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Title
Potential methods and tools for estimating biosecurity consequences for primary production, amenity, and the environment.
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Summary
The aims of this report were to o catalogue potential methods and tools for estimating impacts on social amenity and the environment, o identify tools that can be used to estimate impacts and their importance o characterise the data needs/availability, resource needs of specific tools that may be used operationally to inform these biosecurity decisions, and o develop tools/methods that are suitable to operational biosecurity decision making needs. The report delivers an outline of available methods for characterising impacts under the three primary categories of economics, social amenity and the environment. The potential for these methods to serve effectively in a biosecurity setting is evaluated. The most appropriate tools and methods for characterising the impacts of invasive pests and diseases are identified from the sets available. Their strengths and weaknesses are discussed, and examples are provided of their applications. The report develops specific approaches for assessing primary production, social and environmental impacts and describes them in detail. The methods include; o A partial equilibrium economic model coupled to a simple biological spread model applicable to a broad range of pests and diseases; o The use of constructed scales to encapsulate a range of social, cultural and amenity values; o The use of the probability of extinction as an appropriate measure of impact on biodiversity. Given the difficulty of estimating this aspect directly, a method is outlined for estimating an impact metric that may be applied to a broad range of native and culturally important species, and which provides a proxy measure of the probability of extinction.

The report also discusses situations where a broader general equilibrium analytical framework is appropriate and outlines what this approach would involve. It explores how the shapes of consequence curves can affect investment decisions, particularly in the longer term.

Due to the difficulty in servicing traditional economic impact simulation models with sufficient and accurate data, we suggest the need for a broader range of criteria on which to base biosecurity investment decisions. In particular, we advocate the use of quantitative models of prediction in conjunction with structured decision-making techniques so that the full extent of market and non-market invasion consequences can be appreciated by decision makers.

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Potential methods and tools for estimating biosecurity consequences for primary production, amenity, and the environment.

ACERA 1002

Improved biosecurity decision-making through better characterization of consequences.

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Executive Summary

The aims of this report were to

- catalogue potential methods and tools for estimating impacts on social amenity and the environment,
- identify tools that can be used to estimate impacts and their importance
- characterise the data needs/availability, resource needs of specific tools that may be used operationally to inform these biosecurity decisions, and
- develop tools/methods that are suitable to operational biosecurity decision making needs.

The report delivers an outline of available methods for characterising impacts under the three primary categories of economics, social amenity and the environment. The potential for these methods to serve effectively in a biosecurity setting is evaluated.

The most appropriate tools and methods for characterising the impacts of invasive pests and diseases are identified from the sets available. Their strengths and weaknesses are discussed, and examples are provided of their applications.

The report develops specific approaches for assessing primary production, social and environmental impacts and describes them in detail. The methods include;

- A partial equilibrium economic model coupled to a simple biological spread model applicable to a broad range of pests and diseases;
- The use of constructed scales to encapsulate a range of social, cultural and amenity values;
- The use of the probability of extinction as an appropriate measure of impact on biodiversity. Given the difficulty of estimating this aspect directly, a method is outlined for estimating an impact metric that may be applied to a broad range of native and culturally important species, and which provides a proxy measure of the probability of extinction.

The report also discusses situations where a broader general equilibrium analytical framework is appropriate and outlines what this approach would involve. It explores how the shapes of consequence curves can affect investment decisions, particularly in the longer term.

Due to the difficulty in servicing traditional economic impact simulation models with sufficient and accurate data, we suggest the need for a broader range of criteria on which to base biosecurity investment decisions. In particular, we advocate the use of quantitative models of prediction in conjunction with structured decision-making techniques so that the full extent of market and non-market invasion consequences can be appreciated by decision makers.

1.0 INTRODUCTION

The original project agreement for ACERA 1002 outlined the following deliverables.

1) A catalogue of potential methods and tools for estimating impacts on social amenity and the environment, and a detailed analysis of tools relevant to BSG operational needs and constraints (considering both pre- and post border environments)

- Define what tools can be used to estimate impacts and their importance.
- Characterise the attributes (e.g. data needs/availability, resource needs) of specific tools that can potentially be used operationally to inform these biosecurity decisions.
- Develop a catalogue of tools/methods that are suitable to operational biosecurity decision making needs.

2) A catalogue of potential methods and tools for estimating impacts on primary production addressing the following themes:

- Define what tools can be used to estimate impacts for primary production.
- Characterise the attributes (e.g. data needs/availability, resource needs) of specific tools that can potentially be used operationally to inform biosecurity decisions, including strengths, weaknesses and limitations to the use of data, information transfer, and expert judgment in estimates of economic impact.
- Recommend tools or methods that are suitable to operational biosecurity decision-making, including
    - guidance on input-output analysis, especially on how far an economic analysis should go in capturing secondary flow-on impacts; and
    - how recommended tool(s) might be extended to include other market-based impacts, such as consequences of pest invasion on public and private infrastructure.

A previous report under ACERA 1002, *The characterisation of consequences for prioritising pests and actions*, detailed a structured decision-making (SDM) framework for biosecurity. It concentrated on presenting and illustrating a systematic way for combining estimates of impacts across economic, environmental and social concerns. Section 2 of this report delivers the agreed objectives of the project outlined above, namely, reviews of potential methods for estimating impacts, characterisation of their relevant attributes in a biosecurity context and recommendations for approaches for operational biosecurity decision-making, for each of primary production (Section 2.1), social amenity (Section 2.2) and the environment (Section 2.3). Each of these sections contains a subsection that outlines a catalogue of options, and a separate subsection that details the suggested approach.

Following discussion with DAFF staff, this report goes further than the original objectives and addresses some fundamental, unresolved issues in multicriteria consequence assessment, topics that are of special interest to biosecurity. Section 3 addresses the sensitivity of consequence estimates to the time frame over which impacts are estimated for a range of important decision values. Section 4 revisits the issue of how to combine different values when making a decision, in the light of the suggested approaches outlined in Section 2, contrasts the strengths and weaknesses of the approach, and reassesses the question of importance compared to traditional benefit-cost analysis. Section 5 provides a discussion of the strengths and weaknesses of the recommended approaches, in the context of biosecurity decision-making.

One important concern highlighted in the Beale report is the development of a common risk-management strategy that will link pre-border and post-border decision making frameworks. In the pre-border world, a key planning focus is on Import Risk Assessment (IRA), and relevant objectives would relate to the potential consequences associated with a commodity that results in pest entry, establishment, and spread within Australia. In a post-border world, the planning focus is on determining which actions to take to eradicate or control a pest, and determining whether the pest is considered 'nationally significant' in terms of its potential impact on people, the environment, or business activity (the latter being an important determinant of who pays for control actions). Despite some differences in the specific decision settings pre-border and post-border, there is a spine of common concerns. A single set of fundamental objectives across the biosecurity continuum would usefully include the minimisation of impacts on primary production, social amenity and the environment. The methods developed in this report are applicable across the biosecurity continuum.

2.0 ESTIMATING CONSEQUENCES

The task of estimating consequences is informed by clarity in objectives associated with biosecurity decision-making. Objectives should appeal to fundamental values that biosecurity agencies seek to protect. This report does not provide a detailed or exhaustive treatment of all methods available for the prediction of all potential impacts of biosecurity concern.

A set of fundamental objectives was developed at a workshop conducted as a part of ACERA 1002 in April 2010 (Table 1). Section 2.1 of this report deals with impacts associated with primary production, incorporating loss of revenue from production loss or market loss, and management costs associated with treatment of pests or disease. Other economic impacts that can be readily described in monetary terms are not treated here, including impacts on households and consumers, and costs to government in the management and prevention of pest incursions.

Section 2.2 deals with estimation of environmental impacts, focusing on species extinction risk as a surrogate for degradation of ecological communities or broader loss of biodiversity. We make no attempt to reproduce the substantial and burgeoning literature dealing with ecosystem services (see Seppelt et al. *in press* and Seppelt et al. (2011) for recent reviews).

In section 2.3 we explore estimation and characterisation of social impacts. Here we touch on how measures of income or unemployment can be used as proxies for community instability arising from economic stress. Greater detail is given to describing how constructed scales can be used to capture intangible impacts of pests and disease, including spiritual values and aesthetics. We do not provide any substantial commentary on estimation of recreation impacts. Methods for estimating and valuing loss of recreation opportunities is well established in the economics literature (see Nelson (2010) and Thiene and Scarpa (2009)).

Finally, we note that a separate report under ACERA 1002 dealing with human health impacts is in preparation.

The role of attributes

Attributes are used to assess the performance of alternative decision options. They include things such as the economic cost to a farmer, or the area of habitat of a threatened species displaced by an invasive organism. Also known as performance measures or evaluation criteria, attributes clarify the meaning of an objective and provide metrics for expressing and communicating the implications of different management or decision alternatives.

Table 1. Preliminary table of fundamental objectives for biosecurity management and their treatment in this report.

Objectives	Considerations	Treatment in this report
MANAGEMENT COSTS		
Government	Surveillance, eradication, control, research, etc.	Not included
Business	Surveillance, eradication, control, research, etc.	Section 2.1
Household / Consumers	Primarily site-level control costs	Not included
ECONOMY		
Business	Loss of revenue from production loss or market loss	Section 2.1
Household / Consumers	Loss of property value, cost of repair/replacement	Not included
ENVIRONMENT		
Ecological Communities	Keystone species, character species	Section 2.3
Biodiversity	Other species, structural / functional attributes	Section 2.3
Ecosystem Services	Water and air quality, soil salinity, resistance to fire/flood	Not included
HUMAN HEALTH		
Mortality	Attributable deaths	Not included
Morbidity	Attributable illness, from discomfort to hospital visits	Not included
SOCIETY AND CULTURE		
Community Stability	Includes employment/displacement effects	Section 2.2
Spiritual Values	Places of spiritual importance	Section 2.2
Aesthetics	Landscapes, views, waterways	Section 2.2
Recreation and Culture	Recreational, leisure, cultural activities	Not included

Desirable properties of attributes include that they are (Keeney and Gregory, 2005):

- Unambiguous: a clear relationship exists between an objective and the description of consequences under each alternative using the attribute.
- Comprehensive: the attribute levels cover the range of possible consequences for the corresponding objective under all alternatives, and value judgments implicit in the attribute are reasonable.
- Direct: the attribute levels directly describe the consequences of interest.
- Operational: in practice, information to describe consequences can be obtained and value trade-offs can reasonably be made.
- Understandable: consequences and value trade-offs made using the attribute can readily be understood and clearly communicated.

As the examples below will make clear, there are situations where compromises among these desirable qualities are needed. In all cases, however, care in the development of attributes is essential.

Types of attributes

The range of performance measures used in SDM typically incorporates input from three types of attributes: *natural measures*, *proxy measures*, and *constructed measures* (Keeney, 1992; Keeney and Gregory, 2005).

Natural measures are in general use and have a common interpretation: just as the objective to 'maximise profits' is naturally measured in dollars, the objective to 'reduce health effects' might count the number of sting-related hospital visits per year in an area affected by a pest. Natural measures should be used whenever possible because they are unambiguous, easily understood, readily estimated, and because they are direct and readily communicate what is at stake.

Proxy measures often are also in general use. For example, 'number of dead trees observed per hectare' is used as a proxy for the health of a forest community, and air emissions (measured in ppm) are used as a proxy for health-related impacts that are harder to measure (such as health impacts related to a proposed spray treatment). However, proxy measures are less informative than natural attributes because they only indirectly indicate the achievement of an objective.

Constructed metrics are used when no suitable natural or proxy measures exist. In such situations, analysts may develop a suitable, artificial scale. For example, to measure impacts on community relations resulting from widespread culling of a disease host, a constructed metric could be as simple as a -3 to +3 index, where -3 meant 'very poor relations' and +3 represented 'very good relations'. Different governance or implementation alternatives could be evaluated using this scale, with improvements in the index (e.g., a change from a score of -2 to a score of +2) demonstrating the benefits of a proposed new procedure or mitigation action.

Section 2 of this report describes measurable attributes that can be used to report expected consequences. For impacts on agricultural productivity the *natural unit* of monetary cost is used. For amenity impacts, examples of *constructed scales* are provided. And for environmental impacts, methods are developed for estimating species loss as a *proxy* for environmental impact.

Throughout this document we use the terms *pests and diseases* and *invasive species* interchangeably to describe species which when introduced to a new environment establish and spread to the point where they exert a net negative impact on society (Cook et al. 2010). In so doing we do not discount the fact that significant positive effects can be felt from non-indigenous organisms, perhaps the most notable example being the European honeybee (*Apis mellifera*) which more than offsets its damage to human health (see Harvey et al. (1984)) by delivering pollination benefits (i.e. Gordon and Davis (2003)). To be deserved of the title *pest* a species has to inflict more harm than it dispenses good. Furthermore, reference to pests and diseases in this document are made to both established species as well as exotic species (i.e. those that have not yet entered and become established in Australia). The analytical techniques we describe in section 2 are equally applicable to established and exotic species. We describe the consequences of invasion as *expected* consequences to capture impacts felt both today, and those we expect to be felt in the future.

2.1 PRIMARY PRODUCTION

This section summarises a range of options and outlines a preferred method of economic analysis for estimating the market impacts of an invasive species. It recommends a static, partial equilibrium microeconomic model together with a population diffusion model to examine the consequence a pest species is likely to have on the Australian economy. By assuming there is no government intervention upon introduction and establishment and simulating spread pests through domestic host plant industries, this technique estimates a ceiling for eradication costs in the event of an actual incursion. Moreover, endemic and exotic invasive species threats can be compared and prioritised on the basis of expected consequences.

2.1.1 *Options to estimate consequences*

Analyses of economic impact may be based on *general equilibrium* or *partial equilibrium*. A partial equilibrium approach analyses relationships within a particular market or a particular subsector, and assumes that this subsector is a completely self-contained entity within the economy. That is, changes in conditions within the subsector concerned will not flow on to the rest of the economy and affect, for instance, the general level of prices, employment or output. It follows that in the context of biosecurity pest or disease consequence assessment, a partial equilibrium approach is most appropriate where the effects of a pest are not expected to lead to general consequences throughout the rest of the economy. Instead, the effects are likely to be contained within a certain sub-sector that does not have significant linkages to other sectors of the economy.

However, it must be recognised at the outset that there are circumstances where a narrow partial equilibrium view does not sufficiently capture the true consequence of a pest on the economic system. *General* equilibrium analysis provides a much broader picture of how an adverse shock in one sector of the economy, such as a pest or disease incursion in an agricultural sector, will affect the entire economy. Moreover, it seeks to describe the process by which flow-on effects to the rest of the economy are felt, and how these in turn can feed back into the original sector affected. General equilibrium analysis therefore relies on describing the nature and strength of intersectoral linkages using techniques such as input-output analysis. This approach is appropriate where a sector with substantial linkages to the rest of the economy is likely to be affected.

The use of Computable General Equilibrium (CGM) models in economic analysis in Australia began with the development of the ORANI model (Dixon et al. 1982). Since then, a variety of CGE models have been developed, some of which capture State level activities. One such model is the Monash Multi-Regional Forecasting (MMRF) model (Adams et al. 2002), which in turn has been developed from the MONASH model introduced by Dixon and Rimmer (2002). The MMRF model is capable of both forecasting levels of economic activity in different economic sectors using inputs from organisations such as the Australian Bureau of Agricultural and Resource Economics (i.e. ABARE (2006)), as well as deviations from these forecasts that may be induced by shocks. These include policy changes, such as new taxes or subsidies, as well as exogenous (i.e. outside of the economic system) shocks like pest or disease incursions, changes in world commodity prices or the introduction of new technologies (Wittwer et al. 2005).

In the case of Australian agriculture, a significant pest of cereals or livestock may warrant the use of general equilibrium analysis. Indeed, this approach has been used in Wittwer (2005) to examine the potential consequences of Karnal bunt (*Tilletia indica*) on the Australian wheat industry, and in turn for the rest of the economy. However, for smaller intensive agricultural industries such as horticulture or floriculture, a general equilibrium framework may not be preferable since the consequences of a pest are expected to be contained within an industry. Hence, the broader economy may not experience measurable change as a result of its impact on this industry. Exceptions may occur in cases of highly polyphagous pests such as fruit flies or nematodes that have the capacity to severely damage large numbers of industries.

Since crops such as cereals and pulses are often used as inputs to the production processes of many other industries, changing production environments within these industries can have severe indirect as well as direct consequences. Because of the interrelationships between industries, the efficient allocation of resources in one is dependent on the input requirements of all other industries in the economy. Within CGE models, any *correct* (i.e. shortage-free as well as surplus-free) set of output levels for all industries in an economy must be consistent with all the input requirements in the economy so that no bottlenecks arise (Chiang 1984). If the costs associated with induced shortages are taken into account, the consequences of agricultural pests and diseases can be far greater than indicated by the primary production losses captured in a partial equilibrium analysis. Hence, a CGE model has more explanatory power when indirect consequences of pest and disease incursions are large.

To illustrate, take a typical large agricultural industry. Beyond the farm sector, industries reliant on the produced commodity in their production processes might include storage, transport and handling industries, wholesale marketing agents, primary processing industries, packaging and distribution firms, secondary processing industries, retailers and exporters. If a pest is introduced to the initial agricultural industry and causes a reduction in farm output, all subsequent (or 'down stream') industries dependent on it as a source of inputs will also be affected. It follows that if measures can be employed to reduce the likelihood of the pest entering in the first place, indirect, or *flow-on* benefits accrue to each sector beyond the farm gate[1].

Input-output (I-O) tables record all transactions of individual enterprises with other economic agents within a given period of time – inputs being what are purchased

[1] Of course, we should take account of the elasticity of substitution of the good concerned (i.e. measuring the relative ease with which it can be replaced by an alternative input) and entrepreneurial behaviour on the part of secondary sectors who may be able adjust their production processes accordingly, particularly in the long-run when no capital is fixed.

and outputs what is sold. An important application of I-O tables lies in the calculation of *I-O multipliers*, which provide a statistical indication of the extent of flow-on effects resulting from exogenous supply shocks such as pest outbreaks. An input-output multiplier is simply a ratio of the flow-on effects to the initial effect on a particular industry, and therefore provides a measure of the dependence of other enterprises within the economic community on the affected industry (Islam and Johnson 1997). For example, assume that we predict that a certain pest could cause damages to agricultural production of $10,000,000 per year if introduced. If the output multiplier produced from I-O tables corresponding to the industry affected by the pest is 1.2, it implies that the benefits of preventing its entry and establishment are actually more than twice this amount. For every dollar of output saved through its exclusion, a further $1.20 in flow-on benefits accrues to downstream industries. If these beneficiaries are taken into consideration, the benefits from keeping the pest out of Australia could be $22,000,000 per year. So, the omission of information concerning flow-on effects can lead to an underestimation of the possible consequences of a pest or disease.

It should be noted that in addition to national input-output multipliers produced by models like the MONASH model, estimates of input-output multipliers specific to states such as Western Australia (WA) are readily available and may be incorporated into economic assessments of (potential) pest damage specific to that State. Clements and Ye (1995) developed an input-output table for WA using a template of the national input-output table presented in ABS (1996). But, due to concerns that this model was not truly representative of WA agriculture it was re-visited by Islam and Johnson (1997)[2]. The resultant input-output table was comprised of 111 sectors, 10 of which were agricultural sectors (Islam and Johnson 1997).

2.1.2 A suggested approach

When the consequences of a pest or disease are (or are expected to be) restricted to a relatively small sector of the economy, as in the case of intensive agricultural pests, a partial equilibrium modelling approach is appropriate to analyse the possible consequences of invasion over time. This is certainly the case if multiple pest threats are to be compared and contrasted with one another, requiring simultaneous analyses. The process we describe here was developed in Waage et al. (2005) to rapidly assess the potential economic consequences of pest and disease threats to a country's economy using relatively few parameters. It combines a simple model of pest spread with a partial equilibrium model to estimate the potential damage host markets can expect to suffer if a pest becomes established and spreads.

Models have been developed for a few organisms to simulate spread over time, capturing their specific idiosyncrasies (e.g. Cowled and Garner 2008). Unfortunately, such detailed models are unavailable for most invasive organisms or their hosts. In

[2] It was felt that WA agriculture differed from a national average due to agro-climatic conditions (e.g. soil types, topography, rainfall patterns, temperatures and hydrology), implying that the technological structure and sales patterns on which agricultural industries were formed would also differ (Islam and Johnson 1997).

this sub-section we use a simple spread model that provides a reasonable approximation of the invasion process for many taxa (e.g., Reeves and Usher 1989; see Neubert and Parker 2004). This approach requires a few general characteristics of a biological model representing the kind of information that will be available realistically for most invasive organisms, which will then be linked to an economic model.

When dealing with exotic species, our simple model first requires probabilities of arrival and establishment. These parameters might be available as outputs of risk analyses using techniques such as social network analysis (Drake and Lodge 2004), artificial neural networks (Worner and Gevrey 2006; Paini et al. 2010) or expert elicitation (Tyshenko and Darshan 2009). In terms of our discussion, assume that entry and establishment probabilities are available and therefore exogenous to our economic model. This being the case, species arrival and establishment is formalised in the spread or population model as the probabilities of entry (P_{ent}) and establishment (P_{est}) which are combined to give the probability of invasion P_i.[3]

$$P_i = p_{ent} \times p_{est} \quad \text{where} \quad 0 < p_i < 1 \qquad (1).$$

Where established species are concerned we would simply assume $P_i = 1$. In forming a useful spread model to characterise the movement of pest species through an environment once it has become established, we are able to draw from ecological models established over many years. Skellam (1951), for instance, employed reaction diffusion models (originally developed by Fisher (1937)) to model the spread of mutant genotypes in populations, to the spread of muskrat populations in Europe. This kind of model has provided a reasonable approximation for the spread of a diverse range of organisms (Dwyer 1992; Holmes 1993; McCann et al. 2000; Okubo and Levin 2002). A generic result is that a population diffusing from a point source will eventually reach a constant asymptotic radial spread rate of $2\sqrt{rD}$ in all directions, where r is the population's intrinsic rate of growth. Hence we assume that once established the population spreads by a diffusive process such that area occupied expands following the function (Hengeweld 1989; Lewis 1997; Shigesada and Kawasaki 1997):

$$A_t = \pi\left(2t\sqrt{rD}\right)^2 = 4D\pi rt^2 \qquad (2).$$

where:

A_t = area occupied at time t;
D = population diffusion coefficient; and
r = the intrinsic rate of population growth.

It is assumed that the population is in a homogenous environment and expands at an equal rate in each direction. It would be a straight-forward modification of this

[3] Hinchy and Fisher (1991) describe a Markov chain process by which these probabilities change over time according to a vector of transitional probabilities of moving from one pest/disease state to another. This is employed in Cook et al. (2007) and Cook and Matheson (2008).

model to limit the rate of spread to be constrained by area and shape of potentially suitable habitat, or the distribution of the host. We emphasise that where such modifications are not made predictions will tend to overestimate the imperative for eradication or control because neither habitat suitability nor assets at risk will be distributed uniformly across the landscape.

Equation (3) allows prediction of the spread of a species on the basis of an estimated intrinsic rate of population growth, while an estimate of D can be derived from the mean dispersal distance ($\bar{d}$) (Andow et al. 1990):

i.e. $$D = 2\left(\bar{d}\right)^2 \qquad (3).$$

But area alone is insufficient for generating an economic consequence assessment since the density of the invasive species population influences the control measures required to counter the effects of the invasive species. Therefore, the model also assumes that in each unit of area occupied by the expanding population, the local population density $N(t)$ grows over time following a logistic function until the carrying capacity of the environment (K) is reached:

i.e. $$N(t) = \frac{K}{1 + \left(\frac{K}{N_{\min}} - 1\right)e^{-rt}} \qquad (4).$$

Here, $N_{\min}$ is the size of the original influx.

The area and density functions of expressions (2) and (4) may be combined with a logistic function that generates changes in the number of nascent foci (or *satellite* sites) as the invasion process continues. These are events that 'jump' the expanding population beyond the invasion front, and are attributable to causes external to the invasion itself such as weather events, animal or human behaviour. The number of new satellites created in each unit of time ($s(t)$) is:

$$s(t) = \frac{s_{\max}}{1 + \left(\frac{s_{\max}}{s_{\min}} - 1\right)e^{-\mu t}} \qquad (5).$$

Here, μ is the intrinsic rate of new foci generation, $s_{\max}$ is the maximum number of satellite sites generated in a single time step (i.e. 1 year) and $s_{\min}$ is the minimum number of satellite sites generated per time step (Moody and Mack 1988). Once a satellite site is established, the population begins to grow and expand in the same manner as the original population. Total occupied area of the original site and satellites grows until $A_t = A_{\max}$ (maximum habitable area), at which point total area remains constant. This point can be referred to as the *carrying capacity* of the environment. Environmental and demographic stochasticity leading to non-uniform invasion is not considered in such a simple model.

Linking a spread model to a partial equilibrium economic model

The intention of the previous section is to provide a generic spread model. A more specific, detailed model could be used for any species, if data are available, or in the event of an actual incursion, particularly if the pest or disease concerned has been well studied. In this sub-section we link our generic spread model to a simple economic framework. While the biological model predicts the likely spread of an invasive, the economic model will be used to convert this to a measure of cost.

In the first instance, let us assume that after a pest enters and becomes established within a region, no action is taken to mitigate spread or eradicate. This assumption is necessary to form a base case against which intervention or spread mitigation policies can later be compared. The logic is that by comparing the likely costs over time of the pests under a no control and a control scenario, we can easily determine the benefits over time the control in question will bring to the economy. This can then be used in a benefit-cost analysis to determine if it is likely to produce a net benefit for society over time.

The model makes several economic assumptions. It assumes that invasive species affect specific marketable commodities and their production, and is therefore most appropriate to agricultural pests affecting food crops or livestock. Section 2.2 of this report deals with social amenity values and Section 2.3 describes a method for capturing environmental consequences. Section 4 outlines how SDM deals with assigning weights as a descriptor of the relative importance of market and non-market impacts. For now, we assume that the economic model captures invasion consequences on goods that can be quantified and have a market value.

To gauge the real significance of invasives to the economy it is also necessary to assume that producers receive no assistance from public institutions and incur all consequences and control costs themselves. The risk of invasive species incursions is then, in effect, simply a risky production parameter. It is not suggested that such a situation will eventuate, but it is necessary to determine what the true benefits to producers are from maintaining area freedoms, and therefore how much effort should be expended on their maintenance.

Other assumptions necessary in the discussion to follow are that there is one invasive organism of concern to a particular country or region, and that this organism has an impact on one known agricultural or environmental good in a homogenous environment. This host-specificity is purely for notational simplicity. Secondly, assume the domestic market for the potentially affected commodity is perfectly competitive, implying product homogeneity. Thirdly, assume that the contribution of domestic producers of the affected commodity to the total world supply is insufficient to exert influence on the world price, the exchange rate and domestic markets for other commodities. On this basis there are two economic parameters used in determining invasive species-induced producer surplus losses:

1. *Total management cost increments* – Production cost increases result from the need for additional invasive management activities necessary to minimise crop damage. Depending on the nature of the invasive species, this may involve chemical applications (including additional vehicle and labour costs), the destruction of infected/infested hosts, habitat manipulation and/or biological control techniques[4].
2. *Revenue losses* – Firstly, this comprises direct loss of marketable product. Despite incorporating an invasive control program into normal management practice, a certain amount of production loss may still occur through the effects of an introduced organism. This effect may be as high as 100 per cent in some cases, while in others it may be negligible. Secondly, revenue losses may include the loss of export sales. In many cases the loss of pest-freedom status can have profound consequences on export revenue since the ability to sell products to markets around the world is compromised. This does not necessarily mean that all exports of an affected commodity are lost. The subsequent loss of earnings represents a cost associated with an invasive species' naturalisation.

The timing of these costs depends on the organism concerned. For instance, in the case of a crop disease such as Karnal bunt, all wheat exports cease as soon as one outbreak is diagnosed in a country or trading region (Brennan et al. 1992; Stansbury and Pretorius 2001; Thorne et al. 2004). Considering the scale of wheat production in Australia and the linkages to other sectors of the economy, it may be more appropriate to adopt a general equilibrium modelling approach for such pests and diseases (as explained in section 2.1.1). In other cases, the export of susceptible products is only banned from the immediate area of infection (or areas in close proximity to an infected site), as in the case of plant diseases such as Black Sigatoka (*Mycosphaerella fijiensis* Morelet) of bananas (WAQIS 1999). Where exported products have been processed or refined, there may be no loss of export revenue resulting from a pest outbreak at all.

The total area affected by a pest is determined using the spread model. The *expected* consequence at an original site in time period *t* (EC_t) is given by:

$$\mathrm{EC}_t = P_i \times (MDC_i \times N_t \times A_t) \qquad (6).$$

where:

$MDC_i = \text{marginal damage cost for invasive species } i;$

$N_t = \text{invasive density at time } t;$

$A_t = \text{area affected at time } t;$

Here, the average total cost increment and total revenue loss comprises the factors explained above, i.e.:

[4] We make no attempt here to predict the development and availability of new and improved control agents for resistant pests, the likely cost of these products or the capacity of pest species to develop resistance to them.

$$MDC_i = \Delta C_i + \Delta R_i \quad (7).$$

where:

ΔC_i = increase in average total cost of production attributable to species i;

ΔR_i = decrease in total revenue attributable to species i.

A constant marginal damage cost (or average damage cost) is assumed, which can then be combined with the biological spread model.

Reconciling the analytical model and microeconomic theory

Having established an analytical model it is useful to provide some basic microeconomic theory to clarify what it is we are actually measuring when we use it to evaluate the consequence of a pest on an industry. We use a static, partial equilibrium model to explain the economic implications of invasive species. Once again for simplicity, this discussion centres on a species that is host-specific, affecting a commodity, *q*. Recall we assume the following:

i. The organism can be controlled (to some extent) by additional local activities, the costs of which are borne by producers (i.e. raising the Average Total Cost (ATC) of *q* production);
ii. The domestic market for *q* is perfectly competitive;
iii. The domestic price for *q* is above the 'landed' price of imported (identical) product; and
iv. The contribution of Australia to the supply of *q* is insufficient to exert influence on the world price, exchange rate or domestic markets for other goods.

Consider an enterprise producing *q*. The *production function* describes the relationship between physical quantities of factor inputs (*I*) and the physical quantities of output involved in producing *q* given the state of technological knowledge possessed by the producer. So, the level of output she produces is some function, call it *f*, of *I*:

$$q = f(I) \quad (8).$$

In reality, the level of output chosen by the producer is subject to uncertainty. For the purposes of this discussion assume any risky factors in the production process simply take on their average values.[5]

Generally, to be of *biosecurity significance*, *x* must have a negative affect on output when established in a production area such that the quantity of inputs required to

[5] An introductory explanation of the impacts of invasive species and biosecurity policies on output and profit distributions appears in Hinchy and Fisher (1991).

produce any given level of output increases due to the presence of the organism[6]. For instance, should a producer of *q* have to use an additional chemical treatment to those already used for other invasive species control, the costs per unit of outputs must increase. Thus, an invasive species consequence can be seen in much the same light as a negative technological change.

The economic welfare implications of an invasive species-induced change depend on the cost and revenue functions. In short, Total Revenue (TR) for any producer supplying the market for *q* depends on the quantity sold and the price (*p*) at which it is sold (i.e. TR = *pq*), while Total Costs (TC) are a function (call it *c*) of output (i.e. TC = *c*(*q*)). Profit (π) is simply stated as TR minus TC. Given that the price facing a competitive, profit-maximising producer of *q* is dictated by the market as a whole, their profit maximisation decision can be stated as:

$$\max_{q}.\pi = pq - c(q) \qquad (9).$$

To simplify the following discussion *c*(*q*) will not be divided into its fixed and variable components. Let us assume fixed costs of production are zero, so ATC equal average variable costs. Let us also assume that *c*(*q*) demonstrates varying returns to scale such that, with fixed input prices, TC increases with output at a decreasing rate and then at an increasing rate. The gradient of the TC function at any given level of output is termed the Marginal Cost (MC). Given the characteristics of the TC curve, both the MC and ATC curves (plotted in the left hand frame of Figure 1) are u-shaped.

At prices above the minimum value of ATC the MC curve relates the grower's profit-maximising output to price, and thus represents their individual supply curve, *q*(*p*). Where MC is equal to the slope of the TR function (termed Marginal Revenue (MR)), *p*, the differential between TC and TR is maximised. This is a *necessary* (i.e. first order) condition for profit maximisation. Since the slope of the demand curve is constant and zero in a perfectly competitive environment, *sufficiency* (i.e. second order) is dependant on the sign of the cost function, which must be positive if the profit maximising solution lies on the portion of the TC function where MC are increasing. Hence, for profit maximisation we can state that *q*(*p*) must identically satisfy the first-order condition $p \equiv c'[q(p)]$ and the second order condition $c''[q(p)] \geqslant 0$.[7]

The supply curve for the industry can be found by horizontally summing the supply curves of all producers supplying the market for *q*. If there are *n* suppliers and the

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[6] An exception may occur where there are human health and/or environmental implications to invasive species introductions, as mentioned above. This will be discussed in Section 3, but for now assume the only host of *x* is the commodity *q*.

[7] Note that it is not necessarily the case that the producer's choice of output of *q* will be positive. Where the minimum value of ATC exceeds the prevailing market price it is in the interests of a profit-maximising producer to produce no output in order to minimise losses.

supply curve for the i^{th} farm is denoted $q_i(p)$, then the supply curve for the industry ($Q(p)$) is given by:

$$Q(p) = \sum_{i=1}^{n} q_i(p) \qquad (10).$$

This industry supply schedule, which formalises the relationship between industry output and collective marginal costs of production, can be used to calculate industry profit under different production conditions.

Recalling the cost implications of introducing a pest of biosecurity significance, the effect on an individual grower's profit-maximising output decision becomes clear. As the level of inputs needed to produce each unit of q increases in response to costly efforts to keep a newly introduced invasive species at bay, or at least subdued, so too must MC and ATC. The extent of this change is represented by MDC_i in equation (7). To repeat, given the characteristics of $c(q)$ (i.e. increasing returns to scale at low output levels and decreasing returns at higher output levels), the ATC curve will be u-shaped, as depicted in the left frame of Figure 1. Here, two sets of cost curves are shown dealing with both a 'with invasive species' (MC^* and ATC^*) and 'without invasive species' scenario (MC and ATC).

A profit-maximising producer will choose to produce a level of output corresponding to the point where p equals the MC of production. At this point, the difference between total cost and total revenue is maximised. Assuming the prevailing domestic market price, p, is below a closed market equilibrium price (shown here as p_D in the right hand frame of the diagram), a grower characterised by the cost curves MC and ATC would choose to produce quantity q_0 (i.e. where p = MC) and earn a profit of ABCp in the absence of an invasive species. Once again, note that output will be positive as long as the price received by the producer remains above the minimum value of the ATC of production.

If all growers in the industry behave in a similar manner, the industry supply schedule produced by the horizontal summation of each producer's output at different prices would resemble the curve S in the right hand frame of Figure 1[8]. According to the industry demand schedule (D_I) domestic consumers will demand the quantity Q_1 at price p. Of this, Q_0 will be supplied by domestic growers, and Q_1 - Q_0 by imports. In this situation, producer surplus is given by the shaded area HIJ, and consumer surplus by JMN. Note that under a domestic closed-economy equilibrium scenario (i.e. E_D) producer surplus would be the larger area HE_Dp_D, and consumer surplus the smaller area p_DE_DN. Hence, the 'traditional' *gains from trade* is shown as E_DMI.

If an invasive species, x, were to now enter the production region and become established, the effect at the farm level will be rising ATC (and MC), recalling

[8] Note industry demand and supply curves in both Figures 1 and 2 are depicted as linear purely for ease of illustration.

assumption (i) above. A greater cost is now involved in producing each unit of *q* after the outbreak than before it (so $MDC_x > 0$). At the prevailing market price *p* the increased costs of production would lower producer output from q_0 to q* where profit to the individual producer is the heavily shaded area EFG*p*.

If the probability of *x*'s entry and establishment is *P* (recalling equation (1), p. 8), then the expected loss of profit at the farm level (EC_F) associated with the organism can be expressed as:

$$EC_F = P \times (ABCp - EFGp) \tag{11}$$

At an industry level, the domestic supply curve will contract (from S to S^* in the right frame of Figure 1) in the face of added growing costs. Domestic producer surplus will decline to the heavily shaded area KLJ, representing a loss of HILK. So, the expected consequences to the collective industry from *x* (EC_I) can be expressed as:

$$EC_I = P \times HILK \tag{12}$$

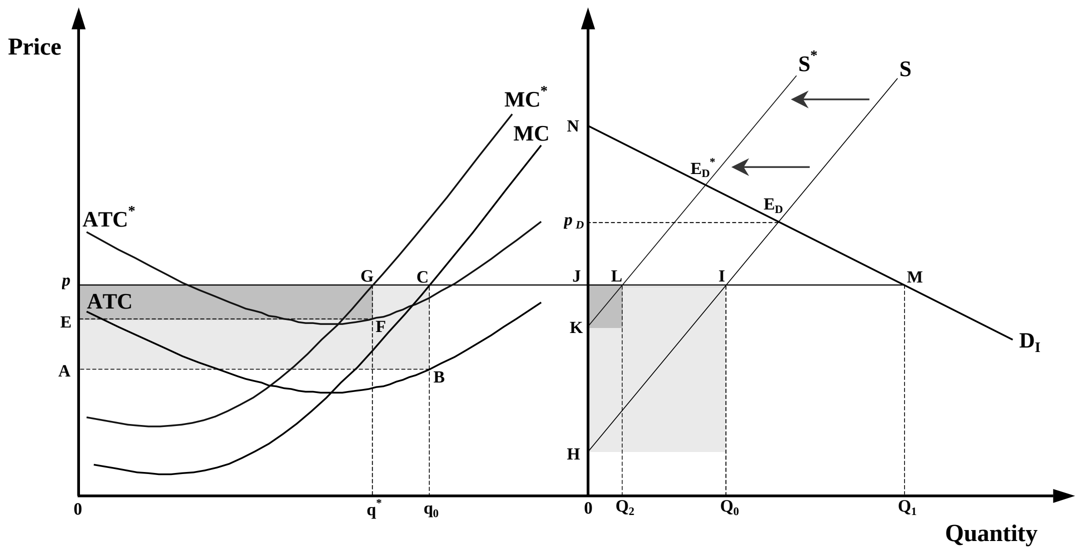

Figure 1. The economic consequence of an invasive species – imported goods.

Assumption (iii) above specifies that the domestic price of *x* is above a world price, but what if we now reverse this assumption? If the world price is now assumed to be above a domestic market equilibrium price, growers can earn more revenue by selling *q* on the world market. The effect of a pest like *x* on an exported commodity is illustrated in Figure 2. Here, the prevailing world price for *q* is p_w. Consider the pre-invasion supply schedule, S_0. At price p_w, the domestic demand schedule in the right hand frame of the diagram reveals the industry is willing to supply Q_0, while the domestic demand for *q* is only Q_1. The industry can sell the residual $Q_0 - Q_1$ and earn a total producer surplus of ABC (shaded). Consumer surplus is the area MNC. A producer within the industry characterised by the cost curves ATC_0 and MC_0 in the left frame of the diagram earns a profit of DEFp by producing and selling q_0 and the price p_w.

Now consider the consequences of the invasive species x on the industry. Once again, necessary changes to the production process to deal with x raise the ATC and MC curves of a typical producer up to ATC_1 and MC_1. They still receive the world price p_w, but it is now only economic to produce q_1, at which they accrue the producer surplus IJKp_w. Therefore, if the probability of entry and establishment of x is P, EC_F can be expressed as:

$$EC_F = P \times (DEFp_w - IJKp_w) \quad (13).$$

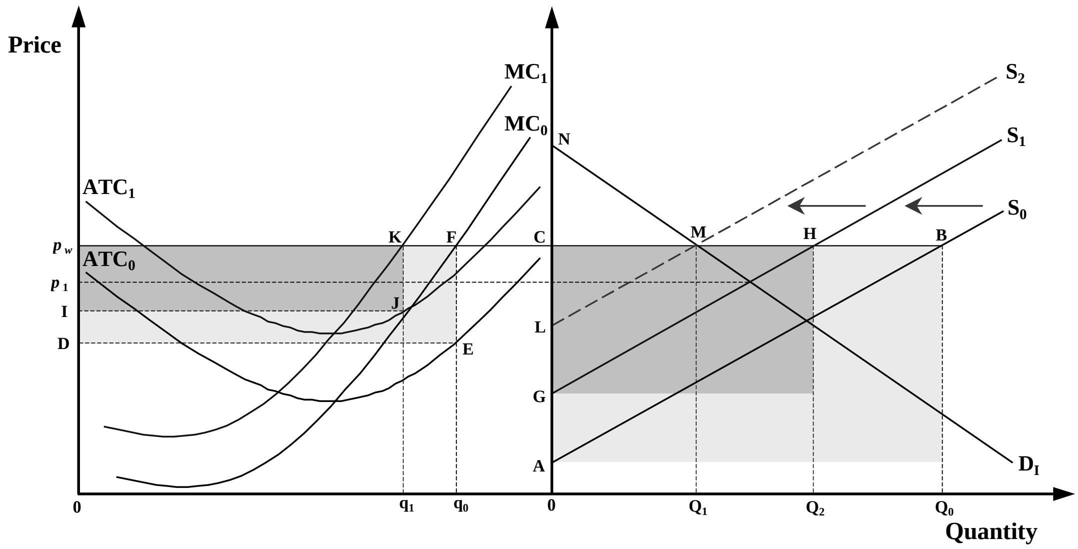

Figure 2. The economic consequences of an invasive species – exported goods.

The aggregate effect of x across the industry is a contraction of the supply curve in the right hand frame of the diagram to S_1. In a closed market this would result in a domestic market price of p_1. But, as this is below p_w the industry can continue to supply the world market and earn more than it would in a closed market. The heavily shaded area GHC indicates total producer surplus. Consumer surplus is unaffected since the price remains at p_w (recalling assumption (iv)), and remains MNC. Hence, in terms of the diagram EC_I can be expressed as:

$$EC_I = P \times ABHG \quad (14).$$

Note that had the contraction in supply induced by the entry of the invasive species been much worse, it could have spelled the end for all exports of the commodity q. If, for instance, the post-invasion supply curve resembles S_2, all exports would cease. The industry could still supply Q_1 to the domestic market, but only earn a producer surplus of LMC. Sales of $Q_0 - Q_1$ would effectively be lost to the effects of x. Note also that at the farm level, such a dramatic cost increase may be sufficient to push individual suppliers out of the market if the minimum value of their ATC function were to exceed p_w.

By describing how an invasive species affects the behaviour of producers, its strategic significance to the economy can be measured. Using the no control or the naturalisation assumption allows us to estimate the true benefit to the economy of keeping a species out, and therefore its biosecurity significance. Measures taken against the pest to either eradicate it, slow its spread or reduce its impact can then be evaluated next to this naturalisation base case.

2.2 SOCIAL AMENITY[9]

This section explores the development and application of measures of social and cultural values in SDM.

2.2.1 *Options to estimate consequences*

Decision-makers increasingly accept that social impacts need to be considered. But capturing and treating social concerns is not a simple process, and is much less developed than is the assessment of monetary impacts. For example, social impacts could change as a policy alternative is implemented, and they may be perceived as negative by some community members and as positive by others. It can be difficult to define measures of social impact. Some social impacts can be captured with natural units or proxies. Natural attributes or proxies are occasionally appropriate for social impacts, but can be difficult to identify in many circumstances. For example, the social dislocation and psychological stress of economic hardship may be adequately captured by the unemployment rate or measures of inequity in income, but generally these are distal descriptors of impact. Where natural attributes or proxies are unavailable we advocate use of constructed scales.

How have social concerns been addressed in multi-attribute decision-making within natural resource management? We conducted a systematic search of the literature using the keywords 'multicriteria decision analysis' OR 'multicriteria evaluation' OR 'multi-criteria analysis' AND 'social', with five search engines (Web of Science, SCOPUS - V.4, Academic Search Premier, Expanded Academic ASAP and ScienceDirect). To qualify for inclusion in our analysis a paper had to describe a complete multi-attribute problem, including identifying objectives, alternatives and a set of attributes to evaluate them. Papers had to include social issues in the analysis of alternatives. A total of 41 peer-reviewed articles satisfied these requirements.

Table 2 shows results of the literature search. The formal capture of social impacts is common in decisions involving energy, waste management and water resources. Only one paper by Cook and Proctor (2007) reported social amenity as a specific consideration in invasive species management. Constructed scales were used to capture and describe social impacts in approximately 50% of case studies. 25% of applications used natural units and 25% used proxies. Typically, natural units were used for employment (e.g. Jones et al. 1990, McDaniels et al. 1999, Gregory 2000). Proxies included the number of cultural sites affected (Apostolakis and Pickett 1998), the number of farms receiving government benefits (Marttunen and Hämäläinen 1995) and population size of towns affected (Sharifi et al. 2002). Constructed scales were developed for community resilience (De Marchi et al. 2000) cultural loss (Cook and Proctor 2007) and socio-psychological effects (Hämäläinen et al. 2000).

Maguire (2004) explored the deployment of decision analysis tools for invasive species. She noted that risk assessments for invasive species management often

[9] We interpret 'social amenity' broadly. It includes consideration of all non-market aspects of human welfare (*sensu* Peterson et al. 1988).

confound the social values at stake in invasive species decisions with the scientific knowledge necessary to predict the likely impacts of management actions. She provided a case study involving feral pigs in Hawaii, where it was obvious that predicting the numbers of pigs in different locations was important, but it was even more important to predict the impact of pigs on native plants, nonnative competitors, mosquito habitat, mosquito populations, disease organisms, and so on. It was more difficult still to express preferences of various human constituencies over the range of predicted outcomes. For example, utility functions expressing relative preferences for different population densities of feral pigs or for different rates of extinction of native forest birds are likely to differ dramatically for Nature Conservancy land managers and native Hawaiian hunters. The 'values' model portion of decision analysis feeds back to the probability-modeling portion by showing where a better prediction of what is likely to happen are necessary to discriminate among alternative management actions versus where decisions can be made with confidence even under considerable uncertainty.

Maguire's (2004) analysis involved a mixture of natural, proxy and constructed scales, appropriate for the case at hand. This is a general property of studies that consider social and cultural values. Typically, they employ a range of measures, and there is no consensus on a 'best' approach. The choice depends on the context of the study, the availability of data, the cultural diversity of stakeholders, their concerns and the interactions of these factors with the biology of the invasive organism.

2.2.2 A suggested approach

When thoughtfully designed, constructed scales can support defensible management decisions. More importantly, they increase stakeholder support for decision-making processes by defining precisely the focus of attention and by incorporating social and cultural concerns that too often are left out of formal evaluations.

There is a tendency to think of constructed scales as subjective and hence less defensible than natural scales. Yet the selection of any scale (including natural scales) requires subjective judgments; the bottom line of deciding that a particular scale (of any type) is appropriate or reasonable to include as part of a problem description is a subjective judgment.

There also is a misperception that constructed scales are somehow more simplistic, or carry less information, than do other scales. So long as a constructed scale is carefully developed, it can convey substantial context-specific information about the problem under consideration. Constructed scales are flexible. They may involve quantity-quantity comparisons, weighted scales, and scales built using value models. We provide examples below.

Table 2. The number (*n*) of multi-criteria decision analysis publications that address social impacts according to decision domain.

Domain	*n*	Topics	Studies
Energy systems	15	Bioenergy, coal energy, domestic electricity consumption, energy policy, energy power system, hydrogen energy, nuclear power, renewable energy, small-scale energy, wind energy	Keeney and Sicherman (1983), Jones et al. (1990), Georgopoulou et al. (1997), Haralambopoulos and Polatidis (2003), Afgan and Carvalho (2004), Cavallaro and Ciraolo (2005), Lipošćak et al. (2006), Begić and Afgan (2007), Burton and Hubacek (2007), Doukas et al. (2007), Gamboa and Munda (2007), Chatzimouratidis and Pilavachi (2008), Buchholz et al. (2009), Kahraman et al. (2009), Browne et al. (2010)
Forest resources	6	Fire control, forest planning, timber harvesting	Gregory (2000b), Kangas et al. (2001), Russell et al. (2001), Laukkanen et al. (2002), Ananda and Herath (2003), Laukkanen et al. (2004)
Environmental restoration and waste management	6	Contaminated land management, energy-from-waste, hazardous waste landfill, nuclear and radioactive accident, wastewater projects,	Apostolakis and Pickett (1998), Al-Rashdan et al. (1999), Hämäläinen et al. (2000), Longden et al. (2007), Sorvari and Seppälä (2010)
River and water management	6	Recreation and tourism activities, river basin planning, water resources availability, water supply alternatives, wetland management	Duckstein et al. (1982), Marttunen and Hämäläinen (1995), De Marchi et al. (2000), Herath (2004), Proctor and Drechsler (2006), Salgado et al. (2009)
Conservation biology	4	Marine protected area management, park management, wilderness preservation	McDaniels and Roessler (1998), McDaniels et al. (1999), Brown et al. (2001), Sharifi et al. (2002)
Marine resources	2	Coral reef management, fisheries management	McDaniels (1995), Fernandes et al. (1999)
Land management	1	Land resource allocation	Malczewski et al. (1997)
Invasive species	1	Exotic plant pests	Cook and Proctor (2007)

At the later stages of a policy assessment process, the introduction of constructed scales can facilitate the explicit consideration of tradeoffs across stated levels of different types of objectives, which helps to maintain transparency in decision making and broaden the base for dialogue among participating interests. For example, decision makers can ask whether it is worth taking an additional six months as part of a consultation process to increase support from a particular stakeholder group from level 4 to level 2.

A common setting for the development of a constructed scale makes use of either a narrative or the results from studies to define different levels of impact. As one example of what is commonly termed a 'defined-level' scale, consider the constructed scale shown in Table 3, used to measure community support for a proposed management practice (modified from Keeney and Sicherman, 1983). Because no natural scale exists to measure support, an index (e.g., 1-5) can be created, with each rating denoting a different level of support. Level one could be defined as 'strong action-oriented opposition' and a level 5 as 'support.' Note that the ratings in constructed scales do not need to be symmetric; in this case, the neutral level is given a rating of 4 rather than 3.

Constructed scales developed for describing biosecurity impacts can make use of an explicit value model to capture the relative significance of underlying factors. An example would be an attempt to measure the anticipated effects on a loss in public amenity values, such as recreation activities or aesthetics, due to a pest outbreak. Consider the case of a possible reduction in the number of beach picnics as the result of an invasive pest that interfered with a family's enjoyment of an important outdoor experience. One way to estimate this loss in value would be to use any of a variety of methods from economics (e.g., contingent valuation methods, travel cost models) to develop a measure of residents' willingness to pay to reduce or eliminate this pest. For many people, however, the required line of questioning (e.g., 'how much would you be willing to pay to reduce or eliminate the level of annoyance') would make little sense, since this is not a normal market transaction and beach access generally is free.

A different approach to developing a constructed scale for recreational losses would be to highlight the likely number of people to be affected at different levels of pest invasion. For example, experts in outdoor leisure recreation or tourism could be asked to estimate the range of people who might be affected, with this information translated into the following five-point scale (Table 4). Note that the scale also helps to bound the anticipated impact; the largest event would affect no more than 20,000 people, which could be significant on a local or regional scale but perhaps not at a national level.

Table 3. Constructed scale to measure anticipated public support.

Attribute level	Description: level of public support
1	Strong action-oriented opposition: two or more groups have organised opposition to the action
2	Action oriented opposition: one group is actively opposed whereas others appear to be neutral.
3	Controversy: one or more groups are opposed, but not actively. Other groups appear to be neutral.
4	Neutrality: all groups appear to be indifferent or uninterested.
5	Support: no groups are opposed to the action and at least one group shows organised support.

Table 4. Constructed scale to measure recreation loss.

Scale Number	A	B	C	D	E
Description	No loss for any recreationists	A small sense of loss for a small number of people(1-100)	A large sense of loss for a small number of people (< 1000)	A small sense of loss for a large number of people (1000 -20,000)	A large sense of loss for a large number of people (10,000 – 20,000)
Value	0	0.1	0.5	0.6	1

An alternative and possibly more broadly acceptable approach would be to develop a constructed scale using a value model that might seek to minimise the likelihood of having this beach pest interfere with a scheduled family picnic. This approach to developing a constructed scale would highlight the understanding that the significance of any loss in amenity value would depend on how many picnickers are affected and the severity of the effect. In this version there could be a quality variable (s= severity of the impact, expressed as a proportion of amenity value lost) and a quantity variable (n = number of picnicker-days). A value model would show the effect on the public picnic amenity as

$$p = s \times n,$$

calculated as a weighted sum over 3 - 5 levels of impact severity. This constructed scale is known as a quality – quantity scale for constructed attributes (as described in Keeney, 1992).

Constructed scales also could be useful in highlighting potential triggers that might constitute no-go zones or stopping points for potential biosecurity management actions. One context is when management actions might be viewed as taboo from a cultural perspective, for example when pest eradication activities are proposed for a location sacred to indigenous people. In such cases, responses from managers that might involve decreasing the quantity of an action or changing its timing are likely to have no effect because an irrevocable cultural or spiritual practice has been violated (Turner et al. 2008).

Suppose that eradication of a pest required that the host animal be destroyed. If this host animal is sacred to a group of citizens in Australia, then its destruction may be strongly opposed. In such cases a constructed scale could be very simple, composed of two points (1 = culturally acceptable to an indigenous population, 2 = not acceptable) or perhaps three (with an intermediate rating of 'acceptable under carefully specified and exceptional conditions.').

Obviously this type of constructed scale is only a placeholder for a much richer dialogue, but it serves the purpose of making a concern visible and putting the loss created by violation of a principle into the same matrix of consequences as other, more conventional expressions of impact. In this example, the scale would either highlight for biosecurity managers that vigorous opposition to an action should be expected or that some other (even if more expensive or less effective) management action would need to be considered.

Constructed scales have been used in other contexts to assist in evaluating this type of issue. In British Columbia, for example, an attribute was created as part of multi-stakeholder water management deliberations to consider which alternatives for a managed river system would violate a fundamental principle of the We Wai Kum (Campbell River) First Nation, which is that water which naturally flows in one direction should not be made to flow in the opposite direction.

A related plant importation example involves the possible importation of a genetically modified plant or pest. Despite potential economic, trade or health benefits associated with this import, some part of the regional or national population might irrevocably be opposed on the basis of a fundamental fear of this type of GM organism. This concern is now being debated in New Zealand in the context of the possible importation of GMO foods. Earlier work by Maori tribes in conjunction with western scientists (Tipa and Tierney, 2003) defined a four-component constructed scale for 'cultural health' impacts, based on interviews with elders who identified key indicators pertaining to the health of waterways. More recent interviews (Satterfield et al. 2009) have helped to identify constructed scales for Maori values that underlie the GMO decision making process, emphasizing the need to incorporate and attend closely to the principle of 'tikanga' (which, loosely translated, refers to the collective understanding of correct practices and customs).

A final example describes the development of an explicit constructed attribute scale for measuring impacts on cultural and spiritual quality. An example comes from the development of a BC Hydro Water Use Plan for the Bridge River hydroelectric system in British Columbia, where as part of a multi-party resource management consultation one of the 'things that matters' to local participants from the aboriginal St'at'imc Nation is the 'spirit' or 'voice' of the river, which became the basis for development of a Cultural and Spiritual Quality constructed scale (Failing et al. 2010).

Both St’at’imc elders and western scientists observed that in moving from a water budget flow rate of 0 to 3 m^3/s, there have been noticeable improvements in conditions for conventional ecological outcomes like fish, wildlife, and riparian vegetation. But in addition, elders observed that there were significant improvements in the ‘spirit’ or ‘voice’ of the river. To obtain information to better define this objective, we collected input from interviews with St’at’imc people, and were able to define four key components of Cultural and Spiritual Quality:

- Sound, including: the voice of the water and birdsong.
- Smell, including the smell of the water itself and the ambient smell at water’s edge.
- Movement, including the movement of water (seasonally appropriate) and the diversity of movement (pools/riffles).
- Interaction (of people and water), including shore access and ‘wade-ability’ (the ability to walk in and/or across the river at certain locations).

These four components clearly do not provide a universal definition of cultural or spiritual quality - they define the aspects of cultural and spiritual quality thought to be relevant for the evaluation by St’at’imc of a suite of alternative flow regimes and habitat enhancement activities on the Lower Bridge River, within the flow ranges considered. As with any other scale (natural, proxy, or constructed), further questions relating to specific measurement and timing considerations are also critical; in this case, discussions concluded that obtaining quantitative measures using the scale would involve:

- A committee of three to eight St’at’imc members to act as observers.
- Observations to be taken four times per year under a range of test flows.
- Observations to be taken at two designated sites per reach over three reaches.
- A visual record at each observation site using video camera and still photography.
- A simple, replicable and transparent scoring system for assigning scores to each component in each reach; and
- A transparent and defensible methodology for weighting and aggregating scores across observers, components, reaches and seasons.

Development of this constructed scale not only greatly improved the quality of consultations and relationships among participants, it also decreased a very real threat of protests and/or lawsuits. By demonstrating a level of commitment and rigour for this constructed spiritual and cultural quality scale that is equal to that given to a science-based scale, an important benefit has been the slow creation of trust (and overcoming of historical mistrust) between the utility and the local aboriginal population. Its development also has opened a door to discussions of trade-offs among objectives—for example, one flow alternative may prove to be more or less beneficial for salmon but less or more beneficial from the perspective of cultural and spiritual quality, in which case choices now can be addressed based on the preferred balance across defined objectives.

A final consideration is that attributes are not intended to only capture concerns or issues, they also seek to depict and convey emotions to the extent that such affective considerations arise as part of the consideration of different biosecurity risk management alternatives. One place that often invokes a strong emotional response is with reference to possible changes in aesthetics. For example, efforts to locally eradicate willows often encounter resistance on the grounds of the aesthetic appeal of willows. In such cases, any constructed scale that attempts to capture people's feelings in words or numbers may be dismissed as insensitive or misleading. One alternative that has been used in constructed attribute scales is to make use of pictures – either photographs or drawings – that depict the possible changes in aesthetics. Standards for using a visual attribute are similar to those for a defined level scale, as discussed earlier: the range of impacts is covered through pictures showing the worst to best outcomes, with each level characterised by a different picture. A well-known example of such a constructed scale concerned the evaluation of treatments for children with the congenital problem of cleft lip and palate (Krischer, 1976) The development of this creative scale helped to influence clinical decision making and eased both communication and the understanding of different treatment options among children, families, and doctors.

Defining good attributes is not easy, especially for social impacts where the concerns to be captured are not universal (like the monetary impacts of pests on agricultural productivity), but rather demand specific consideration of the decision context and affected stakeholders. Constructed scales offer a flexible approach for accommodating decision-specific concerns.

Several of the examples in the previous section have emphasised that one of the key advantages of an SDM approach is its ability to put hard-to-quantify amenity objectives on a level playing field with more conventional quantifiable ones. Although users of SDM and other decision analytic approaches have met with some success, each decision context presents its own set of considerations and there is no 'one-size-fits-all' approach to the development of responsive attributes.

A key challenge is often how to develop a responsive and comprehensive yet concise set of attributes. For example, it is often difficult to describe all the relevant aspects of social considerations as part of a small set of attributes. Weighted indices provide a useful and helpful approach; for example, different types of social implications of a proposed management action can be weighted, normalised and aggregated to create a single index of the anticipated effects. Yet this requires information collection, which may involve the aggregation of data of differing quality, and it requires explicit weighting of the components of the index, which can be difficult in terms of reaching agreement among the different management agencies, involved citizens, or decision makers. Issues of comparability also need to be addressed, because at some point value-based comparisons will need to be made across the different attributes that describe a set of policy alternatives (Gregory and Failing, 2002).

2.3 ENVIRONMENT

For environmental pests, consequences may involve marketable goods, e.g. a reduction in water supplies caused by extraction by invasive alien tree species (van Wilgen et al. 2001). More often, however, the environmental goods affected by the pest will not have a clear market value (e.g. the conservation of biodiversity). To date, there has only been limited success in quantifying consequences of invasive species on environmental goods such as biodiversity (OTA 1993), and it is difficult to incorporate non-market information in general models.

Estimation of the environmental consequences of pest invasion demands clear description of fundamental concerns or objectives. In particular there is a tendency to double count impacts where ecosystem services providing social or economic outcomes are confused with the objective of minimising impact on biological heritage. This section outlines options for estimating environmental impacts and then describes a suggested approach.

2.3.1 *Options to estimate consequences*

Visual structuring tools such as means-ends networks (Keeney, 1992; Clemen, 1996) and influence diagrams (Schacter, 1986) are useful for identifying fundamental objectives. These visual tools link the things that can be controlled (the management actions) to the 'ends' – what people fundamentally care about. The use of these tools aids in the development of clear objectives and evaluation criteria that will be used to assess different alternatives.

For example, consider the means-ends diagram shown below (see Figure 3). If a farmer cares about how the introduction of a new species might affect the economic value of a crop (a fundamental objective), then the water retention of the new species, its ability to disperse in the environment, and its resistance to herbicides may all be important considerations because of their impact on the means objectives associated with controlling weeds and avoiding increases in soil salinity. These considerations influence crop production that, in turn, affects the economic value of the farmer's crop. Means are important to the overall decision making process, but they need to be identified and treated differently so as to help clarify appropriate characterisation of consequences.

The maintenance of ecosystem services is a 'means' objective that relates to a range of fundamental 'ends' objectives, including primary production and social amenity. This section assumes that, where they are deemed important, impacts of ecosystem services are accounted for in estimates of impacts on economic and social objectives. The fundamental objective for environmental impacts is taken to be the conservation of our biological heritage, for which the appropriate proxy is the minimisation of species loss.

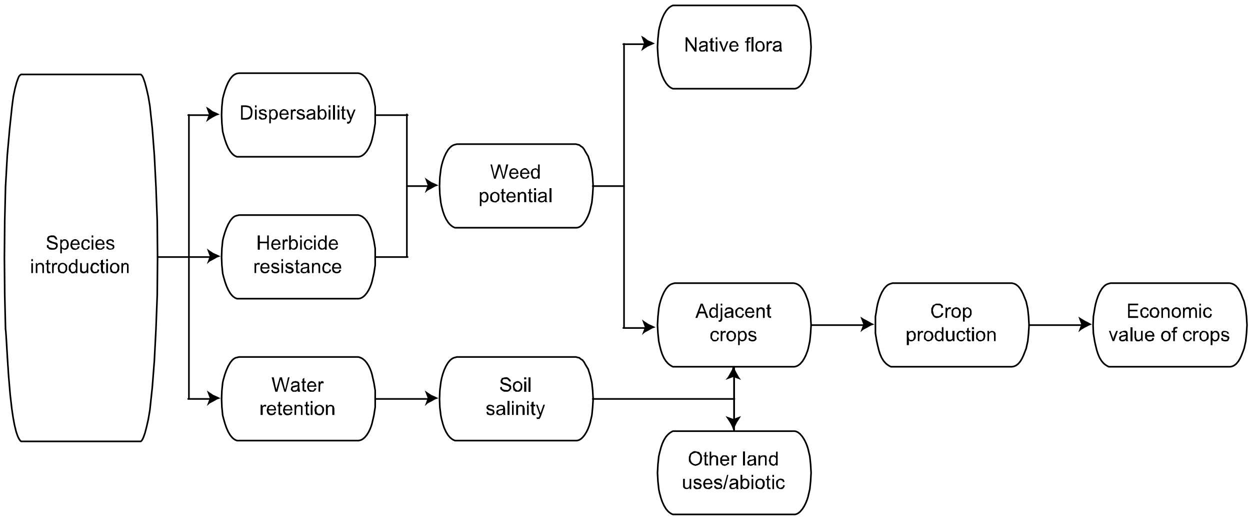

Figure 3. Example means-ends diagram.

There are several approaches to estimate species loss. An indirect approach is to build a constructed scale on the basis that habitat loss is proportional to extent (spatial scale) of impact × intensity of impact (Burgman and Lindenmayer 1998). This is an example of one of the more common variations of constructed scales - a weighted scale, where two or more concerns are combined to determine a metric. For example, the ecological importance of an impact could be measured using a 'weighted habitat loss' scale that combines importance (e.g., reflecting the contribution of a species to overall ecosystem productivity) and the degree of degradation (e.g., measured as square kilometres of expected habitat loss). In such cases, explicit value weights would be elicited for the different levels of impact shown on the scale. The illustrative weighted scale shown below in Table 5 incorporates the non-linear valuations of each defined category, so that a rating of five on the scale is more than five times worse than a rating of one.

Table 5. A constructed scale incorporating extent of habitat loss and its importance.

Scale Number	1	2	3	4	5
Description	No habitat loss	Less than 10 km sq of habitat loss; minimal harm to species and ecosystem values expected	Between 10 and 100 km sq of habitat loss; somewhat significant harm to species and ecosystem values expected.	Between 100 and 250 km sq of habitat loss; significant harm to species and ecosystem values expected.	More than 250 km sq of habitat loss; very significant harm to species and ecosystem values expected.
Value	0	0.1	0.25	0.5	1

Point scoring procedures and rule sets

Government and nongovernmental organizations around the world use a range of protocols to evaluate the level of threat to species. These protocols vary from qualitative assessments, where threat level is inferred from information on the species such as geographic range, population size, number of populations, and trends in these attributes based on expert judgment (Master 1991), to more transparent methods that use rules or point-scoring approaches which infer threat from these attributes (e.g. Millsap et al. 1990; Lunney et al. 1996; Carter et al. 2000; World Conservation Union 1994). Outcomes are used to assess potential adverse impacts on species, to determine whether there is a need for legal or other protections of species, to set management priorities for resource allocation (e.g., reserve design and recovery planning), and for state of the environment reporting (see Regan et al. 2005).

The following examples illustrate the use of three different approaches to threat assessment.

- The US NatureServe protocol assesses species according to 12 biological and external factors that may affect their persistence (Master 1991). Each factor is scored in quantitative ranges, and then experts use this information in a qualitative manner to derive a 'conservation status rank', a numeric code that reflects the relative risk of extinction of an element (a species or an ecological community or system) (Master et al. 2003).

- Millsap et al. (1990) designed a point scoring prioritization scheme to rank Florida's vertebrate fauna according to their conservation needs. Biological variables reflect population status and life history. A score ≥33 places species in the 'endangered' category, between 33 and 29 a species is deemed as threatened and between 29 and 24 points a species is considered of special concern. A score of <24 points indicates the species is thought to be non-threatened.

- The IUCN Red List Criteria (IUCN 2009) are a set of decision rules that have five criteria (A–E), which require ecological data on range size, population size, rates of decline, and other parameters. The threshold values for each parameter differ for each of the threat categories, critically endangered, endangered, and vulnerable. Meeting any one of these criteria qualifies a taxon for listing at that level of threat.

All of these systems (and others like them developed in various jurisdictions around the world) provide estimates of the levels of threat faced by taxa. The categories of threat into which species are classified are proxies for the risk of extinction. Of the systems currently available, the IUCN protocol is the most widely accepted internationally. However, it provides no means for estimating the reduction in risk that may eventuate from management of an invasive species.

Population viability analysis

Explicit estimation of risk and risk reduction resulting from the management of invasive organisms is most readily accomplished through model-based risk assessments for species, called Population Viability Analyses (PVA). In this context, risk is viewed as the magnitude of a decline of a population (or a set of populations) within some time frame, and the probability that a decline of that magnitude will occur. The objective of PVA is to provide insight into how resource managers can influence the probability of extinction (Boyce 1992, Possingham et al. 1993). PVA may be seen as any systematic attempt to understand the processes that make a population vulnerable to decline or extinction (see Burgman et al. 1993, and Regan et al. 2005).

The most appropriate model structure for a population depends on the availability of data, the essential features of the ecology of the species or population, and the kinds of questions that the managers of the population need to answer. The model may include elements of age or stage structure, spatial distribution of habitat, dispersal, behavioural ecology, predation, competition, density dependence, or any other ecological mechanism that is important in determining the future of the population.

Once the deterministic form of the model is established, elements of stochasticity are added to represent specific kinds of uncertainty. The inherent variability that results from random birth and death processes may be represented by sampling survival parameters from Binomial distributions, and by sampling the number of births per adult from a Poisson distribution (Akçakaya 1990). Environmental uncertainty may be represented by time-dependent survivorships and fecundities, often sampled from lognormal distributions (Burgman et al. 1993). The result is a cloud of possibilities for the future of the population. These projections are analysed to generate cumulative distributions of falling below specified population sizes within the time frame of the analyses (the probability of quasi-extinction; Burgman et al. 1993).

It may be important to distinguish between different genders, ages or life stages because, for instance, different kinds of individuals respond in different ways to management activities. Age or stage structure is simply a matter of replicating the equations for birth and death for the different stages, and incrementing the composition of the stages appropriately. Variation in the environment also will affect survival and fecundity, driving the parameters down in poor years.

Wildlife managers implement plans to minimise risks of decline, and sometimes to maintain populations within specified limits, so that the chances of both increase and decline are managed. The probabilities generated by stochastic models allow different kinds of questions such as; What is the worst possible outcome for the population? or, Which parameter is most important if variability resulting from competition from invasive organisms or disease can be reduced? Answers to such questions are a kind of sensitivity analysis. They provide guidance on where it would be best to spend resources in field measurements to estimate parameter as accurately as possible, and to understand ecological processes.

PVAs are hampered by lack of data and lack of validation. Results often are sensitive to uncertainty in the data (Taylor 1995). It can be difficult to verify stochastic predictions. Models are sometimes misinterpreted and they impose overhead in terms of computational effort and technical skill that may not be warranted by the problem at hand (Beissinger and Westphal 1998, Burgman 2005).

If all that is required is an estimate of risk, subjective judgments and other approaches may be cost-effective. However, model-based risk assessments have additional advantages (McCarthy et al. 2004). The rationale behind their predictions is explicit. The models are open to analysis, criticism and modification when new information becomes available. Their assumptions may be tested. The models can be used to help design data collection strategies. They help to resolve inconsistencies. Models may be more useful for their heuristic rather than their predictive capacities.

2.3.2 A suggested approach

Unfortunately, the time, resources and skills necessary to develop PVAs may be unavailable, especially for the full set of species that may be impacted by an invasive organism. Instead, simpler approaches may be warranted, and this is likely to be the case in most biosecurity contexts for the foreseeable future. Point scoring and rule based systems may be developed to work well enough for some situations. But there are useful approximations that make more complete use of available information, and are less susceptible to subjective interpretation than more qualitative approaches. These are outlined below.

Plants

The method for assessing the potential impacts of invasive pests and diseases on plant species of concern depends on the following general principles (following Burgman et al. 2001): (1) All populations face some risk of decline because they are exposed to unpredictable natural events, even in habitat unaffected by pests or diseases. These so-called 'background' risks can be approximated by generic population models that account for environmental and demographic variation. (2) Measurements of the consequences of biosecurity impacts on the environment are limited to 'added risk', the increase in risk of extinction that arises from exposure to a pest or disease. (3) Australian environments are characterised by natural disturbance regimes such as fire, drought, and predation with characteristic scales and effects, to which plants are adapted. (4) Impacts can be estimated by calculating the reduction in the expected proportion of effective habitat available to a species or ecosystem, and translated into measures of the added risk.

The estimation method is divided into ten steps, each of which accounts for an aspect of habitat or a deterministic or stochastic process that affects the area that supports a natural population. The data for implementation of the method are relatively modest (Table 6) compared with those required of detailed population models (Boyce 1992; Beissinger and Westphal 1998).

Table 6. Definitions of parameters used in the equations that make up the method for estimating pest and disease impacts on plants.

Parameter	Definition
H_i	area of potential habitat in disturbance region *i* (ha)
N_i	number of adult plants within potential habitat, H_i
F	the population size that faces a 0.1% chance of falling below 50 adults at least once in the next 50 years, assuming no detrimental effects from the focal invasive pests or diseases (a benchmark for evaluating risks across different taxa)
p_i	annual probability of disturbance (or the proportion of habitat disturbed annually) in disturbance region, *i*
n_d	time taken for the species to recover from disturbance
n_u	number of years after a disturbance until habitat is no longer suitable for the species (assuming no further disturbance)
L_i	proportion of remaining habitat lost to a deterministic process, *i*
r_i	proportional reduction in local density due to effect, *i*, that reduces populations within their habitat, such as disease, predation and competition

The approach here requires that the analyst provide a best estimate and plausible upper and lower bounds for each parameter. Methods for estimation are outlined below. In most instances, confidence intervals or other formal statistics of dispersion are unavailable, in which case bounds may be estimated subjectively (Seiler and Alvarez 1996, Speirs-Bridge et al. 2010). Uncertainties in the parameters can be incorporated by applying the rules of interval arithmetic (Alefeld and Herzberger 1983).

The method requires an assessment endpoint for species loss. Here we nominate a 0.1% probability of falling below 50 adults at least once in 50 years, to provide a background risk against which to measure added risk.

The benchmark of 50 years reflects the fact that concerns are with risks on a scale over which current management prescriptions may be effective. Risks measured over relatively short time frames sometimes suggest management actions at odds with those measured over longer periods (Menges 1998). Over longer periods, other priorities and environmental strategies are likely to take precedence. The benchmark of 50 adults acts as a common reference point for a variety of different taxa and represents the lower bound for the size of the population we find unacceptably small for any species.

We elected to concentrate on adult plants, defined as reproductively mature individuals, to provide a means of dealing with species with different life forms and life histories and to remain consistent with the conventions of the World Conservation Union (1994). For example, many plants have soil-stored seeds that provide a buffer against adverse environmental events, whereas others persist by means of underground organs or dormancy. These factors are accounted for in the estimation of the parameters for the equations used to calculate sufficient population sizes experiencing background disturbance regimes. Overall, the criteria represent a modest target for the conservation of species within a realistic management time frame.

The quasiextinction risk criterion is expressed in terms of current population size by estimating an initial population size for each taxon such that there is 0.1% chance of the population declining to 50 individuals at least once over the next 50 years. This implies that it is acceptable for about 20 of the approximately 20,000 Australian vascular plants to become critically endangered within the next 50 years.

Step 1: Map potential habitat.
This method assumes that a map of the potential habitat for the species of concern is available. Potential habitat may be defined as all areas considered by an expert to be capable of supporting viable populations of the species. Alternatively, it may be defined by a set of spatial climate and/or environmental data layers and a bioclimatic model, or by a regression model of existing locations together with data layers for each of the explanatory variables (Austin et al. 1990; Elith 2000). Caution must be exercised in estimating the area of potential habitat to account for competition, predation, and disturbance, which might exclude a species from otherwise suitable locations.

Step 2: Delineate disturbance regions.
Analysts should identify populations or groups of populations that experience similar disturbance regimes, here termed disturbance regions (equivalent to 'locations' in the IUCN Red List protocols). Disturbance regions represent areas of the landscape subject to similar sources and intensities of disturbance. The area of habitat within each disturbance region is H_i in Table 6. These disturbances typically affect areas smaller than the extent of the species' habitat.

Disturbance regions capture differences in the frequency and extent of disturbances that affect the abundance and distribution of the plant species in question. Care should be taken to include only those disturbance processes that have not contributed to the natural variation that underlies the estimation of F, above. In many cases land tenure may be a reasonable guide to defining disturbance regions. Perform all subsequent analyses on each disturbance region.

Step 3: Estimate the density of adult populations
Analysts estimate the number of adults, N_i, within specified areas of *occupied* habitat in each disturbance region, i. Estimates may be derived from quantitative survey information or expert knowledge. Population sizes and the areas of occupied habitat patches (A_i) are then used to estimate the density (D_i) of adult populations in each disturbance region, i,

$D_i = N_i / A_i$ (16)

The population density, D_i, should represent the average density of reproductively mature plants within occupied habitat.

It may be difficult and time-consuming to have the experts arrive at density figures for each disturbance region. It may be preferable to use the density figure based on a single habitat model for all calculations. In most cases, the long-term average density will be best reflected in the disturbance region that has been subjected to least anthropogenic disturbance.

Step 4: Identify background disturbances

Background disturbances are the processes used to define disturbance regions in step 3 above. The annual extent of these disturbances is smaller than the species habitat and the species typically recovers from their impacts within a management time frame of 50 years. The method uses estimates of the characteristics of these disturbances to calculate the proportion, *S*, of potential habitat, available to the species at any one time.

Stochastic events that may cause an area to become unsuitable could include a single event such as a prescribed fire at a particular time of year or a logging event. More typically it will be a combination of events such as two or more fires within a short time. These are termed adverse regimes 1, 2, 3. . . . The average annual area of these effects should generally be less than the total potential habitat. We assumed that these events are randomly and independently distributed across the landscape with respect to the distribution of the taxon. This is a useful model for a surprisingly broad class of disturbance processes (Gardner et al. 1987; Johnson and Gutsell 1994; Pacala et al. 1996; McCarthy and Gill 1997).

If a disturbance has a characteristic annual probability independently distributed across the landscape, the expected proportion of areas that are *n* years old is equal to the probability that an area was disturbed *n* years ago (*p*), multiplied by the probability that the area was not disturbed subsequently, $(1 - p)^{n-1}$ (McCarthy and Burgman 1995). The proportion of the landscape expected to be disturbed each year is equal to the probability that a point in the landscape will be disturbed. For example, if 10% of the landscape burns each year, then there is an annual probability of 0.1 that a random point in the landscape will burn.

Relatively small-scale disturbances are modeled as processes that have similar consequences for the ecology of the species in question. The parameters *p* may be estimated if any of the following characteristics are known or can be estimated:

- proportion of the landscape (or population) that is, on average, more than *n* years old;
- proportion of the landscape (or population) that is, on average, less than *n* years old;
- average size of disturbance events (annual total area disturbed within the potential habitat); or

- return time between events (average time between disturbances at a point in the landscape).

This information may be based on expert knowledge of disturbance processes or on recorded information such as fire perimeter records and spatial and temporal analysis of disturbance regimes. It may be that the disturbance regimes are too complex to allow a reliable estimate of the parameters p and n. In such circumstances, it may be easier to estimate S directly.

Step 5. Estimate recovery time
Habitat requires n years before it again supports adults—known as the *recovery time*—the time between disturbance and the re-appearance of reproductively mature plants. The parameter n_d is the time between disturbance and the point at which a plant has developed sufficiently to reproduce. It includes the time to reach reproductive maturity for plants eliminated by recurrent disturbance, such as obligate seeders.

We also define n_u, the time between disturbance and the point at which the habitat has developed so that it is unsuitable for the species. This time period is relevant for species that inhabit early successional stages within a landscape and that rely on periodic disturbances of particular kinds for germination or regeneration. For these species, the absence of a disturbance may result in unfavorable habitat beyond n_u years.

Step 6. Estimate the proportion of habitat that is outside the recovery window
Given p_x, the proportion of the potential habitat within a disturbance region disturbed on average each year by process x, the proportion of the landscape, q_u, that is undisturbed each year by a total of z disturbance processes is

$q_u = (1-p_1)(1-p_2)\ldots(1-p_z)$

where p_1, p_2 are the probabilities of disturbance from processes 1, 2, and so on for z independent processes.

Given n_d and n_u, the average proportion S of the potential habitat that will be suitable for the species at any time within a management time horizon of 50 years (that is, the effective proportion of habitat), accounting for disturbances that are either too frequent or too infrequent, is

$$S = q_u^{nd} - q_u^{nu} \quad (16)$$

The parameters n_d and n_u define the window of opportunity for the species. Before n_d the area is too young for a seed-producing individual to have developed, and after n_u the area is too old to support the species. This assumes that the recovery time following disturbance is the same, irrespective of the kind of disturbance.

For example, assume a taxon is adversely affected by unseasonal (prescribed) fire, and this disturbance regime is imposed on the background of a natural fire regime. The taxon recovers naturally after fire because its soil-stored seed bank is stimulated to regenerate by fire. Suppose, however, that a 10-year lag occurs between the fire event and the development of adults that will replenish the seed bank (n_d = 10). If the additional fire events burn about 1/80 of the potential habitat annually, then the probability of disturbance for a site, p, is 1/80. We assume that there is no upper bound, n_u, in this example. The proportion of the potential habitat that will be suitable for the taxon, given this additional source of disturbance, is

$$S = \left(1 - \frac{1}{80}\right)^{10} = 88\%$$

and

$$H_{i,s} = H_i S \tag{17}$$

That is, about 12% of the potential habitat, on average, will support populations that are too young to withstand other disturbances such as unplanned wildfires, because they will not have produced seed to replenish the seed bank that was depleted following the most recent disturbance. Fire-management activities effectively reduce available habitat by 12%.

At this point, it is possible to verify that the conceptual models are consistent. Within each disturbance region, the area of occupied habitat should be approximately equal to $H_{i,s}$, the area of potential habitat multiplied by the proportion of habitat that lies within the disturbance window. That is, occupied habitat area estimates should exclude habitat that is temporarily unoccupied due to stochastic disturbances.

Step 7: Estimate habitat area affected by existing deterministic impacts
Impact calculations assume that disturbances affect species either by exerting deterministic trends on the species' potential habitat, or by reducing population density. Adverse deterministic trends cause permanent loss of habitat (at least within the management time frame). For example, loss of habitat may be due to land clearance for agriculture or urban expansion. The parameter L is the rate of loss of potential habitat (the proportion of remaining habitat lost to this process) per year due to irreversible attrition. The extent of the potential habitat remaining at the end of 50 years is $H_i(1 - L)^{50}$.

If there is more than one such process in a region, its rate should be estimated independently and the losses of habitat summed,

$$H_{i,s,d} = H_{i,s}(c_1(1\text{-}L_1)^{50} + c_2(1\text{-}L_2))^{50}. \tag{18}$$

The formula assumes that a proportion c_1 of the habitat is threatened by process 1. A proportion c_2 is threatened by process 2, and so on. If processes 1 and 2 are coincident in space (such as land clearance and salinization), then they should be treated as a single process.

Step 8: Account for processes that permanently reduce the density of populations within their potential habitat.
Some impacts result in more or less permanent reductions in local population density without an ongoing decline in abundance or range. Such processes may not eliminate the taxon from any location, but they could reduce the viability of a taxon at a site. Examples include grazing of livestock or increased disease rates, which result in reduced local population density. For these processes, estimate the proportional reduction in local density, r_j, due to each of the j effects. Then,

$$D_{i,r} = D_i r_j \tag{19}$$

For example, if grazing reduces by 10% the average density of a population within its extent of occurrence and a disease reduces population density by 20%, then *r* for grazing is 0.9 and *r* for disease effects is 0.8, and overall population density will decline to 72%.

Step 9. Estimate the added risk from pest or pathogen
Evaluate whether the effects of the invasive organism generate a deterministic loss of habitat, a permanent reduction in density, or both. Use the equations under step 7 (generating $H_{i,s,d,p}$) and / or step 8 (generating $D_{i,r,p}$) to estimate the expected potential habitat and population densities in each disturbance region, in the presence of the invasive organism. Combine estimates across disturbance regions and estimate the impact of the invasive organism as

$$I = 1 - \frac{\sum_{i=1}^{n} N_{i,p}}{\sum_{i=1}^{n} N_i} \tag{20}$$

where

$$N_i = H_{i,s,d} \times D_{i,r}$$

is the expected number of adult plants in potential habitat at the end of 50 years, without the invasive, and

$$N_{i,p} = H_{i,s,d,p} \times D_{i,r,p}$$

is the expected number of plants in the presence of the invasive.

Step 10. Estimate the 'benchmark' adult population size and evaluate extinction risk
The importance of a reduction in population size may be translated into an

approximation of a risk of extinction by comparing the population estimate to critical values that are symptomatic of species at risk. This comparison is optional, but requires that an analyst select a benchmark population size, a reference point that can be used to make risk assessments on different species comparable. This is the population size (*F*) likely to persist under the influences of demographic and environmental uncertainty, assuming an environment free of the relevant pest(s) or diseases(s). We suggest analysts estimate a the population size that results in an equivalent risk of quasiextinction, specifically a 0.1% probability of falling below 50 adults at least once in 50 years, to provide a background risk against which to measure added risk.

Ideally, the values for *F* should be based on the best available population model, taking into account factors such as the spatial distribution of habitat, dispersal, seed bank dynamics, disturbance response mechanisms, life history, demographics, outbreeding and selfing characteristics, and genetic homogeneity. In the absence of a species-specific model, *F* can be calculated based on a simple birth and death model. In the absence of any model, expert judgement may be sufficient.

To assist analysts to estimate the benchmark population size, *F*, we calculated values of *F* for several taxa based on detailed population viability analyses for individual species and on more generic models reflecting broad life-history traits. These species represent several of the functional groups identified by Noble and Slatyer (1981), including obligate seeders and resprouters, species with short- and long-lived seed banks, and species in which adults are susceptible to disturbance (Table 7 is only a guide).

Table 7. Examples of values of the initial population size, *F*, necessary to achieve a probability of 0.1% of falling below 50 mature individuals at least once in the next 50 years (see Burgman et al. 2001, for details)

Taxon	**Survivorship (annual)**	**Coefficient of variation**			
		0.05	**0.1**	**0.2**	**0.25**
Hypothetical	0	520	1000	23000	60000
	0.2	480	800	17000	50000
	0.9	280	550	9800	40000
	0.98	180	500	1600	38000
Detailed models	**Longevity (years)**				
Banksia goodii	300	300			
Banksia cuneata	50	6400			
Alnus incana	20	750			
Arisaema triphyllum	?	11100			
Pentaclethra macroloba	100	2300			

Values for survival and variation and hence for *F* may be adjusted to reflect the characteristics of biology and life history that are likely to affect background risks of decline. For example, persistent soil-stored seed reduces the probability of extinction of a local population and reduces the value of *F.* Species with poor dispersal abilities may require larger *F* values (Table 8).

Any such modifications can be guided by a simple model that accounts for demographic variation and moderate levels of environmental variation in an unstructured or stage-structured, single population model without density dependence (Table 7). The number *F* may be smaller than the current population size, especially for abundant species. If *F* is less than the current number, this implies that if there are no additional detrimental processes or catastrophes, there may be the loss of some individuals but the species may still have an acceptably low risk of quasiextinction.

Table 8. Ecological factors that affect the initial population size, *F*, required to provide adequate chances of species persistence, assuming no additional sources of disturbance.

Positive circumstances (resilience)	*Negative circumstances (vulnerability)*
many large populations	few small, isolated populations
widespread distribution	restricted distribution
habitat generalist	habitat specialist
not restricted to a temporal niche	restricted to a temporal niche
no extreme habitat fluctuations	subject to extreme habitat fluctuations
no genetic vulnerability	genetic vulnerability
vigorous post-disturbance regeneration	weak post-disturbance regeneration
rapid, vigorous growth	slow, weak growth
quickly achieves site dominance	a poor competitor
all life stages resilient particular	life stages vulnerable
short time to set first seed or propagules	long time to set seed or propagules
long reproductive lifespan	short reproductive lifespan
robust breeding system	dysfunctional breeding system
readily pollinated	not readily pollinated
reliable seed production	extremely variable seed production
high seed production and viability	low seed production and viability
long seed or propagule viability	short seed or propagule viability
propagules not exhausted by disturbance	propagules exhausted by disturbance
good dispersal	poor dispersal
generally survives fire and other damage	generally killed by fire
adapted to existing grazing, drought, fire	not adapted to grazing, drought, fire
able to coppice or resprout	not able to coppice or resprout
not vulnerable to pests and disease	vulnerable to pests and disease
not dependent on a vulnerable mutualist	depends on a vulnerable mutualist

Animals

Wilson et al. (2010) propose a simple model to estimate risks to mammal populations. Parameters needed for the model are listed in Table 9.

Table 9. Definitions of parameters used in the equations that make up the method for estimating pest and disease impacts on plants.

Parameter	Definition
M	An acceptable mean time to extinction
r_i	Intrinsic mean growth rate of the population in the absence of the invasive
r_a	Current mean growth rate of the population in the absence of the invasive
r_p	Estimated mean growth rate of the population in the presence of the invasive
σ	Standard deviation of the growth rate of the population
K_T	Target population size that satisfies M
K_N	Current population size
K_a	Estimated population size in the presence of the invasive

The mean time to extinction of a species can be approximated by the formula (Lande 1993, McCarthy et al. 2005):

$$M = 2K^b/\sigma^2 b^2 \quad . \qquad (21)$$

where K is the population size, r is the growth rate of the population, σ is the standard deviation of the growth rate of the population, and $b = 2r/\sigma^2 - 1$.

The parameter b is typically between 0.5 and 2.5. For example McCarthy et al. (2005) estimate that $b = 0.55$ for a sheep and 0.87 for a glider. Carnivores have higher b values, often over 1. Herbivores and granivores tend to have higher variances in population growth rate and should have lower b values.

Assume that M = 100,000 years is an acceptable mean time to extinction. By rearranging equation (21), we can calculate a target population size K_T for any vertebrate:

$$K_T = (100000\sigma^2 b^2/2)^{1/b}. \qquad (22)$$

In general, data on r and σ^2 are unavailable. Sinclair (1996) found the maximum instantaneous rate of growth of a population of mammals over a year (r_m) to be approximated by a function of the body mass via $r_m = 1.375W^{-0.315}$, where W is the adult live body mass of females in kilograms. Sinclair (1996) also found the instantaneous rate of change between censuses, r_t, to relate to body mass via $r_t = 0.805W^{-0.316}$, with r_t approximated by r_m/T, with T calculated according to $1.74W^{0.27}$ (Miller and Sammuto 1983). These approximations can be used to estimate r_i the *intrinsic* mean growth rate of the population, and σ^2. As a rough guide, typical values are provided in Table 10.

Table 10. Typical values of the intrinsic growth rate r_i, variance in the growth rate, σ^2, and associated values for *b* and the target population size K_T that satisfies a mean time to extinction *M* of 100 0000 years.

Species	r_i	σ^2	***b***	K_T
Typical rodent	0.35	0.40	0.75	252 079
Rodent with high variability in growth	0.35	0.45	0.56	8 221 433
Herbivore	0.20	0.20	1.00	10 000
Carnivore	0.10	0.08	1.50	433
Elephant	0.05	0.04	1.50	273

If an estimate of the current population size K_N is available then a comparison can be made with K_T and an assessment of the risk facing a species made. If K_N is less than K_T the expected time to extinction is less than 100 000 years. This is essentially equivalent to Step 10 of the plant protocol described above. While the comparison between K_N and K_T may be of interest, it does not tell us anything about the magnitude of risk or impact posed by an invasive species.

To estimate impact, *I,* we adapt equation 20 above, so that

$$I = 1 - \frac{K_p}{K_a}, \qquad (23)$$

where K_p is the population size in the presence of the invasive species and K_a is the population size in the absence of the invasive. Both K_p and K_a are calculated using Equation 22, except that *b* is estimated using the growth rates r_p and r_a, respectively. That is, estimates are required of the extent to which an invasive species will depress (or stimulate) growth rates. If we assume that the ratio of *r* to σ^2 is constant, the growth rate in the presence and absence of the invasive is the key parameter requiring information or expert judgment.

We note that the proportional response of the impact index varies from species to species. Figure 4 reports values of *I* for the species listed in Table 10. The analysis assumes $r_a = r_i$ and that the growth rate in the presence of the invasive is reduced by 20% (i.e. $r_p = 0.8r_a$). It also assumes that the ratio of *r* to σ^2 is constant in the presence and absence of the species, so that *b* does not change from figures reported in Table 10.

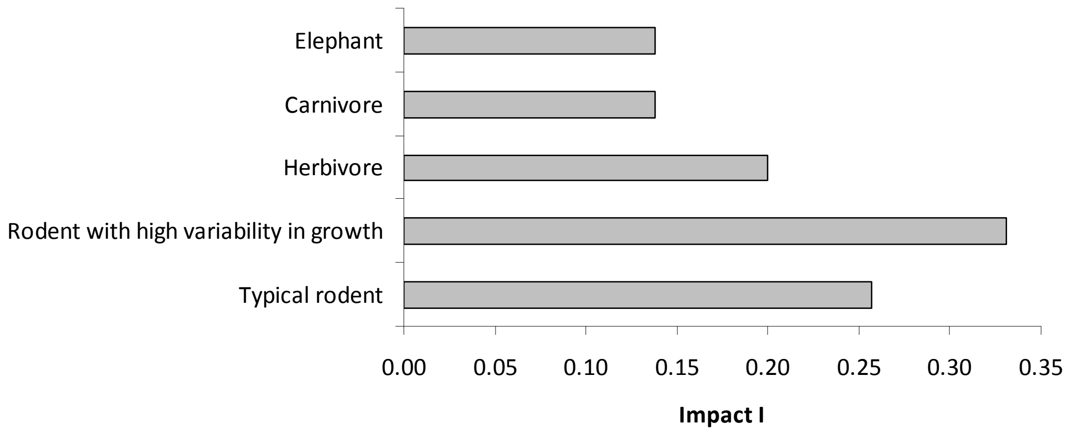

Figure 4. Values of the impact metric *I* calculated for species listed in Table 10 using equation 24.

3.0 INVASION CONSEQUENCES OVER TIME

By calculating the consequences of species invasion using the approach outlined in sub-section 2.1.2 and plotting the likely damage we expect to be inflicted over time, we start to see a picture of that species' impact. Moreover, in plotting the consequence-time relationship and comparing it to other invasive species we can prioritise species according to their capacity to cause damage over designated time periods. In doing so, it becomes apparent that the time frame over which we wish to make this comparison is critical in determining which species are treated as the highest and lowest priority when economic consequences are used as the sole ranking criterion. This section has been adapted from Waage et al. (2005), and describes the types of consequence-time relationships describing different types of species invasions.

3.1 Types of consequence curves over time

In applying the general analytical model of sub-section 2.1.2 across invasive species from different taxonomic groups, Waage et al. (2005) observe that there are three basic consequence-time relationships in relation to pest species. Furthermore, these relationships are dictated by the way in which a pest or disease spreads over time - or, more precisely, the way in which it influences the parameters of the biological model and the economic model. If we assume that a standard time frame over which an impact simulation study is run is 30 years, the following sub-sections describe these three basic consequence-time relationships and the types of invasive species they characterise.

Constant expected consequence increments over time

There is an almost linear relationship between invasion consequence and time for some pests and diseases. That is, between 10, 20 and 30-year time intervals the degree to which expected invasion consequences increase remains relatively constant. Figure 5 provides an abstract view of the flow of expected consequences over time for these pests. Clearly, the change in consequences is identical between years 0-10 (ΔEC_{10}), 10-20 (ΔEC_{20}) and 20-30 (ΔEC_{30}). The resultant expected consequence curve is linear (EC_A).

In general, the invasive species demonstrating consequence curves of this nature tend to be crop pests in that their major economic consequence is associated with their effect on crops of one type or another. In addition to the probabilities of entry and establishment and yield loss, the parameters with the highest sensitivities in quantitative analyses indicating a linear relationship tend to be biological in nature. In particular the intrinsic rate of spread (*r*) and diffusion coefficients (*D*) are highly influential in terms of expected consequences over time (Waage et al. 2005).

A linear EC curve certainly does not imply a linear biological spread pattern. The reason why we observe linear EC curves is largely attributable to the process of *discounting* future consequences. Both *private* and *social* consequences are captured by the EC curve of Figure 5, both of which are discounted, but for different purposes. A discount rate is applied to private contexts to reflect the opportunity cost of investment decisions.

Assume that a farmer is spared $100 worth of crop damage due to the exclusion of an invasive species in one time period. In the following time period, the farmer has the option of putting this additional revenue back into cropping, or to invest it in something else. For instance, she may choose to put the money into stocks, shares or bonds and earn an interest rate of, say 10 per cent per annum. The rate of interest that could be earned by the $100 is an opportunity cost of reinvesting in cropping. So, $100 worth of crop damage prevented in the second time period is only worth $90 in the current time period due to discounting with this private discount rate.

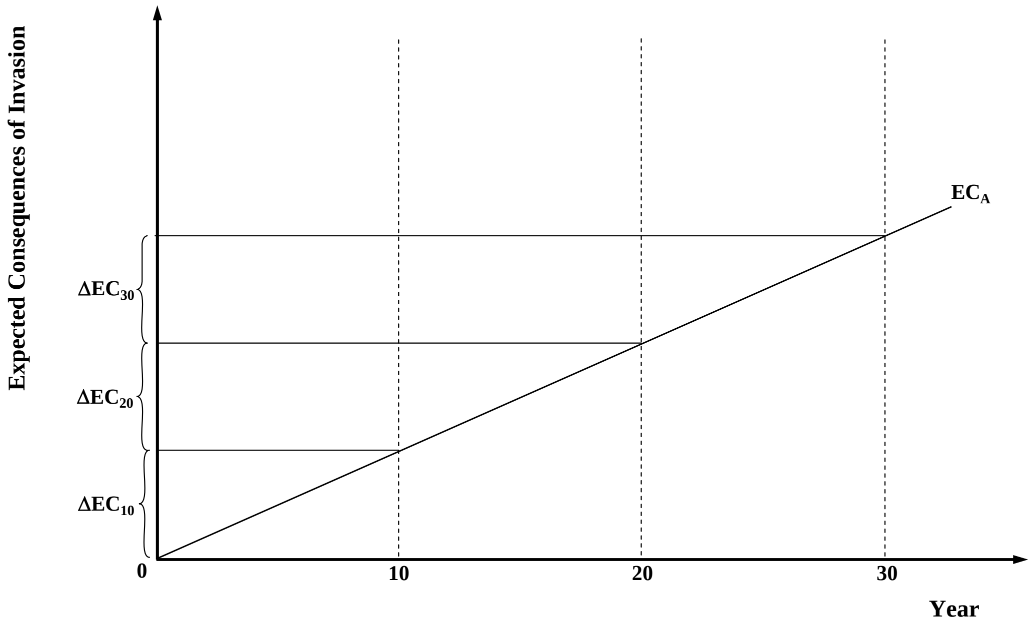

Figure 5. Constant expected consequence increments over time

The social discount rate is harder to define. In the social or government context, discounting reflects the view that future generations will be 'better off' than the current generation. Technological progress is making the production of goods and services cheaper over time, and real incomes are rising. So, $100 to the average person in the society of 20 years time is worth less than $100 is to the average person in society today. It follows that the social benefits of biosecurity policies accruing in the future should be discounted. The problem is how to choose a social discount rate (Waage et al. 2005).

In the absence of clear information on opportunity costs relevant to a specific project (e.g. like the control of an invasive species), economists usually cite government guidelines which recommend a standard discount rate. For instance, in Australia the cited rate consists of a margin of between 1 and 3 per cent on top of a real risk free rate of 5 per cent (Department of Department of Finance 1991). This risk free rate, applying to streams of uncertain benefits adjusted for the cost of risk-bearing to risk-averse individuals, can be revised downwards to reflect a precautionary attitude to radical ecosystem changes (Cook et al. 2007). When applied uniformly to pest and disease consequences any positive discount rate erodes future values, thus affecting investment decisions made over multiple time periods.

Diminishing expected consequence increments over time

Pests and diseases whose damages are far less gradual have distinctly different expected consequence relationship associated with them. In some cases the majority of economic consequences occur relatively early in the invasion process. In terms of the expected consequence curve over a 30-year period, between the 10, 20 and 30-year time intervals expected consequences rise at a steadily decreasing rate. Figure 6 provides an abstract view of the flow of expected consequences over time for these types of pests and diseases. Here, the change in expected consequences between years 0-10 (ΔEC_{10}) is larger than in the 10-20 year interval (ΔEC_{20}), which in turn is larger than the change in expected consequences over the 20-30 year interval (ΔEC_{30}). The resultant expected consequence curve is labelled (EC_B).

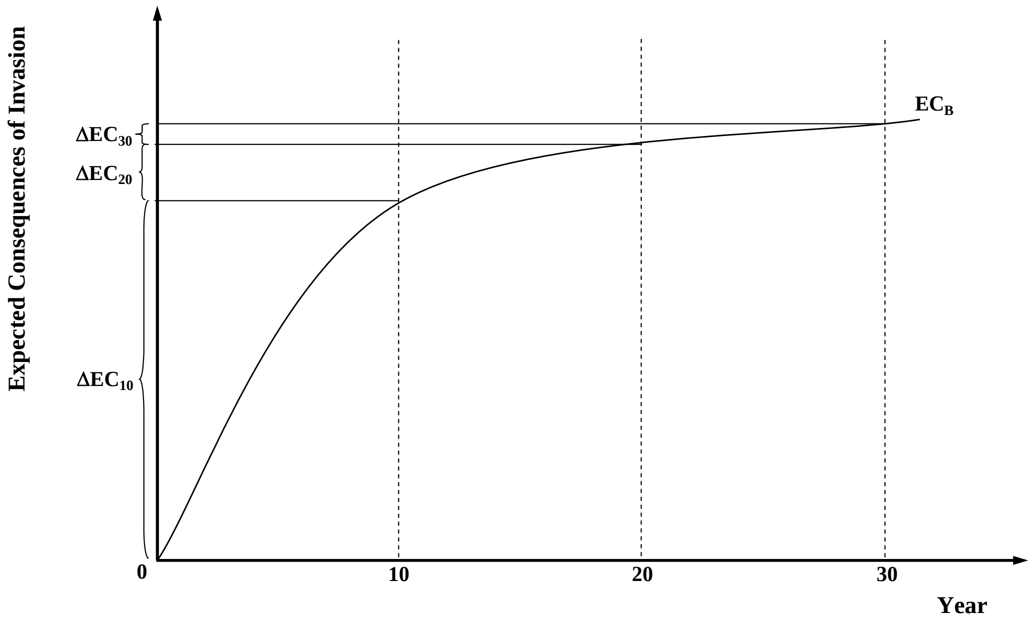

Figure 6. Decreasing expected consequence increments over time.

The variables of the highest significance in these kinds of consequence-time relationships tend to be the probabilities of entry and establishment and the impact of a loss of area freedom on the export market for a host (Waage et al. 2005). If these revenue losses are felt immediately upon detection, the consequences on the economy between the years 0-10 (depending on the assumed probabilities of entry and establishment) are severe. This explains the growth in the expected consequences during this time interval. Thereafter, the impact increases at a slower and slower rate, resulting in an expected consequence curve the shape of EC_B in Figure 6.

Increasing expected consequence increments over time

Non-market value information is seldom used in quantitative studies of invasive species due to problems associated with the collection of reliable data. If this information can be effectively incorporated into empirical analyses it is almost certainly the case that we will see a third type of consequence curve – that of increasing expected consequences over time.

Two factors influence the inter-temporal value of environmental goods:

1. Income elasticity of demand with respect to environmental benefits. The income elasticities for environmental goods are thought to be large and positive, although comprehensive empirical evidence is lacking (Whitby 2000). Two reasons have been put forward as to why this might be the case: (a) there is a tendency for a strategic misrepresentation of preferences when expressing utility derived from environmental goods, and; (b) demand for specific environmental assets (or marginal changes in the health of an environmental asset) tend to be embedded within stated or revealed preferences for much broader environmental issues (Kristrom and Riera 1996; Whitby 2000). Nevertheless, the value of specific environmental goods is expected to increase as incomes rise[10].

2. Supply and demand. As the scientific community and media distribute information about the general state of health of the environment and its components more effectively, a positive influence is exerted on the public demand for environmental benefits. Moreover, unlike agricultural systems, social valuation of an environmental system is likely to be exponentially positively related to the perceived extent of its deterioration.

Figure 7 demonstrates the impact of these two effects conceptually. Consider the case of an invasive species with a negative effect on environmental goods. The change in the expected consequence of such species between years 0-10 (ΔEC_{10}) is less that that occurring between years 10-20 (ΔEC_{20}), which in turn is less than that occurring between years 20-30 (ΔEC_{30}). The slope of the resultant total consequence curve (EC_C) is positive and increasing at an increasing rate. Note that this curve shape implies that the effect of the income elasticity of demand and price rises for environmental goods overrides the effects of discounting.

[10] This is not to say that there is a tendency for people in lower income brackets to be ignorant of the natural environment. On the contrary, lower income groups tend to recognise the individual value of specific environmental assets, and consequently the opportunity cost of investing in the preservation/rehabilitation of one at the expense of others (Kristrom and Riera 1996).

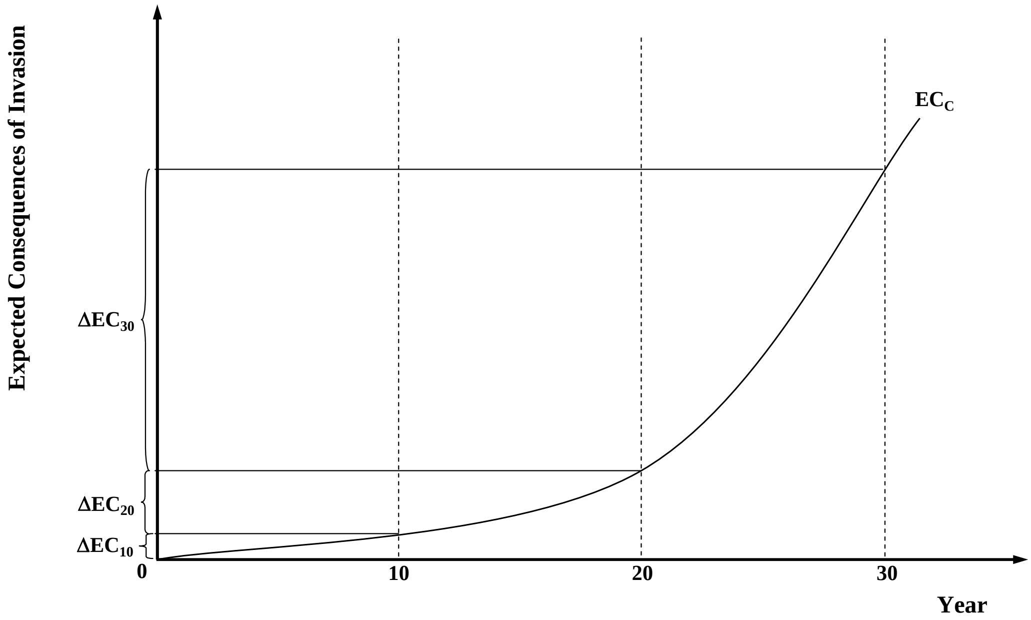

Figure 7. Increasing expected consequence increments over time.

3.2 Cross-over effects and variability

Having postulated three expected consequence curve shapes, it is possible to speculate how a policy-maker might begin to make decisions and compare between invasive species according to the flow of invasion consequences over time. Conceivably, consequence-minimising policy makers may be faced with the situation where the most desirable biosecurity risk management strategy is quite different for different time horizons. It may be that expected consequence curves related to different species cross over one another at some point in time.

Assume, for instance, a policy-maker is concerned with three hypothetical invasive species, A, B and C, characterised by the respective expected consequence curves EC_A, EC_B and EC_C shown in Figure 8. Further, assume they have sufficient information to track the flow of invasion consequences accurately over time, including environmental effects. So, variance around the mean is negligible. The choice of risk management strategy will largely depend on the length of time policy-makers consider relevant.

If an appropriate time horizon for biosecurity policy-making were 10 years, a consequence-minimising strategy would involve the targeting of pest B. The explosive impact of this pest early in the time horizon gives it high biosecurity significance in the short term. By making its exclusion a policy priority, the invasive consequence is minimised. Pests A and B are less significant in the short term, so there appears to be little strategic merit in targeting biosecurity policies towards them over a 10-year time frame. By year 18, the EC_C curve crosses the EC_A curve as the presence of pest C begins to cause economic damage, but by year 20 the benefits of targeting pest B still outweigh those of other invasive species. However, by years 25 to 30 the situation is quite different. The EC_C curve has now crossed over the EC_B curve, and pest C now represents the species of greatest biosecurity significance to the region. Hence the time horizon for policy-making clearly determines the focus of resource allocation.

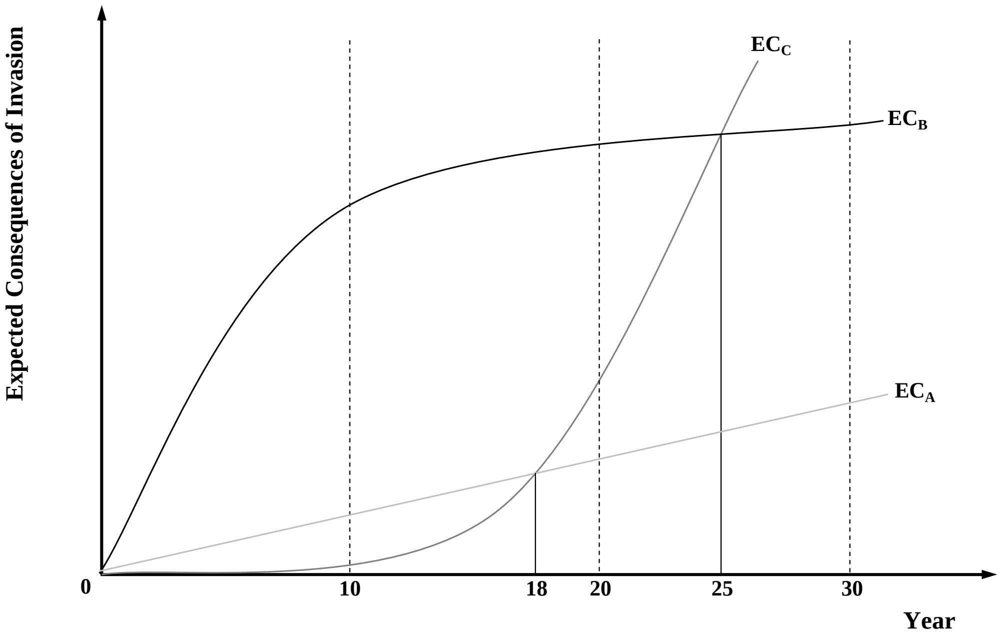

Figure 8. Cross-over effects

If pest A can be thought of as a typical disease of crops, pest B as an environmental invasive and pest C as an animal disease, the optimal risk management strategy begins to look familiar. In the short term, the explosiveness of animal diseases (particularly those singled out by the World Organisation for Animal Health (or OIE) as requiring a ‘stamp out’ policy upon detection) creates a great deal of policy interest. However, over a longer time horizon, the significance of other pests can increase dramatically. A failure to control such pests early effectively passes the burden of control on to future generations. This explains why many environmental pests are often overlooked by biosecurity policies for considerable periods of time (Simberloff 2006). Once they begin to feature prominently in biosecurity risk profiling for specific regions, as in the case of pest C by the 25-year mark of Figure 9, they can be extremely difficult to control and impossible to eradicate.

Information constraints involved in modelling invasive species consequences invariably mean policy-makers will be faced with a much broader range of estimated consequences over time on which to base investment decisions. Hence, the mean is often a poor representation of the distribution of expected invasion consequences over time. Predictive modelling of species introduced to new environments typically involves the use of broadly defined parameter distributions rather than point estimates due to a lack of data.

If we now assume this is true for our hypothetical pests A, B and C, the decision of which to treat as a biosecurity priority becomes more complicated, as Figure 9 shows. If only the mean expected consequences curves are considered the situation has not changed as far as the decision-maker is concerned. Organism B has the highest invasion consequence until year 25 after which it is surpassed by C, and organism A has the second highest impact up until year 18 when it is surpassed by C. But, if the variability of consequence estimates is taken into account, indicated by the broken lines either side of EC_A, EC_B and EC_C, prioritising becomes dependent on the decision-maker's attitude to biosecurity risk and uncertainty (Waage et al. 2005).

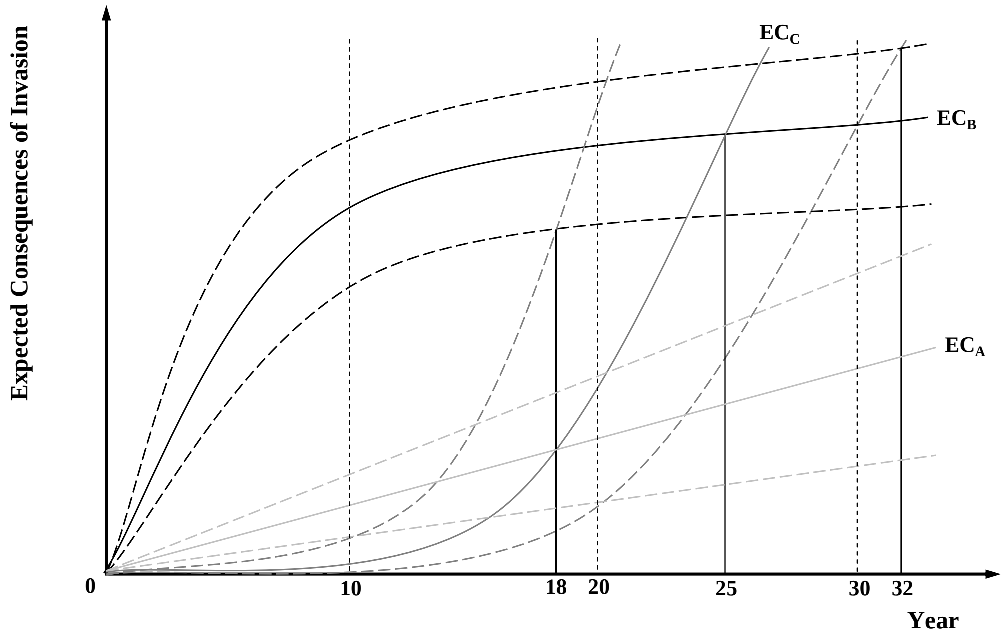

Figure 9. Uncertainty and consequence assessment.

For instance, the impact of organism C may exceed that of B by as early as year 18, so a decision made over a 20-year time horizon by a risk averse policy maker may lead to C being more intensely targeted with biosecurity resources than B. Similarly, the consequence of A may be exceeded by that of C as early as year 10. On the other hand, the cross over between C and B may occur as late as year 32. In light of this information, a policy-maker looking over a 20-year time horizon may choose to target resources towards the exclusion of B at the expense of A and C. Note a decision-maker is indifferent between A and C over 20 years (Waage et al. 2005).

It follows that using the expected invasion consequences as the sole guide for preventative biosecurity investment decisions is problematic. This is particularly true in the case of species that have not been well studied by scientists. Trying to predict the likely economic consequences of a new invasive species is extremely difficult when we face a situation of almost complete ignorance in terms of how to parameterise a spread model and simulate spread and impact over time. If non-market impacts are added into the mix, the problems are exacerbated by difficulties in ascertaining environmental and social values. For these reasons we require a broader set of decision criteria on which to base investment decisions and a credible framework for aggregating these criteria. Sections 4 and 5 of this report outline arguments for using a structured decision-making approach as a sound alternative to traditional benefit-cost analysis.

4.0 ESTIMATING THE IMPORTANCE OF CONSEQUENCES

The task of resource allocation in biosecurity is multi-faceted when agricultural, social and environmental consequences can result from invasion, and outcomes of decision-making impact a variety of stakeholders with different priorities or objectives (Linkov et al. 2004). Economic analyses using a narrow, partial equilibrium model needs to be supplemented with other information. The difficulties involved in quantifying the non-market consequences of invasive pests may lead to their exclusion in economic analyses. However, if policies directed by such analyses are to reflect social preferences, a more formal recognition of potential non-market damage is needed.

Both benefit-cost analysis (BCA) and structured decision-making (SDM) require the estimation of consequences under different alternatives using the sorts of methods explored in Section 2 of this report. The identification of objectives and potential impacts is an important first step irrespective of whether BCA or SDM is subsequently applied. Established conventions in BCA provide rigour in the avoidance of double counting. This rigour is not always evident in multi-criteria decision analyses. The two methods depart in the details of the approach they use to estimate the importance of consequences of different kinds. BCA can be regarded as a special case of SDM when most or all of the consequences are measured in monetary terms. Benefit-cost analysis translates all market and non-market consequences into monetary units. Non-market impacts are typically estimated using revealed (e.g. travel-cost method) or stated (willingness to pay or accept) preference techniques and studies. These techniques can be demanding in terms of the time and expertise required. Some of the controversies associated with BCA arise when some consequences are ignored simply because they are too difficult to value in monetary terms.

Although easily presented conceptually, difficulties remain in monetising non-market values over time. There has certainly been a marked increase in research in environmental valuation since the early 1990s. The catalyst for much of this work came in 1989 when the oil tanker *Exxon Valdez* struck Bligh Reef in Prince William Sound, Alaska, spilling more than 11 million gallons of crude oil. The spill endangered millions of migratory shore birds and waterfowl, as well as many other species such as sea otters, porpoises, sea lions, and several whale species. In response to public outcry over an environmental catastrophe of this magnitude, the number of environmental valuation studies increased dramatically.[11] However, while much of this research has been effective in estimating *use values*, including values associated with morbidity and mortality, recreation values and property value changes, it has been less successful in eliciting non-use values (Adamowicz 2004).

The heterogeneity of preferences based on non-use values, and indeed the heterogeneity of specific sites and marginal changes in their condition present major

[11] The number of publications concerning environmental valuations jumped from less than 10 in 1990 to almost 100 by 1991. By 2003, the number was over 470. Of these, stated preference techniques demonstrated the highest growth in number (Adamowicz 2004).

obstacles for environmental and ecological economists. The use of behavioural economics and stated preference techniques may yield some answers in future by modelling preferences in the context of respondent memory, attitudes and opinions. This is set to become a particularly exciting area of research involving attitudes to environmental sustainability issues, ecosystem and species uniqueness, irreversibilities and irreplaceabilities. But, to date there is no practical means of eliciting non-use values for environmental goods cheaply, rapidly and accurately. 'Benefit transfer' seeks to overcome the bottleneck of non-market valuation by making available databases of preference studies (Bergstrom and de Civita (1999)). Canada hosts an international database that is partly funded by the Australian government (see https://www.evri.ca/Global/Splash.aspx). However the legitimacy of extracting results of a study undertaken in one decision context in one part of the world and using it in another is controversial. Only a subset of techniques used in stated preference studies have been shown to be transportable (Morrison et al. 2002).

In SDM approaches decision-makers express their preferences across the range of anticipated consequences by assigning importance weights. The concept is equivalent to stated preference techniques in that weights reflect the point of indifference between two amounts of two entities (in BCA one of those entities is money) (Keeney 2002). For example, imagine two candidate management actions to address public health concerns associated with a zoonotic disease. The economic cost of Alternative A is $50 million and that of Alternative B is $75 million. The respective number of hospital cases under A and B is estimated to be 200 versus 80. The identification of both costs and human health as objectives will permit decision makers to state which of these two impacts they consider to be most important: a cost difference of $25 million or 120 additional seriously ill people. A variety of different techniques (including the commonly used swing-weighting method; see Clemen, 1996) can be employed as aids in this judgment. If equal weights are assigned, a decision-maker (in the circumstance where she had to make a choice) is indifferent to an additional 120 ill people and a cost of $25 million. If she assigns health outcomes twice the weight of cost, her point of indifference (assuming linearity) is 60 ill people and a cost of $25 million.

This illustrative situation is clearly over-simplified. Further information needs to be provided about how these objectives were estimated, and in most cases other objectives (relating to environmental, social, or other concerns) also will be important. Yet even this simple example makes clear two important aspects of how objectives are used in decision making. At the early stages of a resource management process, objectives define what are considered to be relevant consequences. Resources are then allocated toward finding management alternatives to address these objectives, and analytical tools for assessing consequences on them. Later, objectives clarify the difficult value judgments across seemingly incommensurate dimensions of the problem. This makes for a more robust decision making framework and it helps to promote dialogue and understanding among decision-makers and stakeholders and, ultimately, assists in the choice of a broadly acceptable management alternative.

5.0 DISCUSSION

Data and resource needs

The suggested methods presented in Section 2 for estimating impacts on primary production, social amenity and the environment represent the authors' views on an appropriate trade-off between rigour and the demands for data and expertise. Certainly the suggested methods are more detailed and demanding than many of the protocols used currently in biosecurity decision-support. But we believe that the weaknesses of current protocols emphasise the need for greater rigour.

We advocate the more information intensive general equilibrium approach to modelling impacts on primary production only in circumstances where a sector (e.g. wheat production) with substantial linkages to the rest of the economy is likely to be affected. Apart from data demands, expertise is required in judicious use of input-output analyses and multipliers. But for lesser impacts on smaller sectors, micro-economic theory is accessible and the judgments required for parameterising a simple partial equilibrium model are modest.

For social impacts that are inadequately captured by natural attributes or proxies, we advocate constructed scales because of their flexibility and intuitive appeal. We warn that the apparent simplicity of capturing social impacts using constructed scales may be deceptive. Considerable experience and expertise is required. We recommend a small team within Biosecurity Services Group be established to develop specific scales for specific decision contexts. The team might usefully include one or two social scientists and one or two analysts from pre-border and post-border settings. Before tackling specific problems, the team will need to acquaint themselves with details of different types of constructed scales (see Keeney 1992).

The parameters required to estimate environmental impacts using the approach we advocate are described in Tables 6 and 9. Many of these parameters for many species are available in the published literature. Expert judgment will be required at other times. Much of this ecological and taxonomic expertise is already available in DAFF and its immediate stakeholders and affiliates. The computations involved are simple, requiring no more than basic numeracy and use of an excel spreadsheet. These data and computational requirements are far less onerous than state-of-the-art modelling in population viability analysis (Coulson et al. 2001).

Representing uncertainty in attributes

Reporting consequences will in some cases require exposing differences in the *range* of possible outcomes associated with different actions or management alternatives. For example, consider two alternative pest control actions: Action A is expected to contain any losses suffered by farmers associated with a potential pest invasion to about 80 square kilometres, whereas the best estimate for Action B is an affected area of 150 square kilometres. Clearly, with other things equal, Action A would be preferred. However, assume there is uncertainty in the estimates of potential losses: although the best estimates of experts suggest that Action A would outperform Action B, under some plausible conditions Action A could lead to widespread losses across several states (so that the upper bound of the experts' estimates shows losses to 650 square kilometres) whereas the worst plausible performance for Action B would involve losses to 250 square kilometres. In this case, the alternatives involve different degrees of risk, and different decision makers might make different choices depending on their risk tolerance. When this is the case, it's important for attributes to provide concise information on the distribution of possible outcomes. Uncertainty also can be easily expressed using ranges to estimate the anticipated effects on constructed scales; an attribute showing the expected public support for two different management actions could be expressed in terms of the best estimate accompanies by a plausible range.

Presenting information on expected distributions of effects (best estimates and ranges) is only one of several ways to expose the implications of uncertainty in impact estimates (for more discussion, see Burgman 2005 or Walshe and Burgman 2010). The preferred method will depend on both the nature of the problem and the intended audience. However, because some decisions hinge almost entirely on such risk-risk trade-offs, in such cases it is important to avoid the use of deterministic measures or simple averages that fail to reveal the underlying uncertainty associated with the decision attributes.

Benefit-cost analysis and structured decision-making

In an SDM process, consequences are expressed using a variety of metrics. This is in contrast to a conventional cost-benefit analysis, which translates diverse impacts into monetary measures. The primary advantages of an SDM approach are:

- The magnitude and significance of the actual impact can be more readily understood. Affected habitat can be expressed in terms of square kilometers of area, changes in recreation opportunities can be expressed in terms of visitor days, costs can be expressed in terms of dollars, and less tangible concerns such as a lack of trust in a management agency or a fear that a culling program will get out of hand can also be included through the development of context-specific indices or scales.
- The attributes can be used to help identify useful management actions that may reduce adverse consequences.

- The values of decision makers (or other participants in the process) play a central role in both identifying and evaluating alternatives.
- Controversial monetization techniques are avoided.

The primary disadvantage most often cited is that decision makers will need to deal with complex trade-offs – balancing ‘apples and oranges’ and other concerns expressed in different units. However, one can argue that these apples and oranges are what real world decisions are all about, and it is arguably the job of decision makers – not technical analysts – to make these tough judgments. Of course SDM and CBA are not mutually exclusive, and those consequences that are readily amenable to monetization are often used in an SDM analysis. But in our view the vast majority of decisions benefit from careful characterization of impacts in non-monetary units.

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