washed off by rain to accumulate beyond the confines of timber yards (i.e. referred to as *off-site* contamination of soil or groundwater). All three metals pose a potential threat to the environment. According to the US EPA: 'The amount and rate at which arsenic leaches, however, varies considerably depending on numerous factors, such as local climate, acidity of rain and soil, age of the wood product, and how much CCA was applied.' (Office of Pesticide Programs 2002).

Table 7 shows the stakeholder jury's score for each management alternative using the scale defined in Table 2. Once again, the authors provided jury members with suggested scores for each regulatory action alternative with respect to this criterion.

Table 7. Quantitative assessment results for the *Environmental* Criterion

Sub-Criterion	**Management Alternative**		
	Do Nothing	**State-Wide Building Restrictions**	**Delimited Building Restrictions**
On and Off-Site Damage	VG	F	G

Three rounds of sub-criteria weighting were carried out in the workshop to allow for constructive debate about the relative importance of each to different stakeholders within the group. This process involved participants clearly articulating their preferences relative to others. The sub-criteria weights given by the jury in each round are expressed using the terms described in Table 2. The linguistic results are given in Table 8 (p. 41). The authors concede that these weights were assigned by the jury without consideration of the range of impact scores. Hence, in instances where each regulatory option was assigned the same score for a single criterion, the weight of that criterion becomes irrelevant. In a more comprehensive DMCE exercise the criterion would in fact be removed from the assessment.

With the data contained in Tables 5-8 we can determine the crisp performance index (i.e. see equation (12), Section 4.1.1, p. 36) for each management alternative. Using a scale between 0 and 1, the cells of Table 9 (p. 42) indicate the Hamming distance for each management alternative. This reflects the performance of each option relative to the positive and negative ideal solutions for each criterion. The values indicated in each of the cells result from the summation of equations (10) and (11) in section 4.2. A crisp performance index and rank is shown for each alternative in the final two rows of the table. Note that the performance index (derived from equation (12) of section 4.1.1) is shown as a percentage.

It is immediately apparent from Table 9 (p.42) that the total performance index and the ranking of management alternatives changed very little across the three rounds of weighting. However, the process of deliberation between rounds one and two appears to have been the most significant in defining the preferred course of management action. Although equally as desirable to the *State-wide building restrictions* alternative in round one, preferences for the *do nothing* alternative were revised downwards in round two and remained low in round three. As indicated in Table 8 (p. 41), alterations were certainly made to sub-criteria weightings between rounds two and three, but these were not sufficient to produce a change in the ranking of management options.

Table 8. Weighting of the sub-criteria by members of the stakeholder jury

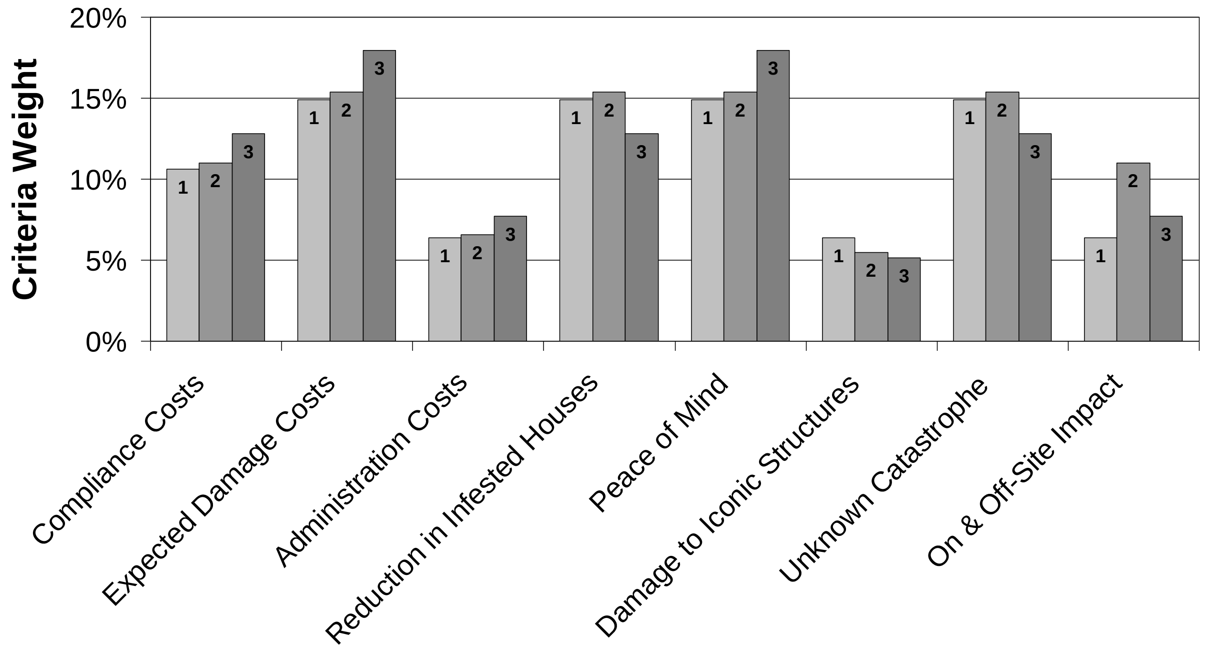
		Economic Sub-Criteria			Social Sub-Criteria				Environmental Sub-Criteria
	Jury Member	**Compliance Costs**	**Expected Damage Costs**	**Administration Costs**	**Reduction in Infested Houses**	**Peace of Mind**	**Damage to Iconic Structures**	**Unknown Catastrophe**	**On and Off-Site Impact**
Round 1	1	More	Most	Less	More	Less	Least	More	Fair
	2	More	Most	Fair	More	Most	Fair	More	Most
	3	More	Most	Least	Most	More	Least	Most	Least
	4	Fair	More	Less	More	More	Less	More	More
	5	Least	Most	Least	Most	More	Less	More	Least
	6	Less	More	Less	More	More	Fair	More	Fair
	7	Fair	More	Less	More	Fair	Fair	Most	Less
	8	More	More	Less	Fair	Most	Less	Most	Less
	9	Fair	More	Less	Less	More	Less	Most	Less
Round 2	1	Fair	Most	Least	Most	Fair	Least	Fair	Less
	2	More	Most	Fair	Fair	Most	Less	Fair	Fair
	3	More	Most	Least	Most	More	Least	Most	Least
	4	Fair	More	Fair	More	Most	Least	More	More
	5	Less	Fair	More	More	More	Least	Fair	Fair
	6	Fair	More	Less	More	More	Less	More	More
	7	Fair	Most	Less	More	More	Less	Most	Less
	8	More	More	Fair	More	More	Least	More	More
	9	Fair	More	Less	Fair	More	Less	Most	Less
Round 3	1	Fair	Most	Least	Less	Fair	Least	Less	Less
	2	More	Most	Fair	Fair	More	Least	Least	Fair
	3	Fair	Most	Less	Most	Fair	Least	Fair	Least
	4	More	Fair	Fair	Fair	More	Less	Fair	Fair
	5	More	More	More	More	More	Least	More	More
	6	Fair	More	Less	More	More	Less	More	More
	7	Fair	More	Less	More	More	Least	Fair	Least
	8	More	More	Fair	Fair	More	Least	Less	Less
	9	Less	More	Least	Fair	More	Least	Most	Less

Table 9. Total performance index

		Round 1			Round 2			Round 3		
		Do Nothing	State-Wide Building Restrictions	Delimited Building Restrictions	Do Nothing	State-Wide Building Restrictions	Delimited Building Restrictions	Do Nothing	State-Wide Building Restrictions	Delimited Building Restrictions
Economic Sub-Criteria	**Compliance Costs**	1.0	0.0	0.5	1.0	0.0	0.5	1.0	0.0	0.5
	Expected Damage Costs	0.0	1.0	1.0	0.0	1.0	1.0	0.0	1.0	1.0
	Administration Costs	1.0	0.0	1.0	1.0	0.0	0.0	1.0	0.0	0.0
Social Sub-Criteria	**Reduction in Infested Houses**	0.0	1.0	0.8	0.0	1.0	1.0	0.0	1.0	1.0
	Peace of Mind	0.0	1.0	1.0	0.0	1.0	0.8	0.0	1.0	0.8
	Damage to Iconic Structures	0.0	1.0	0.2	1.0	1.0	1.0	1.0	1.0	1.0
	Unknown Catastrophe	1.0	1.0	1.0	0.0	1.0	1.0	0.0	1.0	1.0
Environmental Sub-Criteria	**On and Off-Site Impact**	1.0	0.0	0.4	1.0	0.0	0.7	1.0	0.0	0.7
Total Performance Index		30%	30%	40%	28%	33%	39%	28%	33%	39%
Rank		**2**	**2**	**1**	**3**	**2**	**1**	**3**	**2**	**1**

Figure 4 shows the extent of weightings changes by round across the sub-criteria. These are expressed in percentage form, and individual criteria are grouped together along the horizontal axis. The sum of all the lightly shaded bars labelled 1 (i.e. round 1) equals 100%, as does the summation of all the bars labelled 2 and 3.

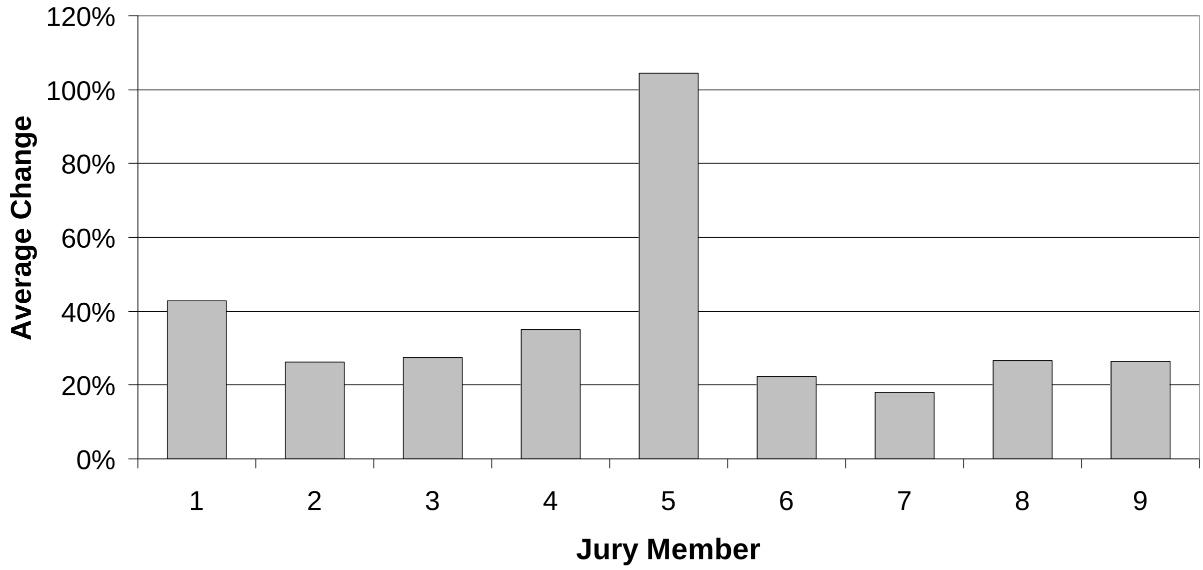
Figure 4. Change in sub-criteria weights by round



The individual members of the jury demonstrated a propensity to change their weighting opinions in response to information revealed and discussed during the deliberative sessions of the workshop. This is particularly the case with jury member 5, and to lesser extent members 1 and 4. But across the board weights were changed significantly over the course of the three rounds. Figure 5 illustrates the extent of weight changes for sub-criteria by jury member between rounds 1 and 3. Each bar indicates the average difference between an individual juror's set of weights in rounds 1 and 3 in absolute value terms.

At the completion of the workshop comments from the jury members indicated a high level of interest in the DMCE technique employed in the workshop, but not necessarily in the scoring procedures (i.e. Tables 2 and 3) or the aggregation functions. This detracted from their confidence in the final evaluation. However, the deliberative process itself was viewed favourably.

Figure 5. Change in sub-criteria weights by juror between rounds 1 and 3



5. Issues

5.1 Treatment of uncertainty in the case study

The group interaction of DMCE makes it more challenging to incorporate uncertainties into decision-making (Gregory 2006). The keys to success are: (a) to explicitly take uncertainty into consideration in the DMCE process, and (b) to eliminate linguistic uncertainty and other sources of arbitrary disagreement. In addition to the standard sensitivity analysis, there are two major ways of treating uncertainty explicitly in the MCDA literature. The first is to apply the fuzzy logic model (as above), and the second involves the inclusion of uncertainty-related criteria in forming the impact matrix (Cook and Proctor 2007; White, Fane et al, 2008; Kiker, Bridges et al, 2008).

When initially designing our DMCE we only adopted the fuzzy model. It was felt that including uncertainty-related criteria separately could be viewed as double counting or redundant. However, during the workshop the criterion *unknown catastrophe* was added by the jury to address the possibility of traumatic yet unseen impacts of EHB on future generations that are not addressed by the existing scientific research. The fuzzy system approach was adopted in an attempt to deal with the uncertainties associated with estimating values in the impact matrix, including model uncertainty, natural variation, measurement error and systematic error in an "all-in-one" manner. This is a coarse approach but it is difficult to separate these components when using other researchers' results in the impact matrix. The same fuzzy system approach was applied to handle the uncertainty due to the subjective judgement of workshop participants in ranking the criteria.

A five-tiered linguistic system (Tables 2 and 3) was used in an effort to reduce stakeholders' cognitive burden in the evaluation process. Research showed that ordinal ranking is a more favourable weighting technique than fixed point, rating, graphical presentation and paired comparison methods (Hajkowicz, McDonald et al 2008). These authors also concluded that it is undesirable to rely upon any single weighting technique as there may be bias associated with that particular technique, therefore, we also asked workshop participants to rank the different criteria's importance relative to their most favoured approach (data to be analysed).

While simple, it is prudent to point out several shortcomings of the five-tiered scoring system we have used (i.e. in Tables 2 and 3). In our attempts to make the process easy for participants to follow, we have introduced several problems. Firstly, scoring alternatives on a generic 1-9 scale (as we have done) risks information that may be critical to a decision being discarded. It is often the case that jury members have a perfectly good understanding of natural units (such as $) that help them to capture this information in their scoring and weighting activities. Moreover, replacing natural units with an assumed scale between the best and worst scores can introduce a scoring bias since it is not obvious what scale should be used. For instance, if a linear scale is used when a log scale would have been appropriate, the relative preference between different scores will be severely understated.

Past research has showed that the choice of different MCDA methods does not lead to significantly different results unless there are mixed ordinal and cardinal data in the impact matrix. More emphasis should be put on the initial structuring of the decision problems (Hajkowicz 2008). We believe this is also true in terms of handling uncertainty. We summarise below our efforts in treating linguistic uncertainty in structuring and presenting the problem.

1) Define criteria as specifically as possible

   It has been argued that people tend to rely on a limited number of "heuristic principles" to help them simplify the process of judgment (Kahneman 1982). Decomposing the problem into more specific units can force people to consider all the aspects associated with a complex decision. In the EHB case this was accomplished by separating the three broad categories of criteria (economic, social and environmental) into eight specific sub-criteria.

To ensure that all the available information was considered during the scoring process, a summary document of the threats and other pertinent information was compiled and provided to workshop participants as a reference guide before the workshop and during the scoring process. On the workshop day, participants reviewed each criterion through open-discussion prior to scoring so that any information that was left out of the document could be considered.

2) Point out explicitly the uncertainty associated with values in the impact matrix

This was accomplished by presenting quantitative data as a minimum-maximum range as well as mean values. We also pointed out the uncertainty of calculation in our presentation on each specific criterion on the workshop day. For instance, in estimating the damage cost of EHB due to collapsed houses, the assumption is that EHB could survive the high roof temperature in WA summers and that may not be the case in reality.

However research has also showed that the benefits of representing systematic uncertainty in the data may be outweighed by the costs of decreased comprehension (Gregory 2006). Although it is beyond the scope of this study to test how much exposure to uncertainty is optimal or the best way of presenting uncertainty in a DMCE process, the project team is interested in exploring the answer to that question in our future research.

3) Emphasize the importance of common understanding of specific terms throughout the workshop.

Part of the eye-opening experience of running a DMCE workshop is how different people's ideas regarding the same term can be. During the workshop day for instance, the group realized that "Doing-nothing" to "manage" EHB could mean either "leave it completely alone" or "eradication only, without forcing the industry to do any timber treatment". Differences in understanding towards this management option lead to differences in the weights put on the sub-criteria "Administrative cost".

Attention was also paid to giving participants ownership of the concepts by being flexible about these definitions. A group decision was made about changing the meaning of "Do nothing" from "eradication only" to "leave it alone" and the corresponding values in the impact matrix were adjusted to reflect the change. In hindsight, this flexibility could have been enhanced by making more time available to add and subtract criteria of the jury formed to address perceived gaps in the other criteria. Unfortunately, due to the limited time and budget available in this particular experience this was not possible.

5.2 The deliberative process

Deliberative valuation has become an increasingly popular area of research in recent years (Spash 2008; Zografos and Horwarth 2008). Only recently has it been applied to risk management in biosecurity (Cook and Proctor 2007) and to our knowledge our current project is the second case study in the field. We believe the DMCE method offers at least two advantages: 1) It bridges the gap between risk assessment and risk management by encouraging a partnership between scientists and stakeholders, and 2) it offers an opportunity to explore social dimensions of biosecurity risk before a collective decision is made.

On the workshop day the deliberative process in the afternoon allowed all outliers (those participants who gave the highest or lowest weights to each criterion) to state their rationale. This open discussion benefited the process in at least two ways. First, it helped to eliminate linguistic uncertainty for the criterion *unknown catastrophe*, which was renamed *long-term impacts* and *intergenerational impacts* during previous rounds of weighting. Only at this point did the group realize that there were at least three definitions for this single criterion. Second, it encouraged social learning and information flows between scientists and stakeholders. For example, during the deliberative process people learned that current chemicals used for timber treatments are much more environmentally friendly than in the past. As a result a drop in the weight for this criterion was observed after the deliberation process.

As observers for this deliberative process it was obvious that risk was socially constructed (Slovic 1987) and this was especially the case when scientific facts are not readily available. The addition of the criterion *unknown catastrophe* and the evolution of its meaning as the day went by was a good example. How should we make a collective decision in the face of uncertainty? Deliberative democracy is one answer to the question (Dryzek 2000) and we believe the DMCE approach provides a platform for diverse voices and preferences to be incorporated and interacted.

By no means do we promote the DMCE technique or the deliberative process in general as a panacea. There are at least a couple of problems in applying the DMCE to make biosecurity decisions. First, deliberative processes pose new challenges to incorporate uncertainty successfully (Gregory 2006). For instance, we do not know why the group's preferences change, whether it is due to a new way of constructing and presenting uncertainty or due to the group dynamic, the time allocated to the process, quality of the group's level of education, or other factors. When designing our DMCE one idea was to ask people to do a fourth round of weighting after all uncertainty components were explicating exposed in one presentation. But we gave up this idea partially because of the compounding effect of this "uncertainty treatment" and group dynamics. This is certainly one issue for future research.

Second, people are generally not familiar with the deliberative process. As a result they may encounter difficulty in participating. At the end of the day one comment from our participants, for example, was that we seemed to force everybody to assign the same weights because only those outliers were asked to speak. At that point we had to state again that the purpose of DMCE is not uniform opinion. Overall the feedback from workshop participants was positive. They felt that the process served to raise awareness, generate new ways of thinking, produce a solidly informed group, and give the group a common language.

5.3 Roles: Participants versus Experts

Our DMCE workshop required participants to make technical judgments on the performance of EHB management alternatives on objectives. We acknowledge that ideally matters of technical fact should be left to the appropriate experts. For instance, if we need to know the probability of EHB spreading from the Perth metropolitan area in the next ten years under a specified set of circumstances, we would use the judgement of experts from DAFWA and elsewhere to arrive at probability distributions (e.g. Clemen and Winkler, 1999). The DMCE analyst would then communicate this information to participants in a form that they could easily understand. Participants should then decide how important that information is in the context of a given decision.

This ideal should indeed be something the DMCE analyst attempts to produce. However, there are some practical problems that must be overcome. Firstly, they must ask themselves "who is a relevant expert?" This is not a trivial exercise when one considers there may be a range of experiences that may be relevant. Is a PhD in the biology of EHB equivalent to a South African public servant who has witnessed EHB infestations in the past? Secondly, there is the potential for experts to testify strategically in order to either promote interest/potential investment in their area, or to discourage it from others. Finally, there may be a general unwillingness on the part of relevant experts to engage in the process of risk communication. We are not in the position of answering these problems and how they relate to DMCE exercises such as the one conducted as part of this project.

5. Recommendations

1. **Strategies for the effective communication of risk and uncertainty in invasive alien species management decisions should form a part of future research proposals**

   Although it has been identified as a component of risk analysis, risk communication is often neglected in biosecurity research. This partly explains the lack of uptake of economic information in past biosecurity risk management decisions.

2. **Traditional economic analysis should retain a significant role in resource allocation decisions.**

   There are a variety of valuation techniques that can be used to elicit social and environmental values related to IAS impacts, and used in a benefit cost or cost effectiveness analysis framework. Multi-Criteria Decision Analyses should not be seen as replacement for traditional economic analyses, but as complements.

3. **Deliberative multi-criteria evaluation should be considered a relevant framework for making invasive alien species management decisions.**

   We have shown the DMCE is a flexible decision facilitation technique that provides a useful structural framework for making decisions. It provides a context for complex information to be communicated to diverse decision making groups. Sensitivity analysis and trade-offs are transparent under this framework, adding accountability to the decision making group.

4. **Efforts to simplify a deliberative multi-criteria evaluation should not detract from the accuracy and specificity of available information.**

   There is a temptation for analysts to simplify the DMCE process to accelerate the decision making process. However, as the case study in this report has pointed out, this can have the opposite effect by causing confusion amongst decision makers about aspects of the process. The purpose of the DMCE, definitions of the alternatives being considered, criteria, scoring and weighting systems should be clear and concise. Where possible, criteria should be scored against indicators in natural units that decision-makers are used to considering. These include $, tonnes, years, etc.

5. **When using deliberative multi-criteria analysis sufficient time should be given to the decision-making group to allow them to come to a decision that they are comfortable with.**

   The DMCE should not be truncated, but allowed to run its natural course. This may involve an open ended time frame, particularly when the decision involves a ‘data rich’ area. In such cases the dissemination and consideration of large amounts of information may take time. However, it must be acknowledged that longer time frames involve higher costs. So, the cost effectiveness DMCE should be compared to alternatives, and to the impact of the decision on social welfare.

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8. Appendix – Methods used in Multi-Criteria Analysis

There are currently a wide variety of MCDA methods available. Although diverse in nature, the methods have a common goal of evaluating and selecting among alternative options based on multiple criteria through the use of systematic analyses that overcome limitations of unstructured individual and group decision-making (Linkov, Varghese et al. 2004). The methods differ in use as each requires different types of raw data and follow different aggregation algorithms. Techniques can rank options, identify a single best alternative, sort alternatives into groups, provide an incomplete ranking, or differentiate between acceptable and unacceptable alternatives (Roy 1985, Linkov, Varghese et al. 2004). The former three approaches (choice, ranking, sorting) lead to an evaluation outcome (Zopounidis and Doumpos 2002). Choice and ranking methods are based on relative judgments thus are products of the set of alternatives considered in the study, while alternatively, the use of a sorting technique requires absolute judgments (Zopounidis and Doumpos 2002).

There are no set rules for selecting a method from the plethora of those available. Zak, (2006) suggest that Electre and the Analytical Hierarchy Process (AHP) are the most reliable and user-friendly MCDA methods. This claim is based on the appreciation of both the process and final rankings. The Utility Theory Additive (UTA) method is recommended for decision issues with a larger number of criteria, while Electre, Oreste and Mappac methods should be applied to smaller criteria numbers, with the AHP method applicable to both scenarios (Zak 2006). Moffett and Sarkar (2006) suggest that Regime and Non Dominated Set (NDS) be used if the process requires only that alternatives be qualitatively ordered. When alternatives and criteria can be quantitatively evaluated, and the criteria are independent of each other, then multi-attribute value theory (MAVT) should be used (with preferences obtained by a modified Analytic Hierarchy Process (mAHP) (Moffett and Sarkar 2006). Overall, the choice may be based on subjective judgements such as the preference for a method with certain algorithmic characteristics or the choice may be purely pragmatic with decision maker ease being the primary driver (Linkov, Varghese et al. 2004). For clarity in this review, methods are categorised into Elementary methods, Single Synthesizing Criterion, Outranking, and Mixed Methods based on the typography of Guitouni and Martel, (1998). A description of some of the common approaches in each group will be provided.

8.1 Elementary Methods

Elementary methods are easy to apply and understand and often no computational software support is required. Elementary methods work to breakdown complex problems into a singular metric resulting in a simplified and often overly conservative representation of the problem (Linkov, Varghese et al. 2004). Methods such as pros and cons analysis, maximin and maximax methods, conjunctive, disjunctive methods, and lexicographic methods tend to be best suited for problems with a single decision-maker and few alternatives. Environmental decision-making settings tend not to show these characteristics (Linkov, Varghese et al. 2004). Nonetheless each problem has unique characteristics and requires consideration of an array of methods to best suite its needs. The following Elementary methods are briefly reviewed in no particular order.

8.1.1 NDS (Non Dominated Set)

NDS is a flexible method due to its minimal assumptions and resulting objective nature (Moffett and Sarkar 2006). NDS is unique in that it is compatible with the results produced by any other rational decision making procedure. The NDS requires only that alternatives be qualitatively ordered by each criterion (Moffett and Sarkar 2006). Additional methods can be selected to refine the NDS depending on the compatibility of criteria.

8.1.2 Maximin and Maximax methods

Maximin ranks alternatives by comparing each on the basis of its worst ranking under the criteria (Yoon and Hwang 1995). The alternative for which the score of its weakest criterion is the highest is preferred. In other words, the Maximin aims to avoid the worst possible performance by maximizing the minimal performing criterion. Maximax, alternatively, produces a ranking of the alternatives by comparing each on the basis of its best ranking under the criteria (Yoon and Hwang 1995). In order to use Maximin and Maximax, all criteria must be comparable through the measurement on a common scale (Linkov, Varghese et al. 2004).

8.1.3 Conjunctive and disjunctive methods

Conjunctive and Disjunctive methods are characterised by requiring satisfactory rather than “best performance” for each criterion. The Conjunctive method requires an alternative to meet a minimal performance threshold for all criteria, whereas in the disjunctive method the alternative has to exceed a given threshold for at least one criterion (Linkov, Varghese et al. 2004). An alternative is deleted from the further consideration if it does not meet the rule (Linkov, Varghese et al. 2004).

8.1.4 Lexicographic method

The Lexicographic method provides a means of ranking criteria in the order of importance. The ranking is set out so that the alternative with the best score on the most important criterion is chosen (Linkov, Varghese et al. 2004).

8.2 Single Synthesizing Criterion

The single synthesizing criterion methods are generally grouped together due to the aim of establishing an aggregation function that best represents decision maker preferences (Guitioni and Martel 1998).

8.2.1 MAUT methods

The Multi-Attribute Utility Theory (MAUT) methods produce criteria weights that reflect the relative importance when scores are from a common dimensionless scale. MAUT models rely on weighting criteria and creating utility functions across the levels of each criteria (Bosworth, Gingiss et al. 1999). Utility functions are created which allow for the transformation of different criteria, be they qualitative or quantitative into the common dimensionless scale. MAUT is one of the most scientifically grounded MCDA methods with a strong foundation in decision theory (Linkov, Varghese et al. 2004). MAUT models are usually well suited for evaluation type problems (Gustafson, Sainfort et al. 1993). As MAUT requires making difficult tradeoffs, the procedure can be a time consuming and often frustrating process for decision makers which can limit its application (Linkov, Varghese et al. 2004). Also, MAUT models are sensitive to missing data, requiring re-weighting of the remaining criteria to arrive at a score (Gustafson, Sainfort et al. 1993).

8.2.2 Simple Multiattribute Rating Technique (SMART)

SMART is the simplest form of the MAUT methods where criteria are ranked in order of importance (Edwards 1977). 10 points are assigned to the least important criterion with the next-least-important criterion having more points assigned to it, and so on, to reflect their relative importance (Edwards 1977). The final weights are obtained by normalizing the sum of the points to one.

8.2.3 The Analytic Hierarchy Process

Criteria are weighted using a decision making tool developed by Saaty (1977) known as the Analytical Hierarchy Process (AHP) which is comprised of pair-wise comparisons. The “pair-wise comparison” refers to the relative importance of one criterion in comparison to another, providing a weighting from 0 to 1 for each. The importance of each criterion relative to another is evaluated on a nine-point scale, ranging from 9 (extremely more important) to 1/9 (extremely less important), with 1 being equally important (Saaty 1977). This scale is shown in Table A1. The numbers in the scale represent the relative importance of each category and criteria. Although any ratio scale can be used in this method; the choice of the 1 to 9 scale recommended by Saaty (1980) is recommended for use in the AHP due to the experimental evidence of having successfully captured user preferences (Harker and Vargas 1987).

Table A1. Rating scale of Saaty (1977, 1986).

1/9	1/7	1/5	1/3	1	3	5	7	9
Extremely	Very Strongly	Strongly	Moderately	Equal	Moderately	Strongly	Very Strongly	Extremely

⬅ LESS IMPORTANT MORE IMPORTANT ➡

An example of how and AHP comparison is carried out for criteria is provided in Table A2 below. The importance of *Weather* criteria in the row relative to slope criteria in the column = 5 (the row variable is strongly more important than the column variable).

Table A2. Pair wise comparison matrix method for assessing the relative importance of the criterion influencing wildlife vehicle collision risk along a highway.

	Landcover	**Slope**	**Weather**
Landcover	1		
Slope	1/3	1	
Weather	3	5	1

The AHP process requires a review of the weights for consistency. Inconsistencies arise when the relative importance of one criterion does not correspond logically to the importance of another. Weightings are assessed for consistency upon the completion of expert or stakeholder weighting and experts are then notified to adjust the weights. The most inconsistent comparison is identified based on a deviation number calculated in Idrisi (Eastman 2006) to provide the expert with guidance on where to begin the re-weighting. Saaty (1977) indicated that matrices with consistency ratio ratings greater than 0.1 should be re-evaluated.

The AHP has been criticised on a theoretical basis due to the potential for rank reversals which violate the independence of irrelevant alternatives axiom of decision theory (Arrow & Raynaud, 1986). A rank reversal is when the ranking of any two alternatives is reversed when a new alternative is introduced (Dyer, 1990).

8.2.4 The modified Analytic Hierarchy Process (mAHP)

The mAHP uses the same pair-wise comparison weighting process as the AHP, but it also constructs a linear value function for each criterion (Moffett and Sarkar 2006). These single criterion value functions are aggregated on the basis of the weights to assign a quantitative value to each alternative. 2004). The mAHP avoids the rank reversal problem (Moffett and Sarkar 2006).

8.2.5 UTA (utility additive), MACBETH, and TOPSIS

The UTA method belongs to the MAUT family which can be applied to solve multi-objective ranking and choice problems (Zak 2006). In UTA, criteria are evaluated by an additive utility function which searches for a function shape that best reflects decision maker preferences. Like UTA, MACBETH and TOPSIS assume independence and aggregativity conditions (Moffett and Sarkar 2006). The three methods offer alternatives in carrying out aggregation which is where they differ from the MAVT and AHP (Moffett and Sarkar 2006). Moffett and Sarkar (2006) state that there is no reason to use any of the three methods over MAVT or mAHP.

8.3 Outranking Methods

Outranking methods provides a means to assess preference of the most favoured alternative. The alternative that performs the best on the greatest number of criteria is ranked highest (Guitouni and Martel 1998). Outranking avoids compensatory optimization functions where criteria are reduced to the single best. Similar to most MCDA methods, outranking is partially compensatory (Guitouni and Martel 1998). Outranking methods are also flexible as they allow semi-quantitative scales (Linkov, Varghese et al. 2004). Outranking methods still require alternatives and criteria to be specified, thus broadly similar in nature to MAUT methods. Outranking methods are appropriate to issues with criteria that are difficult to aggregate, widely ranging measurement scales, and incomparable units (Guitouni and Martel 1998).

8.3.1 ELECTRE methods

Electre is a ranking method founded on the outranking relation (Zak 2006). Decision maker preference is divided into a scale that includes indifference, weak preference, strong preference and incomparability (Roy 1990). The Electre I provides partial rankings and is the simplest of the Electre family. Electre II provides a full ranking of the alternatives while Electre III creates an outranking degree that measures the performance of alternatives against each other (Zak 2006).

8.3.2 PROMETHEE methods

Promethee methods were introduced by Brans and Vincke, (1985). The Promethee output is given by a score with higher values indicating a better performance. The Promethee I method provides partial rankings where some couples of alternatives are comparable, some others are not. The Promethee II is a step more advanced by producing a complete partial ranking. Promethee V uses an Optimization concept under constraints that seeks to maximize the outranking value of the selected alternatives (Brans and Vincke 1985).

8.3.3 Oreste and Mappac methods

The Oreste and Mappac methods suit issues that require the ranking of alternatives with multi-objectives. While both methods are based on the outranking relation, the Mappac method also combines MAUT theory (Zak 2006). Where the Oreste method constructs a complete preorder of variants, the Mappac method uses pairwaise comparison for each pair of criteria (Pastijin and Leysen 1989, Zak 2006).

8.4 Mixed Methods

8.4.1 Qualiflex

Qualiflex creates a rank value for each alternative based on ordinal values from each criterion (Moffett and Sarkar 2006). Qualiflex requires aggregating the rankings of alternatives by a method that has been criticised as being ad hoc and for assuming that each pairwise comparison of the ranking of alternatives has the same value (Moffett and Sarkar 2006).

8.4.2 Bayesian Models

Bayesian models use Bayes' theorem to revise an opinion on the probability of an outcome when new evidence arises (von Winterfeldt and Edwards 1986). Subjective Bayesian models multiply all available data together. Bayes' theorem calculation omits missing data resulting in models that can adapt to incomplete data sets (Bosworth, Gingiss et al. 1999). Bayesian models are typically best suited for diagnostic and prediction-type problems as opposed to evaluation-type issues (Gustafson, Sainfort et al. 1993).