Australian Government
Department of the Prime Minister and Cabinet
Office of Best Practice Regulation

Research Report: Environmental valuation and uncertainty

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Scope and purpose

This research report has been produced by the Office of Best Practice Regulation (OBPR), a division of the Department of the Prime Minister and Cabinet, in consultation with the Department of Environment. It was prompted by a recognised need to improve the quality of guidance material relating to environmental valuation in regulatory impact analysis, and through discussions with community organisations.

Its aim is to enhance the capacity of Australian Government departments and agencies to undertake cost-benefit analysis of policies that are likely to have an environmental impact, or that are characterised by significant uncertainty. The particular emphasis in this guide is on the preparation of Regulation Impact Statements (RIS). However, the relatively general nature of the document means it is likely to be of use to a broader audience, including policy officers in other jurisdictions working on initiatives that require environmental valuation.

This guide should not be seen as introducing new requirements, or new expectations of what the Office of Best Practice Regulation considers an ‘adequate’ level of analysis in a RIS. The intention of the agencies involved in the production of this guide is to explain clearly the state of the art in incorporating environmental valuation into impact analyses, in order to better meet the Government’s existing best practice regulation requirements as set out in *The Australian Government Guide to Regulation* (Commonwealth of Australia 2014a) and the Council of Australian Governments’ *Best Practice Regulation: a Guide for Ministerial Councils and National Standard Setting Bodies*. In this way, it is expected that this guide will support the continual improvement in regulatory impact analysis, particularly for those proposals with significant environmental impacts.

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1. Introduction

Just like other assets, environmental assets are valued by the community, and improve our quality of life. They do this in tangible ways, such as by providing clean water or green space to relax in. And environmental assets improve the quality of our lives in less tangible ways – even just the knowledge that certain species or wilderness areas exist can be a benefit.

As such, any analysis of a project or policy that affects environmental assets should take these values into account. But unlike many other assets environmental assets can be difficult to account for in project and policy analyses for a number of reasons, including:

- the benefits provided by some environmental assets can be hard to understand, and harder still to quantify; and
- because our scientific knowledge of many environmental processes is incomplete, environmental impacts can be hard to know with certainty.

These difficulties can fundamentally affect the analysis of important public policy issues, illustrated by the following examples.

- Public policy around the use of natural resources involves the analysis of complex biological systems, and consideration of the impacts of change on multiple users. The development of the Murray Darling Basin Plan, for example, involved complex analysis of the impact of changed water flows on a large number of environmental assets, and the effect that this is likely to have on a number of groups: agricultural producers and communities, urban water users, and those who value the continued existence of a number of ecosystems. Other emerging issues include the development of coal seam gas resources and alternative energy sources.
- Scientific advice indicates that the global climate will be warmer and more variable in the future, regardless of greenhouse gas mitigation policies (PC 2012). The Australian community will need to adapt to these changes in many ways, and government policy-making will need to reflect the likelihood of climate change. But there is uncertainty about what the future climate will be – we can be confident about some aspects of future climate (for example, temperatures and sea levels will continue to rise), but less confident about others (for example, changes in rainfall on the east coast of Australia). The application of a number of tools to support decision-making under uncertainty is growing in importance.

Because environmental values can be difficult to estimate – especially compared with the costs of policies – analysis of policies with environmental aspects risk being unbalanced; either too much emphasis can be given to non-environmental factors, or the evidence used to examine environmental impacts may be overly subjective.

Much work has been undertaken around the world – both in academia and government – examining how the community values the environment, and how uncertainty affects the decisions we make. The purpose of this guide is to gather this information together in a way that provides a framework to analyse how decisions can affect the environmental assets, and the benefits they provide to the community. This framework, while broadly applicable, has the primary goal of informing the development of regulation impact statements (RIS). In this way, decision makers can more easily take account of *all* of the impacts of their decisions, and allow better decisions to be made. An analytical framework that incorporates economic, social and environmental impacts is also an important means of ensuring that public policies are sustainable.

The rest of this guide is set out as follows:

- Part 2 discusses how to identify environmental changes in a way that allows their value to be examined;
- Parts 3, 4 and 5 provides advice on working out how, and how much, the community values environmental changes, and provides some empirical estimates of these values;
- Part 6 looks at how uncertainty can be incorporated into analyses, and how it affects the decisions that can be made; and
- Part 7 brings all of this together for those preparing a RIS.

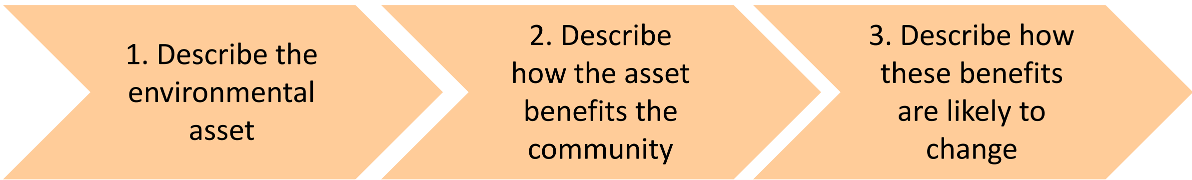
Figure 1. How this guide helps you prepare a RIS

RIS question	How this guidance can help
What is the Policy problem you are trying to solve? What policy option are you considering?	• Describe: ▪ the environmental asset – **page 6** ▪ how the community values the environmental asset – **page 7** ▪ how the environmental asset is changing over time – **page 9**
Why is government action needed?	• Refer to community values when describing the objective – **page 7**
What is the likely benefit of each option?	• Describe the likely changes to the environmental asset as a result of each option – **page 9** • Who wins, who loses? – **page 8** • Estimate the value to the community of these changes – **parts 3, 4 and 5** • Discount future values – **page 35** • Summarise the impacts – **page 34** • Report on any uncertainties – **part 6**
Who will you consult and how will you consult them?	• Use the analysis of how the community values the environmental asset to inform consultation processes – **page 7**
What is the best option?	• Ensure any uncertainties in the impact analysis are realistically reflected in the conclusion – **part 6**
How will you implement and evaluate your chosen option?	• Use identified environmental endpoints to monitor the effects of the changes – **page 8**

2. Identifying environmental impacts

In order to understand how changes to an environmental asset[1] affect the welfare[2] of the community, you must first describe how the community values that asset, and how the changes will affect those values. This process is depicted in Figure 2, and described in detail below.

Figure 2. Identifying the impacts of an environmental change



2.1 Describe the environmental asset

The first step involves a description of the environmental asset in question, in qualitative and quantitative terms. What is its scale, and what are its physical and environmental characteristics? If the asset you are considering is, say, an area of remnant native vegetation, you may want to describe:

- its size,
- the main vegetation types in the area;
- the condition of the native vegetation;
- its location in relation to other areas of native vegetation;
- the species that live in the area, highlighting those that are threatened or endangered;
- the ways in which the (human) community use the area in its current form; and
- the land use patterns in the vicinity of the area.

This process will involve reviews of scientific literature and discussions with experts in the field. In determining the appropriate boundaries of the environmental asset in question, you may need to consider the underlying physical and biological process; how 'connected' is the area in question to other environmental assets, and do they need to be considered together?

You will also need to consider any 'gaps' or uncertainties in our knowledge of the environmental asset, and whether these are likely to be important to the analysis.

The basic idea here is to gain a good understanding of the asset in question and, by considering how it links in with other parts of the natural and man-made environment, determine the appropriate scope of the analysis of environmental impacts.

[1] This report uses the term "environmental asset" throughout. By this we mean the part of the physical environment that is the subject of the analysis - from atmosphere to wetland or watercourse to a threatened species. It is taken to be a 'stock' concept, from which environmental goods and services flow.

[2] "welfare" describes the general wellbeing of an individual or group. Welfare economics is the study of how the allocation of resources, and of goods and services, in an economy affects welfare.

2.2 Describe how the asset benefits the community

The next step is to consider how the community – local, national and potentially international – values the environmental asset. In other words, how the existence of the asset makes the community better (or worse) off.

At this point, it is worth making the observation that there are a number of alternative views on the value of an environmental asset. These range from the view that the intrinsic ecological values of the asset are most important, through to the view that only those aspects of the environment that have a financial value should be taken into account. In establishing a framework for assessing environmental values, this guide adopts an approach that focuses on the value of the environment in attaining *human* goals, be they spiritual, cultural or economic (Pascual et al 2010).

Consistent with this approach, the environment can be viewed as producing a range of 'goods and services' which are in turn valued by people. In identifying how changes in a particular environmental asset will impact on community wellbeing therefore, it is important to draw out the nature of the goods and services provided by that particular asset.

An *ecosystem services* framework is a useful way of thinking about these goods and services. Ecosystem services are the conditions and processes through which ecosystems, and the species that make them up, sustain and fulfil human life (Daily 1997).

The Millennium Ecosystem Assessment (MA 2003) – (figure 3) categorises the various goods and services provided by ecosystems as:

- Provisioning services – the products obtained from ecosystems;
- Regulating services – the benefits obtained from the regulation of ecosystem processes. Regulating services of ecosystems can be final – that is, directly consumed by people – or intermediate service (Kumar 2010);
- Cultural services – the non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences. These values are an important driver of tourism; and
- Supporting services – those that are necessary for the production of all other ecosystem services. They differ from other ecosystem services in that their impacts on people are either indirect, or occur over a very long time.

Figure 3. Examples of ecosystem services

Provisioning services
Products obtained from ecosystems

- Food
- Fibre
- Genetic resources
- Bio-chemicals, natural medicines, etc.
- Ornamental resources
- Fresh water
- Mineral resources

Regulating services
Benefits obtained from regulation of ecosystem services

- Air quality regulation
- Climate regulation
- Water regulation
- Erosion regulation
- Disease regulation
- Pest regulation
- Pollination
- Flood mitigation

Cultural services
Non-material benefits obtained from ecosystems

- Cultural diversity
- Spiritual and religious values
- Recreation and ecotourism
- Aesthetic values
- Knowledge systems
- Educational values

Supporting services
Services necessary for the production of all other ecosystem services

- Soil formation
- Nutrient cycling
- Primary production

Source: MA 2003

In order to relate an ecosystems services framework to economic valuation, we need to define a further term –'ecological endpoint' (Boyd and Krupnick 2009). Ecological endpoints are the environmental goods and services that are *directly* valued by individuals.

This concept can be illustrated by way of example: imagine a wetland system that is polluted by agricultural runoff. Excess nutrients in the water may cause an overgrowth of algae when the algal cells die and decompose; much of the oxygen in the water is used up, which can lead to fish kills and other negative effects. This process is called 'eutrophication'. The overgrowth of algae can also smother wetland plants by blocking access to sunlight, leading to further habitat degradation. In such a case, regulated restrictions on the use of fertilisers on surrounding farmland would reduce nutrient loads and help prevent algal overgrowth in the wetlands, resulting in higher dissolved oxygen levels in the water, reduced turbidity and greater sunlight penetration through the water column.

But it is not likely that people will value these impacts directly – an individuals' knowledge of the biophysical processes involved is probably not good enough to allow them to confidently assign value to the impacts. Instead, higher oxygen levels and reduced turbidity are inputs to further ecological processes which produce outputs – ecological endpoints – that are valued directly. In this case, such outputs could include:

- greater species abundance in the wetland;
- greater numbers of large fish species available for angling;
- better water quality, both for recreation and drinking purposes; and
- a more aesthetically pleasing view.

Similarly, an improved waste management system is not valued directly by the community. It is how the improvements flow on to endpoints such as visual amenity (less litter) and clearer, safer water (less runoff from landfills) that are directly valued that determines the benefits of improving waste management processes. Table 2.1 highlights some other examples of ecological endpoints.

Table 2.1. The distinction between endpoints and outputs

Biophysical output	Further biophysical processes	Ecological endpoint
Dissolved oxygen levels	Habitat and toxicity effects	Availability of larger fish species
Acres of habitat	Foraging, reproduction, migration	Species abundance
Urban forest acres	Shading and sequestration	Air quality and temperature
Vegetated riparian border	Erosion processes	Sediment loadings to reservoirs
Waste to landfill	Chemical leaching to water table	Drinking water quality
Air pollution	Inhalation of toxins, particulates	Human health outcomes

Source: Boyd and Krupnick 2009; Collins 2011

It is important to identify ecological endpoints in impact analyses for a couple of reasons. Firstly, it makes it possible to quantify the economic valuation of the environmental impacts. In some cases (particularly when using stated preference methodologies, which are described later in this guide), meaningful economic valuations can only be obtained when individuals know what it is that they are valuing. And secondly, when you describe the impacts of a policy in terms of ecological endpoints it better conveys the impacts to readers and, potentially, will allow better policy decisions to be made as a result.

2.3 Describe how these benefits are likely to change

The third step in describing the environmental impacts of an environmental change is to determine the likely effect of a policy change on the identified environmental endpoints, compared with a business-as-usual 'baseline'.

Determining an environmental baseline for the asset in question is necessary as it is against this that the impacts of a policy change will be assessed. Starting with the description of the current state of the environmental asset (as discussed earlier) the baseline describes what is likely to happen to the asset – and the identified endpoints – under the existing policy arrangements. It is important to note that the baseline is not just the existing state of the asset, but also expectations about the likely state of the asset, and endpoints, into the future.

Once a baseline scenario has been established, alternative scenarios, representing alternative policy settings, need to be identified. Essentially, the alternative scenarios examine the impact on the identified endpoints of changing important assumptions about the future. Importantly, developing alternative scenarios will depend not only on policy changes and ecological responses, but also on how individuals will respond to the policy changes. In the wetlands example introduced earlier, the alternative scenario would examine the impact on species abundance, water quality etc. of regulated restrictions on fertiliser use. This would involve consideration of:

- how farmers would react to new restrictions (by how much will they reduce fertiliser use? Will use of other chemicals increase? Will land use patterns change?);
- how farmers' reactions affect pollution levels in the wetlands; and
- how changed pollution levels affect biological processes in the wetland, and what this means for the identified endpoints.

After you have established the alternative scenarios, you can determine the impact of the policy change by comparing the level of the endpoints under the baseline scenario with those under the alternative. To enable valuation of the environmental impacts, however, it is necessary to quantify in some way the level of endpoints in the baseline.

Quantification of these endpoints can be a complex process. At a minimum, it will involve a review of the available scientific literature, and discussions with experts in the area. Depending on the size of the study, it may also involve ecological modelling of the baseline and alternative scenarios.

Unfortunately, the gaps in our understanding of the science relating to ecosystem services and the 'production' of endpoints has been identified as one of the most serious problems facing environmental valuation efforts (Bateman et al 2011). This is illustrated by the in-depth assessment of the ecological and economic benefits of increasing water flows in the Murray-Darling Basin, undertaken as part of the development of the Murray Darling Basin Plan (CSIRO 2012). Although the study was able to make use of a number of models looking at environmental responses to water flow scenarios at various locations in the Basin, the state of knowledge was not adequate to draw basin-wide conclusions.

Even if it is not possible to estimate the changes in environmental endpoint quantitatively, you will still need to provide a *qualitative* assessment of the likely impacts under different scenarios. The level and depth of the assessment should be commensurate with the likely impacts of the changes but should include, at a minimum:

- the likely direction of any change in endpoints; and
- a summary of the reasons for this assessment.

If possible, you should also provide an qualitative assessment of how the size of the changes will compare between scenarios: which option could be expected to deliver the largest or smallest changes in endpoints, and why?

The assessment also needs to reflect any knowledge gaps or uncertainties in the analysis. While this is true whether you are making a qualitative or quantitative assessment, it is also the case that the use of quantitative assessments without identification any of the uncertainties involved in their estimation may create the impression of false accuracy. As result, it is particularly important to be clear about any assumptions or caveats underlying such estimates.

3. Valuing environmental changes

Once you have estimated the physical impacts of a policy change on the relevant endpoints – whether qualitatively or quantitatively – the next thing you need to consider is how the community values those endpoints, and how this valuation changes as the endpoints change. This involves considering the ways in which the community values the environment, and how much they value it.

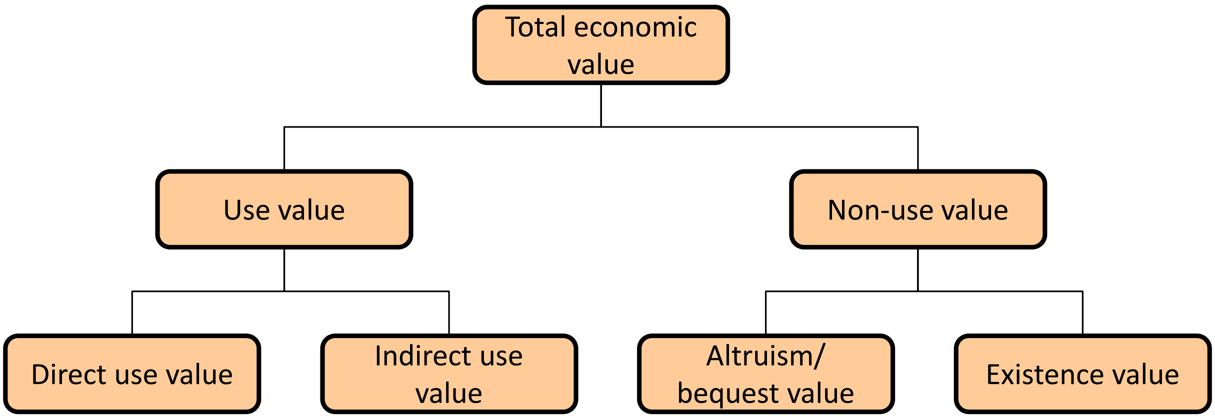
3.1 People value environmental goods and services

There is a distinction between the *value* of a good or service, and its financial *price*. You can experience the difference between value and price when you go for a walk in a park on a pleasant day (Bateman et al 2011). The price of the walk is likely to be zero: there are no entry fees and anybody can just walk in. But the fact that you are dedicating some of your valuable time to the walk shows that it has value. In regulatory impact analysis, we are interested in economic value, not just price.

Like all goods and services, environmental endpoints can be valued in different ways. The sum of these values is referred to as the total economic value; this is described in figure 4. In broad terms, the total economic value (TEV, Pascual et al 2010) of something is the sum of its:

- *use values* – those values associated with tangible things or conditions; and its
- *non-use values* - those values generally associated with experiences that occur in the valuer's mind.

Figure 4. Total economic value[3]



Source: derived from Pascual et al 2010

These values can be further disaggregated. Use values can be either:

- *direct use values* – resulting from the direct human use of the environment. These uses can be consumptive, such as via crops, mining, livestock or

[3] In addition to use- and non-use values, a third class of values – option values – are often identified. Option values are discussed later in this paper in the section dealing with uncertainty.

fishing; or non-consumptive, such as recreational use of the environment, or spiritual/cultural uses.

- *indirect use values* – these are the values that people hold for the regulation services provided by species and ecosystems. Specific examples include pest control, water purification and soil fertility.

And non-use values can be:

- *altruism/bequest value* – this is the value that an individual attaches to the fact that others (whether in this generation or future generations) will be able to benefit from the environment.
- *existence value* – this is satisfaction gained by the knowledge that an environmental asset exists.

Putting together the concepts of ecosystem services/endpoints and total economic value, it is possible to describe in some detail how people – the community – value a particular environmental asset; ecosystem services (and goods) flow from the asset, and these goods and services are in turn ascribed a value by the community. Box 1 illustrates this with an example.

Box 1. Translating ecosystem services into economic values

Consider a State Forest that is sustainably logged. It is accessible by the public for bushwalking and other recreational uses, and provides a habitat for a number of threatened species. The following table describes the ecosystem services flowing from this environmental asset, and the way in which the community values these services under the total economic value framework.

Ecosystem service	Type of economic value
Provisioning	
Timber production	Direct use value (consumptive)
Regulating	
Carbon sequestration	Indirect use value*
Aquifer recharge	Indirect use value
Cultural	
Aesthetic (pretty views)	Direct use value (non-consumptive)
‘Spiritual’ (species existence)	Existence value
Recreation (bushwalking)	Direct use value (non-consumptive)
Supporting	
Range of supporting services	Not directly valued

* Although in the presence of carbon markets or related mechansims it could be argued that carbon sequestration would be directly valued.

3.2 The size of these values can be estimated

In basic terms, a cost-benefit analysis (CBA) measures the impact of a project or regulatory decision on economic welfare. The impacts of the project or decision on all stakeholders are calculated and compared to examine whether the benefits of the proposal outweigh the costs. If the benefits do outweigh costs, the proposal is said to improve community welfare.

In CBA, the value of an impact is equal to the community's 'willingness to pay' for the impact. Willingness to pay is the maximum dollar amount that a person would be willing to forego in order to obtain a benefit, or avoid a cost; summed across all people this becomes the community's willingness to pay.

Some impacts are easier to put a dollar value on than others. The willingness to pay for goods or services that are traded in markets is relatively straightforward to determine – the market price provides an estimate of the willingness to pay for an additional unit of the good or service.

For other goods or services, however, the market may not function fully, or there may be no explicit market, and a price for the service is harder to establish. In such cases it may be necessary to estimate the willingness to pay for these non-market goods indirectly. Many environmental 'goods and services' fall into this category

In the last thirty years, a significant body of economic research has been developed looking at the question of how to value non-market goods and services, and a number of increasingly sophisticated methodologies have emerged. These methodologies take two broad forms: *revealed preference* methods, and *stated preference* methods.

The aim of this part of the guide is to provide an introduction to the main methods – for more detailed information on some of the issues involved in employing these techniques to estimate non-market values, see Boardman et al (2006), Morrison (2009), or Pearce et al (2006).

3.2.1 Revealed preference methodologies

Revealed preference studies seek to elicit peoples' willingness to pay for a good or service by observing their actual behaviour in real, related markets. In this way, for example, the willingness to pay for pretty scenery in an area may be estimated by observing the price of real estate in that area compared with other areas, or the amount of money that people pay to visit that area.

Revealed preference methodologies have the advantage that they are based on actual, observed behaviour. This means that consumers in these markets are taking into account all relevant alternatives to spend their money on, as well as the amount of money that they actually have to spend. However, it can be quite difficult to find relevant alternative markets to examine, and to break out the effect of the willingness to pay for the non-market good in question from the observed market prices.

The major revealed preference methods relevant to environmental values are hedonic pricing analysis and the travel cost method.

Hedonic pricing

Hedonic pricing starts with the notion that the price paid for a good (or service) is really a price paid for a bundle of that good's attributes, and that statistical techniques can be used to identify the implicit prices paid for those attributes. For example, the price of a car may reflect its size, fuel efficiency and safety, while that of a washing machine its reliability, water efficiency and variety of programs (Pearce et al 2006).

For non-market goods, the price may be able to be estimated by looking at a related market. For environmental goods, real estate markets are commonly used[4]. So, for example, the amenity value of a lake is not directly bought or sold, but its value may be reflected in the price of houses in the area. By comparing (statistically) the price of houses in the area with otherwise similar houses in another area, an estimate of the value of the lake's amenity to residents can be derived.

Hedonic pricing is not without its difficulties. How well a price in the observed market reflects the good's attributes depends on how well informed consumers are, and the level of competition in the market. And sometimes it can be difficult to 'tease out' the values for individual attributes. For example, if lake views are usually accompanied by a quieter neighbourhood, it can be difficult to work out whether people are paying for the views or the peace (Pearce et al 2006). Nonetheless, hedonic pricing is an important means of eliciting the willingness to pay for non-market goods and services.

Travel cost

A second revealed preference methodology is the travel cost method, and is often used to estimate the recreation value of an environmental asset. It is based on the assumption that people will spend money travelling to and staying in an area only if the value that they get from these activities outweighs the associated cost. The higher the recreational value of an area, the more people will be willing to pay to be there. In this way, researchers can survey recreational visitors to determine the distances travelled and costs incurred to visit the area, and from this information estimate people's willingness to pay for recreation.

In such studies, the design of the survey is important in isolating the willingness to pay for the non-market good. The possibility of multi-purpose visits (for example, visitors to a national park may also spend part of their time visiting a local town), for instance, mean that the visitors need to be asked what proportion of their time they spent at the site in question, and adjusting the willingness to pay estimates accordingly (Pearce et al 2006).

[4] See, for example, Gibbons et al (2011) who used a hedonic price approach to estimate the amenity value of English nature.

3.2.2 Stated preference methodologies

The second main category of non-market valuation techniques are those that are based on the stated preferences of individuals (Morrision 2009). In broad terms, they involve directly surveying people's hypothetical behaviour in carefully constructed markets for the environmental good/service in question. Because they rely on surveys rather than on observed behaviour, they can be used to estimate a range of non-use values.

The two major types of state preference techniques are contingent valuation, and choice modelling.

Contingent valuation

Contingent valuation is the more straightforward of the two stated preference methodologies. In essence, it presents survey respondents with a hypothetical change (say, the extension of a national park, or the improvement in water quality in a stretch of waterway), and asks them to provide their willingness to pay for this change.

In practice, there are a large number of factors that need to be considered in the design and analysis of the surveys. Even very simple questions require careful consideration of appropriate wording, format and content if they are to elicit accurate information (Pearce et al 2006).

It is also important to be able to present a very clear policy outcome for valuation by survey respondents. Respondents may not have the time or the information available to fully understand the way that environmental changes translate into their own welfare, and therefore into willingness to pay. For example, if someone were asked to provide a willingness to pay for a healthier wetland, they may be uncertain about how this translates into the endpoints that they value – greater species abundance, clearer water, etc. As a result, they would be unsure about their willingness to pay, and the survey results would be less reliable.

This emphasises the importance of considering the ways in which the community values environmental changes, and clearly defining these environmental endpoints before surveying individuals on how much they value them.

Choice modelling

The second set of stated preference approaches are grouped under the term choice modelling. Like contingent valuation, participants in a choice modelling study are asked about their willingness to pay for a certain outcome. In contrast to contingent valuation however, where respondents are asked to value a single scenario, choice modelling respondents are asked to value multiple outcomes (Morrison and Hatton-MacDonald 2009).

Specifically, respondents to a choice modelling survey are presented with various alternative descriptions of a good, differentiated by their attributes and levels, and are asked to rank the various alternatives, to rate them or to choose their most preferred. By including price/cost as one of the attributes of the good, willingness to pay can be estimated from people's choices (Pearce et al 2006)

Table 3.1 describes an illustrative choice modelling question about reducing the amount of sewage overflows that end up in the local lake.

The various outcomes are translated into a set of environmental endpoints (reduced litter, reduced health risk, reduced fish kills), and an amount of money that the option would cost the respondent. By carefully changing the levels of the endpoints in each option, and the amounts of money the respondent is (hypothetically) asked to pay, statistical analysis of the results can determine the community's willingness to pay for the range of non-market endpoints.

Again, the information provided to the respondents, and the way the questions are framed, is very important. In the example, a respondent's willingness to pay for, say, reduced health risk depends on their understanding of what the health risk means: are we talking about itchy skin or certain death? And how much of an increased health risk are we talking about?

Table 3.1. Illustrative choice experiment question

"WHICH OPTION FOR REDUCING SEWAGE OVERFLOWS INTO THE TOWN LAKE WOULD YOU PREFER, GIVEN THE OPTIONS BELOW?"

Endpoint	Current situation	Option A	Option B
Amount of sewage items visible	Some items visible (10% of total litter)	Items almost never visible (1% of total litter	Not present (0% of total litter
Other litter	Present	No change	No change
Water sports health risk	120 days per year increased health risk	4 days per year increased health risk	0 days per year increased health risk
Fish population	8 large fish kills per year	1 fish kills per year	0 fish kill per year
Annual increase in rates	$0 per year	$15 per year	$36 per year

Source: Adapted from Pearce et al 2006

Compared with contingent valuation, choice modelling studies have the potential to provide greater information about people's values (Pearce et al 2006; Morrison 2009). This is because estimates of value can be provided for each of the attributes, or endpoints, of a policy option, not just for the overall scenario as in contingent valuation. This also makes the result of choice modelling studies more suitable for transferring to the analysis of other policies (value transfer – see below). However, choice modelling studies can be more complex to design and analyse than contingent valuation studies (Morrison 2009).

The value of stated preference studies

Stated preference methodologies have an important role to play in environmental valuation, not least because they are the only way that the willingness to pay for non-use values can be estimated.

Despite the growing use of stated preference studies in environmental valuation, however, there remain a number of criticisms of the approach. The first of these is the potential for studies to be biased. Commonly identified sources of bias include:

- the hypothetical nature of the exercise (people aren't spending their own money, so they respond differently than if they were);
- strategic behaviour (people answer in a way that they think supports the result they want); and
- yea-saying (people may not want to disagree with the surveyor).

Secondly, the results from stated preference studies are sensitive to the way the survey questions are asked – the 'framing' of the survey. Also, the level of information the respondent has about the environmental issue at hand, including that provided during the survey, influences the result.

And numerous studies (see discussion in Pearce et al 2006, p. 119) highlight the problem of the embedding effect – that is, individuals' stated valuations can be indifferent to the 'amount' of the environmental good they are valuing. For example, a study by Desvouges et al (1993) estimated a similar community willingness to pay for a program to stop the killing of birds regardless of whether 2,000, 20,000 or 200,000 birds were saved. To the extent that these criticisms hold, the validity and reliability of stated preference results are questionable.

While these are valid concerns, many of the problems associated with stated preference studies can be reduced by clear identification of the environmental endpoints to be valued, and careful design of the surveys. For many policy decisions, consideration of the best available evidence will involve stated preference studies; chapter 6 contains a discussion of how they can be incorporated into regulatory impact analyses.

4. Value transfer

In an ideal world, environmental values would be estimated for each proposed policy, taking into account all of the particular details of the specific policy. However, the use of primary research to estimate environmental values can be costly and time consuming, and in real world policy processes the time and money required often is not available. These realities, combined with the growing literature in the environmental valuation field, makes the possibility of using the values in existing studies an attractive one.

'Value transfer' – often referred to as benefit transfer in the literature – is the process of estimating environmental values in a location of interest (the policy site) by transferring values from studies already completed in another location (the study site). In removing the need for primary research, value transfer can reduce the time and cost associated with estimating the economic value of environmental changes.

There is a need to exercise care transferring data from one study to another. A significant body of academic research has developed looking at the validity of value transfer, and whether estimates of value developed for one purpose can be defensibly used as estimates for other values. While as yet there are no hard and fast rules – value transfer requires judgement and analysis of both the source study and the policy site – a number of points should be considered in order to minimise errors in value transfer.

Firstly, the significance of the policy issue under examination should be considered; in general, the more significant the policy issue, the more careful you will need to be in transferring values from another study.

Secondly, care is needed in order to minimise the errors in value transfer, and these errors potentially come from two sources. Errors can be made in the *source study* – it needs to be of good quality, and produce a valid estimate of the willingness to pay for the environmental good or service in the study site. And errors can be made in *transferring* the estimates from the study site to the policy site.

4.1 Minimising errors in the study site

When you are selecting potential source studies, the first step is to consider the quality of the study – does it do what it purports to do, which is to estimate the willingness to pay for a particular environmental good? Points to consider include:

- Is the study internally valid (Brouwer 2000)? That is, is the survey sample well selected, is the construction of the survey appropriate, and are the statistical techniques used to analyse the results correct? Using more recent studies can help ensure the techniques used to elicit the estimates are state-of-the-art (eftec 2009b)
- Does the size and composition of the sample allow the results to be generalised to the wider community? and
- Are the results what you would expect? So, are the willingness to pay estimates consistent with economic theory (for example, do the estimates

increase as income increases)? And do the estimates increase as the amount of the environmental good that is being valued increase?

4.2 Minimising transfer errors

Probably the most important challenge when selecting suitable study sites for transfer, however, is the similarity of the study site to the policy site. Assuming the studies are of a good quality, the more similar the sites the lower should be the transfer error (Akter and Grafton 2009).

The similarity in the following factors needs to be considered when judging how suitable the study context is to the policy context (eftec 2009a).

- *The policy and study good* – how similar is the change considered in the study site, taking into account the physical characteristics of the goods and the types of use and non-use values derived from the goods?
- *The change in the provision of the policy and study good* – how does the direction, size and timing of the change in the study good compare with the policy good?
- *The locations where the study good and policy good are found* – is the accessibility/proximity to the community of the study site comparable to the policy site?
- *The affected populations in the study site and the policy site* – are similar populations affected by the change in the study site as the policy site? You need to consider the socio-economic characteristics, frequency of use, and types of users of the sites.
- *The number of substitutes for the study and policy good* – for example, the willingness to pay for an additional hectare of fishable lake might be different in Canada to, say, Saudi Arabia.
- *The policy good and market good constructs* – this covers a number of factors, such as the circumstances of the change (is it permanent or a 'one-off'?), the property rights in the cases (is the population 'entitled' to the baseline, or to the changed situation?), the economic situation, etc.

If there are no significant differences between the source and policy sites, and if the source study is of a good quality, then the estimates in the source study are good candidates for value transfer.

If, however, there are significant differences, the estimates from the source study may not be able to be used directly. It may be possible to adjust the estimates to take into account the differences (for a summary of the different ways this can be done, see eftec 2009a, or Pearce et al 2006), or it may be that value transfer is not possible.

4.3 The need to use value transfer carefully

Even after taking account of the factors discussed above and ensuring that source studies are of good quality, there remain questions over the validity of value transfer in environmental valuation. Reasons for unexplained differences between source and policy studies could include the stability of estimates over time (that is, how relevant is a source study undertaken a number of years ago to current environmental values), and completely unobservable differences between populations in the source site and the policy site.

Validity tests of value transfer studies have found that, even when done well, the transfer error (the difference between the transferred value estimate and the ‘true’ value at the policy site) ranges from 0-100 per cent (eftec 2009b). That is, the transfer value is commonly 0-2 times the ‘true’ value.

So, while value transfer is still a viable source of information in environmental valuation – and sometimes the only option open to analysts – a competent application of transfer methods requires informed judgement and expertise (Pearce et al 2006), and the results need to be reported with appropriate caveats. The use and reporting of environmental valuation studies is discussed in part 7 of this guide.

5. Estimates from the literature

As discussed in the previous section, a number of factors need to be taken into account to successfully transfer environmental values from an existing study. The appropriateness of the source study to the policy site needs to be carefully considered, taking into account:

- the context of the environmental changes in both studies;
- the environmental asset – and endpoints – under consideration;
- the size and nature of the environmental changes examined; and
- the relevant populations in the studies.

The study methodology also needs to be considered, including:

- the way the questions in the survey are asked;
- the methodology (stated preference, revealed preference, etc); and
- what adjustments need to be made to the estimates (inflation, income adjustments, exchange rates scope of change, etc).

That said, the following section contains estimates[5] of the value of a number environmental assets. It is not an exhaustive or representative list, but it provides an illustration of the range of studies that have been undertaken in Australia over the last 20 years, and of the estimated willingness to pay for environmental values. You should refer to the individual studies for more information about the estimates.

Reflecting the major issues in natural resource management Australia in the last two decades, the bulk of the environmental valuation work has concentrated on management of remnant native vegetation, and water resources in the Murray Darling Basin.

Table 5.1.Coastal and marine environments

Asset/endpoint	WTP[a]	Unit	Type and year of study[b]	Study
Diving on Great Barrier Reef	$227.17[c]	visitor	CVM - 2004	Kragt et al (2009)
A day's fishing - Queensland	$59.84	fisher/day	CVM - 1996	Campbell and Reid (2000)
A day's offshore fishing	$188.47	angler/trip	TCM - 2007	Prayaga et al (2009)
Beach sand dunes	$83.23	household/ year	CVM - 1992	Pitt (1993)

Notes. **a**. Expressed in $2011 (ABS 2012). Reported values are central measures. Where a range is indicated, it represents the central measures for a number of model specifications, not a confidence interval. You should refer to the original studies to obtain confidence intervals (and other relevant information) for these estimates. **b**. CVM =

[5] The Environmental Valuation Reference Inventory (EVRI), maintained by the Canadian Environment department, is a searchable online database containing details of over 2000 environmental valuation studies. It is a valuable resource for those undertaking environmental valuation research, and can be found at www.evri.ca.

Contingent Valuation Method. **c**. This is the consumer surplus per visitor – that is, the benefit to the visitor over and above the costs of travelling to the area.

Table 5.2.Rivers and wetlands

Asset/endpoint	WTP[a]	Unit	Type and year of study[b]	Study
Improve river quality from boatable to fishable	\$41.83-\$77.20	household	CM - 2000	Morrison and Bennett (2004)
Improve river quality from fishable to swimmable	\$40.30-\$53.84	household	CM - 2000	Morrison and Bennett (2004)
Waterway health	\$0.11-\$0.13	km of waterways in good health/household/year	CM - 2000	Loch et al (2002)
Waterway health	\$0.11	10km of river restored/ household/ year	CM - 2001	Van Bueren and Bennett (2004)
Waterway health	\$0.01-\$0.14	km of waterways in good health/household/year	CM - 2003	Windle and Rolfe (2004)
River estuary in good health	\$0.63-\$4.89	% of estuary remaining in good health/household/year	CM - 2003	Windle and Rolfe (2004)
Wetlands area	\$0.04-\$0.06	km^2 of wetlands/ household	CM - 1997	Morrison et al (2002)
Preservation of wetlands	\$4143	ha/ year	CM - 1998	Mallawaarachchi et al (2001)
Healthy wetlands	\$14.00	1000ha/ household	CM - 2004	Whitten and Bennett (2005)
Recreation	\$0.10-\$2.45	% of river suitable for contact recreation/ household	CM - 2006	Bennett et al 2008
Recreation at lake	\$23.12-\$38.15[c]	visitor (when lake is half full and full, respectively)	TCM - 2006	Crase and Gillespie 2008

Notes. **a**. Expressed in \$2011 (ABS 2012). Reported values are central measures. Where a range is indicated, it represents the central measures for a number of model specifications, not a confidence interval. You should refer to the original studies to obtain confidence intervals (and other relevant information) for these estimates. **b**. CM = Choice Modelling, TCM = Travel Cost Method. **c**. This is the consumer surplus per visitor – that is, the benefit to the visitor over and above the costs of travelling to the area.

Table 5.3. Native vegetation

Asset/endpoint	WTP[a]	Unit	Type and year of study[b]	Study
Healthy riverside vegetation	$1.74-$3.63	% of river covered/household	CM - 2000	Morrison and Bennett (2004)
Native riverside vegetation	$3.71-$5.53	km increase in native riverside vegetation/ household	CM - 2008	Kragt and Bennett (2009)
Healthy floodplain vegetation	$1.67-$1.99	% of healthy vegetation remaining/household/year	CM - 2000	Loch et al (2002)
Healthy floodplain vegetation	$0.88-$4.26	% of healthy vegetation remaining/household/year	CM – 2003	Windle and Rolfe (2004)
Native vegetation	$3.71-$6.23	% increase of healthy vegetation along river/ household	CM – 2006	Kragt et al (2007)
Preservation of tea tree woodlands	$26.52	ha/ year	CM - 1998	Mallawaarachchi et al (2001)

Notes. **a**. Expressed in $2011 (ABS 2012). Reported values are central measures. Where a range is indicated, it represents the central measures for a number of model specifications, not a confidence interval. You should refer to the original studies to obtain confidence intervals (and other relevant information) for these estimates. **b**. CM = Choice Modelling.

Table 5.4. Aesthetics / National Parks

Asset/endpoint	WTP[a]	Unit	Type and year of study[b]	Study
Rural lands restored/ protected	$0.09	10 000 ha restored/ household/ year	CM - 2001	Van Bueren and Bennett (2004)
Hiking – Tropical National Park	$192.33-$686.77[c]	visit	TCM - 2001	Nillesen et al (2005)
National park visits	$24.20-$26.89[c]	visit	TCM - 1995	Bennett (1995)

Notes. **a**. Expressed in $2011 (ABS 2012). Reported values are central measures. Where a range is indicated, it represents the central measures for a number of model specifications, not a confidence interval. You should refer to the original studies to obtain confidence intervals (and other relevant information) for these estimates. **b**. CM = Choice Modelling, TCM = Travel Cost Method. **c**. This is the consumer surplus per visitor – that is, the benefit to the visitor over and above the costs of travelling to the area.

Table 5.5. Animal species

Asset/endpoint	WTP[a]	Unit	Type and year of study[b]	Study
Native fish species	$2.81-$8.71	species/household	CM - 2000	Morrison and Bennett (2004)
Native fish	$0.42	% of fish population/ household	CM - 2004	Whitten and Bennett (2005)
Fish species	$4.65-$6.77	species/household	CM - 2006	Kragt et al (2007)
Waterbird and other fauna species	$1.21	species/household	CM - 2000	Morrison and Bennett (2004)
Waterbirds and native animals	$2.46-$3.93	species/ household	CM - 2007	Kragt et al (2007)
Frequency of waterbird breeding	$14.56-$35.88	waterbird breeding event/ household	CM - 1997	Morrison et al (2002)
Native birds	$0.68	% of bird population/ household	CM - 2004	Whitten and Bennett (2005)
Endangered species	$0.91	species protected /household /year	CM - 2001	Van Bueren and Bennett (2004)
Rare native animal and plant species	$8.21-$14.36	species protected/ household	CM - 2006	Kragt and Bennett (2009)
Presence of endangered and protected species	$4.77-$6.34	species present/ household	CM -1997	Morrison et al (2002)
Leadbetter's Possum	$42.37	household/ year	CVM - 1999	Jakobsson and Dragun (2001)
Mahogany Glider	$32.30	household to protect habitat and secure species survival	CVM - 2002	Tisdell et al (2004)

Notes. **a**. Expressed in $2011 (ABS 2012). Reported values are central measures. Where a range is indicated, it represents the central measures for a number of model specifications, not a confidence interval. You should refer to the original studies to obtain confidence intervals (and other relevant information) for these estimates. **b**. CM = Choice Modelling, CVM = Contingent Valuation Method.

6. Decision making under uncertainty

Uncertainty is present to a certain extent in all policy decisions. And given the state of knowledge of many biological and biophysical processes, this is particularly the case in environmental policy making. As such, you cannot just concentrate on solving problems where there is perfect certainty – no decisions will ever be made. The key is not to ignore uncertainty or to assume it away in your analysis. Uncertainty needs to be acknowledged, described and, where important, taken into account in the decision-making framework.

Identifying the sources of uncertainty; understanding how they contribute to decision uncertainty; and the management of uncertainties within the assessment and decision-making process are therefore essential to making well-informed decisions (Willows and Connell 2003). Uncertainty, and the difficulties in forecasting with perfect accuracy, will inevitably mean that not all decisions will produce the benefits that were intended. However, any decision should, even with the advantage of hindsight, be justifiable on the basis of the available knowledge at the time of the decision (Willows and Connell 2003).

6.1 What is uncertainty, and where does it come from?

Uncertainty means different things to different people, and to different academic disciplines; in economics, a distinction is commonly drawn between risk and uncertainty (PC 2012).

'Risk' is the probability of an event occurring, and the magnitude and/or consequences of that event. In common usage, risk is something that can be measured (Knight 1921), and relatively firm conclusions can be drawn from that measurement.

'Uncertainty' describes a situation where there is a lack of knowledge concerning outcomes (Willows and Connell 2003); the state of knowledge is insufficient to draw conclusions about the probability of an event occurring, or the magnitude and/or consequences of that event.

Risk management has its own challenges for policy makers, but it can be handled in a relatively straightforward manner in cost-benefit analysis (see Commonwealth of Australia 2014c for a discussion of risk in CBA).

Decision making under uncertainty, however, throws up a number of issues that need to be carefully considered – among these are the source and the nature of the uncertainty.

Uncertainty can come from a range of sources, and these sources will vary depending on the type of decision being made. Following Moss and Schneider (2000), uncertainty in the environmental and climate change areas can come from, among other sources:

- *Data problems*: there can be a lack of data, inadequate or incomplete measurement, or an inherent randomness of the value we want to know about.

- *Model problems*: that is, our level of understanding of the way the world works in relation a particular issue may be incomplete. For example, it may be unclear how, or how much, a change in one variable impacts on another that you are interested in. Or the uncertainty may be even deeper[6] than this – it may even be unclear *if* one variable impacts on another.

The nature of uncertainty is also an important consideration. A distinction is commonly drawn between reducible uncertainty and irreducible uncertainty.

Reducible uncertainty is that which can in theory be reduced by acquiring more information, or conducting more research. For example, as more data becomes available about global climate change and as this is fed into better climate models, some uncertainty about the future climate can be reduced.

Irreducible uncertainty refers to that uncertainty that comes from the inherently unpredictable nature of human and natural systems (PC 2012) – even if more research is undertaken, there will be a certain amount of uncertainty that will not be able to be resolved. For example, even with better climate data and models it may not be possible to predict future rainfall levels in a particular location.

Being able to describe the sources and nature of uncertainty involved in a decision is important as it will help define the options that are open to decision makers, and the relative attractiveness of those options.

6.2 How to incorporate uncertainty into decision-making framework

When uncertainty is well characterised and able to be quantified, and the processes underlying the policy issue are well understood, the 'optimum expected utility approach' is the appropriate criterion for selecting the best policy option (Lempert and Collins 2007). In the case of CBA, this involves selecting the option with the highest expected net present value (see Commonwealth of Australia 2014b and Commonwealth of Australia 2014c)

However, as discussed above, this is often not the case. Uncertainty – over values and processes – is common, and in cases where there is significant uncertainty an optimal expected utility approach may not be appropriate (Lempert and Collins 2007).

The precautionary principle has often been put forward as an alternative decision making criteria. This principle states that 'where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation' (United Nations Conference on Environment and Development 1992). Taken at face value, this principle is an appealing one – indeed it is one to which a reasonable person would find unobjectionable (Sunstein 2002). However, there are different notions of what the precautionary principle does and does not

[6] Deep uncertainty can be described as a situation where there is little agreement on or knowledge about models that relate key forces that shape the future; the probability distrutions of key variables and parameters in these models; and/or the value of alternative outcomes (Hallegatte et al 2012)

involve (Morgan et al 2009) – see Box 2 – so more specific approaches to managing uncertainty need to be discussed.

The first response to decision making under uncertainty may be to reduce the uncertainty. Research to provide better information is important, and will allow better decisions to be made in the future.

Box 2. The precautionary principle

One approach to dealing with uncertainty in policy making is the application of the precautionary principle. The precautionary principle is incorporated into many treaties and resolutions, including the Montreal Protocol on ozone depleting substances, The United Nations' 'Rio Declaration' on the environment and development, and the UN Framework Convention on Climate Change.

The precautionary principle can be understood as a continuum. Definitions range from the 'weak', which suggest that a lack of evidence should not be a ground for refusing to regulate – the Rio declaration is an example of this – through to 'strong' definitions that suggest that regulation is required whenever there is a risk of damage to health, safety or the environment (Sunstein 2002).

In its weak form, the precautionary principle is a sound approach to decision making under uncertainty – it recognises that waiting until uncertainty is resolved (if it can be resolved) has its own costs, and that a better approach is to leave open the ability to take action to mitigate a risk if the benefits of doing so are likely to outweigh the costs. The purpose of weak versions of the principle is to act as a 'rebuttal to the mistaken claim that uncertainty warrants inaction' (Wiener 2002, cited in Weier and Loke 2007). This approach is consistent with the RIS process, and the approaches to uncertainty outlined in this guide.

In its strong form, however, the precautionary principle provides little guidance for policy makers. The problem is that under uncertainty, risk can be found on all sides of regulatory choices, and any action (or inaction) can violate the principle (Sunstein 2002). Take the example of nuclear power. The risks and uncertainties associated with nuclear power may warrant, under the precautionary principle, banning its use to prevent accidents and environmental damage. However, this would increase the risks associate with an increased reliance on fossil fuels, namely (slightly different) health and safety risks, and would in itself be a violation of the principle. It is for this reason the Sunstein (2002) calls it the 'paralysing principle'.

However, uncertainty will exist in the interim and is likely to remain to some extent after any research is undertaken, so uncertainty will still need to be incorporated into decision making. A number of decision-making approaches have been identified that explicitly incorporate uncertainty. These approaches move away from finding the 'optimal' policy at a point in time, and instead introduce criteria such as resilience and adaptability to help inform decision makers.

A resilient strategy is one that will work reasonably well across a range of uncertain outcomes. In essence, a resilient strategy will trade some level of optimality for less sensitivity to changes in certain parameters (Lempert and Collins 2007). Resilient strategies range from the relatively simple (such as the minimax decision rule, which states that the best option is one that minimises the loss in the worst case scenario), to more complicated approaches that incorporate sophisticated scenario analysis and computer modelling (Lempert 2002).

An adaptive strategy is one that allows decision makers to respond to new information as it becomes available. Under conditions of uncertainty, particularly where the level of uncertainty is likely to reduce over time as knowledge improves, policy options that offer greater flexibility – that allow adaptation to new information – are likely to have more value than options that are not as flexible.

As an analogy, imagine that you are starting a new business. It is unlikely that you would plan to produce a given mix of products, in given quantities, and at certain prices, and not adjust this in the face of new information about demand, substitutes, etc. And you would also be unlikely to continue to produce and market a particular product if it became unprofitable. You would be more likely to respond to this new information, and adapt your production and marketing efforts accordingly.

Likewise, the ability for a decision maker to flexibly respond to new information as it becomes available is likely to be of value, and this needs to be reflected in the analysis of alternatives open to decision makers. The analysis of 'real options' (see Box 3), is an important way of reflecting this value.

Box 3. Real options analysis

As information improves through time, 'better' decisions can be made – ones that reflect more fully the actual state of the world, and are less compromised by the presence of uncertainty. Keeping open the ability to make a decision until information improves, therefore, has a value: this value is the welfare difference between a decision made under uncertainty, and one made with better information.

In financial markets, the ability to realise this value is made possible by the presence of 'options'. In its simplest form, an option is the right, but not the obligation, to purchase something at an agreed price at a later date. As an example, a wheat farmer planting a crop may be uncertain what the price for his crop will be in 6 months' time when it is harvested, so is unsure about how much wheat to plant. To remedy this, he can purchase an option to sell his wheat at, say, $150 per tonne in 6 months' time – this effectively locks in a minimum price, and allows him to make better planting decisions with greater certainty about the price he will receive. If in the meantime the price rises, he can forgo his right to sell his crop at $150 and sell it for the higher price on the open market.

In a similar way, some policy alternatives made under uncertainty have been likened to 'real' options. That is, by avoiding irreversible decisions until after more information becomes available, a better outcome may be able to be achieved. Take the example of the decision to build a sea wall to prevent the possibility of damage from higher sea levels caused by climate change. Because of the uncertainty surrounding the extent of climate change and its impact on sea levels, it is not clear whether a low or high sea wall should be built. Rather than building a 'worst case scenario' sea wall, a smaller wall on strong foundations can be built so that the height of the wall can be increased to any height in the future to match actual conditions without incurring unnecessary upfront expense (Dobes 2008). This in effect provides the decision maker with the right, but not the obligation, to build a high wall if necessary.

The ability for decision-makers to flexibly respond to new information as it becomes available gives additional 'real option' value to the policies that allow this – this value needs to be recognised in impact analyses.

6.3 The importance of describing uncertainty

When working in a policy environment involving significant uncertainty, the limitations of the our knowledge need to be assessed and presented to decision makers in a way that is useful. Issues that should be considered include:

- What are we uncertain about? Is it values, model parameters, or is the uncertainty very deep?
- What is the size of the uncertainty? Is there a plausible range over which it could be expected the true value lies (and what is the evidence for this), or is the real situation truly unknown?

- How is the uncertainty likely to change over time, and over the lifetime of the decision being made? What research efforts etc are underway, and how is this expected to change the relevant stock of knowledge?

The size of the uncertainty, and how important it is to the decision that needs to be made, should be reflected in the language that is used in the analysis, and the strength with which policy action can be recommended. For example, if the recommendation of the preferred policy option is sensitive to a particular process or value in an analysis that is uncertain, then the recommendation needs to be tempered by appropriate caveats. On the other hand, if there is little uncertainty, or if the policy recommendation is not sensitive to that process or parameter, then more forceful recommendations may be able to be made.

Under significant uncertainty, decision making does not follow a cookbook approach, and analyses incorporating uncertainty do not necessarily yield a single best answer. Rather, looking at options that include resilient and adaptive strategies helps decision makers to use the best available information to distinguish a set of reasonable choices from unreasonable ones, and to understand the trade-offs involved in choosing among them (Morgan et al 2009).

7. What all of this means when you are writing a RIS

For significant regulatory proposals, both the Australian Government and Council of Australian Governments requires the preparation of a regulation impact statement (RIS)[7]. While the specific requirements differ slightly, in general terms a RIS contains discussion of the following elements (Commonwealth of Australia 2014a):

- the problem or issues that give rise to the need for action;
- the desired objectives;
- a range of options (regulatory and non-regulatory, as applicable) that may constitute feasible means for achieving the desired objectives;
- an assessment of the impact (costs, benefits and, where relevant, levels of risk) of a range of feasible options for consumers, business, government and the community;
- a consultation statement;
- a conclusion and recommended option; and
- a strategy to implement and review the preferred option.

The description and valuation of environmental impacts and the treatment of uncertainty particularly affect the problem, impact analysis and concluding sections of a RIS. The issues to consider in each of these sections are outlined below.

7.1 Statement of the problem

The ‘problem’ in the context of a RIS is the issue that has prompted consideration of government action. When preparing a RIS, a clear statement of the problem is important as it provides the context within which government objectives are set, and the impacts of alternative policy options are considered.

In environmental policy, the problem is often about changes that have occurred, are occurring, or may occur to environmental assets. But change, whether in response to human activity or other causes, is a feature of environmental systems; indeed it can be difficult to define a ‘natural’ state in a changing environment (Sprugel 1991). It is important, then, to discuss what it is about the particular changes in question that require government intervention. This could include the:

- causes of the change – for example, where the cause is the result of an environmental externality (see Commonwealth of Australia 2014a, page 23), it is possible that the change is not in the best interest of the community;
- size or pace of the change – particularly large or fast changes to the environment may be a cause of concern, particularly where the presence of ‘tipping points’ make these changes irreversible; or
- value to the community – where features of the current environment are highly valued, change may be undesirable. Examples of such features could

[7] See Commonwealth of Australia (2014a), and COAG (2007b) for a detailed discussion of the RIS requirements.

include the presence of endangered species, the role the environmental asset plays in primary production, or the presence of high recreation values.

You should refer to empirical evidence about the significance of the problem, where available, as well as perceptions of the problem. If the problem involves risk to the public, businesses, workers or the environment, you should include a description of the hazard and a discussion of the likelihood that it will occur. This should include assessing the worst and best outcomes that could occur if a 'do nothing' approach is taken (Commonwealth of Australia 2014c).

In order to present a useful description of the problem, it is important to not only define the problem in biological or physical terms – how will the asset change if we don't do anything? – but also the impact that the changes will have on the community, and its wellbeing.

You should include the following information in the problem section:

- *A description of the environmental asset* – including factors such as its size, location and importance (see discussion in part 2.1)
- *How the asset is valued, and by whom* – this discussion could draw on the ecosystem services and total economic value frameworks outlined earlier in the paper to identify the various endpoints that are valued by the community. The community should also be defined: is the environmental asset in question valuable only to locals, or is it of significance to the whole Australian community?
- *Identification of the baseline* – in the absence of intervention, what will happen to the environmental asset and the relevant endpoints over time? This is the crux of the problem section: it is generally your projection of what will happen to the asset that provides a rationale for intervention to change this course. You should develop this baseline with reference to the best available information, and the evidence on which you base your analysis needs to be presented. Where possible, the changes in the endpoints over time should be quantified.

You also need to report clearly any knowledge gaps or uncertainties in your analysis of the problem, and how this may affect the rationale for government intervening in this particular policy area.

7.2 Impact analysis

The role of the impact analysis is to assess the costs and benefits of each possible option for addressing the identified problem. This involves drawing up alternative scenarios, each representing the 'state of the world' under each policy option, and examining the differences between these scenarios and the baseline.

When the policy options have environmental impacts, you need to examine what is likely to happen to the environmental assets under the different policy scenarios. Firstly, this involves a qualitative description of what changes are likely to occur under the various policy scenarios, and why. In examining why the changes are likely to occur, you should describe the underlying social, physical and biological processes contributing to the changes. Again, this discussion needs to be informed by the best available information.

Once the likely changes in the environmental asset are established, you will need to identify how they affect the relevant endpoints that are identified in the problem section, as discussed in part 2. At this stage, you should quantify the impacts on the endpoints where possible, and identify the difference between the level of the endpoints under each option and the baseline.

In some cases it will not be possible to quantify some of the impacts of policy changes on environmental assets or endpoints. In these cases, it is important not to exclude an important impact category from your analysis, but instead provide a good qualitative discussion of the likely impacts, using the discussion below as a guide. You will also need to explain clearly why a quantitative analysis was not possible.

7.2.1 Valuation of impacts

Where possible, the community's willingness to pay for the identified changes in endpoints should be estimated, using the sort of valuation techniques discussed in part 3 of this guide. Which valuation method to use is best assessed on a case-by-case basis, as it depends in part on the specific circumstances of the environmental changes being examined. One of the important factors to consider is what type of values are being estimated – that is, whether the values are use or non-use values. Some valuation techniques are better suited to certain types of values, as displayed in table 7.1.

It is important to note that when you are applying economic values to environmental impacts, the valuations are generally only applicable to relatively small – or marginal – changes in the environmental asset. This is particularly true when estimating non-use values (Morrison and Hatton-MacDonald 2010). As a result, issues involving non-marginal changes to the environment such as global warming, mass extinctions, or dramatic changes in pollution levels have to be handled carefully instead of merely extrapolating from results based on small changes (Dixon 2008).

Table 7.1. Choice of valuation method

Valuation method	Element of TEV captured	Pros and cons
Market prices	Direct and indirect use	Data readily available and robust, but limited to services for which a market exists
Hedonic pricing	Direct and indirect use	Based on observed behaviour, but generally limited to recreational values
Travel cost	Direct and indirect use	Based on market data, but data intensive
Contingent valuation	Use and non-use	Can capture non-use value, but bias in responses, resource intensive and hypothetical in nature
Choice modelling	Use and non-use	As above

Source: Adapted from DEFRA 2007

When reporting the results of primary research into the valuation of environmental impacts, you need to provide an outline of the study in the RIS. At a minimum, you should discuss:

- the valuation method/s chosen, and a brief description of how the methodology was applied to the environmental changes examined in the RIS;
- how the results compare with studies undertaken into other, similar environmental assets; and
- whether there any particularly important caveats about the results that need to be noted.

When you are using value transfer in a RIS, there are a number of things you particularly need to report.

- *identify the relevance of the source study site to the policy site* – for example, how similar are the environmental assets in question? How similar are the populations that value the environmental services? Are the changes valued in the source site similar in size and type to those you are looking at in the policy site?
- *present a range of estimates* – your analysis should not rely on a point estimate of the value of the environmental asset in question. Value transfer is not an exact science, and differences between the value estimated by value transfer and the 'true' valuation have been found to be up to 100 per cent, even in the best examples of value transfer (eftec 2009b). As such, the RIS should contain a sensitivity analysis of the transferred value; ranges of values may be based on confidence levels in the source study, or based on the ranges found in similar studies (eftec 2009a).
- *discuss the limitations* of the methodology. You should clearly point out that the values transferred were not estimated with reference to the specific environmental changes being examined in the RIS, and that as a result there remains some uncertainty about the community's willingness to pay. At best, value transfer can provide an indication of the order of magnitude of the community's willingness to pay for environmental services.

7.2.2 Reporting on uncertainties

As discussed in part 6, uncertainty plays a role to some extent in all policy decisions, and needs to be addressed in the impact analysis of a RIS. And also as discussed in part 6, we can think about two different types of uncertainty: uncertainty over values, and deep uncertainty over processes.

Uncertainty over values can be taken into account by using a number of various tools and techniques – see Boardman et al (2006) for a good summary. For the purposes of impact analysis in a RIS, two important techniques are (Commonwealth of Australia 2014c):

- *Sensitivity analysis*. Sensitivity analysis is generally applied to assess the impact of changes in a key, uncertain variable on the overall net benefit estimate. In the case of a RIS incorporating environmental valuation, the sensitivity analysis should include an examination of the estimated impacts of the policy on the environmental asset and endpoints, and the valuation that is applied to the impacts. Sensitivity analysis can provide useful insights into the basis of 'worst' and 'best' case outcomes.

- *Probabilistic modelling*. Where data allows, probabilistic modelling can provide a more sophisticated analysis of the effect of uncertainty over values on the likely impacts of a policy. Monte Carlo[8] analysis can be used to evaluate the effects of simultaneously changing a number of assumptions in the CBA and assessing the impact on net benefit estimates.

Where we are talking about 'deep' uncertainty over processes, you need to outline the sources and implications of the uncertainty, as discussed in part 6. This includes describing the uncertainty: its sources, size, and how it is likely to change over time; and the implications of the uncertainty for the decision maker. You will need to make a judgement as to whether the uncertainty is significant enough to consider resilient or adaptive strategies as policy options.

7.2.3 Who wins, who loses?

In addition to assessing the overall net impact of the policy options, your RIS will need to discuss the distribution of impacts – that is, which groups in the community are better or worse off, and by how much. Impacts on different environmental endpoints may be valued by different groups.

As an example, improvements in a wetland may be assessed as having impacts on irrigation water quality, aesthetics and the abundance of an endangered bird species. These benefits may accrue to different groups: farmers in a nearby irrigation district, landholders bordering the wetland, and the broader community respectively, and it is important to convey this information to decision makers.

7.2.4 Summarising the impacts – the net benefit

Once you have identified, analysed and (hopefully) quantified the likely impacts of the proposals, the final step in the impact analysis is to make an assessment of the net impact of each policy option.

In a perfect world, where you have been able to fully quantify all of the impacts, and there is little uncertainty around the values and processes involved, the net present value (NPV) of each option would be determined. An NPV is calculated by subtracting the discounted sum of all of the costs from the discounted sum of the benefits; if the result is greater than zero, the policy is economically efficient, and the community as a whole is better off. (See Commonwealth of Australia 2014b for a more detailed discussion of how to calculate the NPV.)

But we do not operate in a perfect world, and it is rarely the case that all of the impacts will be able to be confidently quantified (US EPA 2010). Information gaps and data uncertainties are common, particularly when you are considering environmental impacts. And in any case it is not so much the quantification of

[8] Monte Carlo sensitivity analysis allows the values of a number of uncertain variables to be chosen at random, according to pre-determined probability distributions. This process is repeated multiple (sometimes thousands) times by computer, generating a probability distribution of model outcomes.

every last impact that is important, but the action of analysis – questioning, understanding real world impacts, exploring assumptions.

An important alternative to a fully quantified NPV as a basis for assessing the net impact of the policy options is a breakeven analysis. Break even analysis – also referred to as threshold analysis – involves quantifying as many impacts as possible, but leaving the uncertain impacts unquantified. The level of impact that causes the NPV of the proposal to equal zero is the break-even value: the policy just breaks even at this value (Sinden and Thampapillai 1995).

For example, suppose all of the costs of a proposed policy to improve the condition of a particular wetland have been quantified, as has the (beneficial) impact on the surrounding farmland. Your analysis has also identified that the broader community gains non-use benefits from the policy; you can identify from other studies the broad size of this value, but cannot pin it down with any confidence.

Under a breakeven analysis, you would subtract all of the quantified costs from the quantified benefits, and identify how large the remaining non-use values would need to be to make the NPV of the policy equal zero, or ‘break even’. You would then discuss the likelihood of the policy delivering non-use values of this size, with reference to the range of estimates gleaned from the existing studies you have identified. In this way, the best available information is presented to the decision maker in a way that appropriately conveys the confidence you have in the information

Where quantification of environmental impacts is just not possible or feasible, the RIS should contain a good, qualitative analysis of the issues using the sort of framework discussed earlier in this guide. You should not exclude consideration of certain impacts just because it is difficult to quantify them.

Where there is deep uncertainty about aspects of the decision, you will also need to consider the value that resilient or adaptive strategies can provide, in addition to the value captured in the estimate of NPV. In most cases it is unlikely that you will be able to estimate this value quantitatively, so you will need to describe these issues, and make a judgement as to how they impact on the attractiveness of the policy options.

Just as important in describing the source and implications of any uncertainty is considering how best to convey this information to decision makers. The form of language in any policy recommendations – and the confidence that it conveys that a particular course of action is the best option – will need to reflect these underlying uncertainties.

7.2.5 A note on discounting

Most projects or policies assessed in a CBA will involve impacts that occur in different time periods. Discounting refers to the process of assigning a lower weight to a unit of benefit or cost in the future than to that unit now –the further into the future that the impact occurs, the less weight it is given (Pearce et al 2006). See Commonwealth of Australia (2014b) for a discussion on the rationale for and calculations involved in discounting.

The OBPR recommends using a real discount rate of 7 per cent, with a sensitivity analysis at 3 and 10 per cent reflecting the uncertainty around the 'true' discount rate (Commonwealth of Australia 2014b)[9]. For analyses involving very long time frames, this uncertainty means that it is appropriate to use a time declining discount rate (see Appendix 1 for a detailed discussion). As such, for analyses involving a period of analysis of more than 30 years, the OBPR recommends using the rates displayed in table 7.2.

Table 7.2. Declining long term discount rate [a]

Period of years	1-30	31-75	76-125	126-200	201-300	301+
Discount rate	7.0%	5.4%	4.8%	4.3%	4.0%	3.7%

Notes:. **a.** See Appendix 1 for an explanation of how these rates were derived.

It is argued that for cost-benefit analyses involving environmental impacts or intergenerational issues, a lower discount rate should be used. The following two arguments are commonly cited.

- Many environmental impacts occur in the long term. The process of discounting means that such impacts are given much less consideration than other, nearer impacts, and this 'tyranny of discounting' (Pearce et al 2006) should be corrected by using a lower discount rate.
- It is ethically dubious to give more weight to one generation than to subsequent generations. Analyses of policies or programs with intergenerational impacts, therefore, should use lower discount rates so that future generations' interests are appropriately considered.

While these are valid considerations, adjusting the discount rate may not be the best way to take them into account. In both cases, adjusting the discount rate requires the analyst to make an ethical judgement about the value that is attached to the welfare of future generations (Nordhaus 2007), or to environmental impacts. The problem with ethical judgements is that there are a number of different ethical stances that can be taken in relation to an issue, and no generally accepted means of assessing which should take precedence.

More generally, CBA is about establishing the *efficiency* of a policy or project. While equity considerations can be examined, CBA is not a good tool for determining whether a decision can be justified on equity grounds. It may be more useful for decision makers to assess environmental or intergenerational impacts explicitly, using the framework described in this guide, than to implicitly take them into account by adjusting the discount rate. Mixing equity and efficiency in cost-benefit analysis can prevent insights into efficiency from being gained and give a false sense of objectivity to the results (Baker et al 2008). And it can obscure the important points about environmental impacts and intergenerational impacts that need to be made explicitly to decision makers.

[9] These recommended discount rates apply to the analysis of the type of 'marginal' policies that are commonly subject to regulatory impact analysis. For regulations that involve large, non-marginal changes to the economy, a number of other factors need to be considered in setting discount rates – see Stern (2008) for a discussion.

Given the uncertainty and debate around the appropriate social discount rate, conducting a sensitivity analysis using the discount rate is an important part of the CBA. If sensitivity analysis suggests that the specification of the discount rate has a significant impact on the preferred outcome (for example, if under a high rate a particular policy option is undesirable, but is preferred under a lower rate), this is important information to convey to decision makers, and can lead to a more illuminating qualitative discussion of the key intertemporal aspects of the policy. Important issues to discuss include:

- how the costs and benefits are distributed over time; and
- who these costs and benefits impact affect. For example, a climate change mitigation proposal may have costs that exceed benefits for the first 30 years, with benefits exceeding costs after that time. The RIS should point out that the costs are borne most heavily by the current generation, with future generations (including many people alive now) receiving the benefits.

7.3 The conclusion

The aim of the conclusion section in a RIS is to provide a clear statement identifying the preferred option; indicate the costs and benefits for the range of groups that are affected; and highlight any areas of uncertainty (Commonwealth of Australia 2014a)

It is vital you provide a *balanced* summary of the impacts of the proposed option. This means that the summary must be supported by the preceding analysis in the RIS, and convey any recommendations in a way that reflects your confidence in the underlying assumptions and findings. Where there are important caveats in the analysis –whether they relate to the methodologies used to estimate or value environmental impacts, or uncertain knowledge about values or underlying processes – these need to be noted.

Appendix 1. Uncertainty and declining discount rates

The OBPR currently recommends using a constant real discount rate of 7 per cent in regulatory cost-benefit analysis (Commonwealth of Australia 2014b). Reflecting the uncertainty inherent in calculating a social discount rate, it also recommends conducting sensitivity analysis using a range of discount rates: following from Harrison (2010), the recommended real discount rates for sensitivity analysis are 3 per cent and 10 per cent.

Research over the last 10-15 years, however, suggests that when discounting in the very long term it is appropriate to incorporate uncertainty in future discount rates in the form of a declining discount rate (Weitzman 1998, 2001; Gollier 2002). Discount rates are used in cost-benefit analysis through the calculation of a discount factor:

$$d = \frac{1}{(1+r)^t}$$

where d is the discount factor, r is the discount rate, and t is the time period (usually expressed in years). Where there is uncertainty about the appropriate discount rate/factor to use, it is appropriate to use a 'certainty-equivalent discount factor' – which is the sum of the probability-weighted discount factors (Guo et al 2006):

$$cedf_t = \sum \rho_i d_{it}$$

where $cedf_t$ is the certainty-equivalent discount factor for a given period, d_{it} is the discount factor associated with a particular (uncertain) discount rate *i* in a given period, and ρ_i is the probability of discount rate *i* being the 'true' discount rate. It turns out that when the certainty-equivalent discount factor for a particular range of discount rates (and their associated probabilities) is calculated through time, the implicit 'certainty-equivalent discount rates':

$$cedr_t = \left(\frac{1}{cedf_t}\right)^{\frac{1}{t}} - 1$$

decline through time towards an asymptote equal to the lowest discount rate in the uncertain range[10].

This can be illustrated using a numerical example (drawn from Guo et al 2006). Suppose you are unsure as to whether the true discount rate is 3 per cent or 6 per cent, and there an equal probability of each being the case. The resulting certainty-equivalent discount rates are shown in table A.1.

[10] See Weitzman (2001) for a formal specification of the distribution of certainty equivalent discount rates.

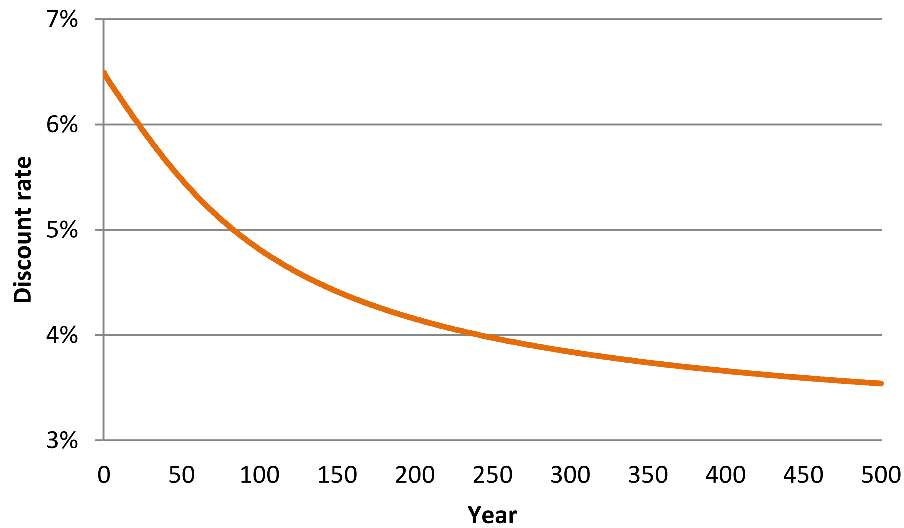
Table A.1: A numerical example of declining discount rates

Discount rates	Discount factors in period t				
Year	10	50	100	200	500
3% (ρ=0.5)	0.744	0.228	0.052	0.003	0.000
6% (ρ=0.5)	0.558	0.054	0.003	0.000	0.000
$cedf_t$	0.651	0.141	0.027	0.001	0.000
$cedr_t$ (%)	4.4%	4.0%	3.7%	3.4%	3.1%

The OBPR's recommendation

The OBPR (Commonwealth of Australia 2014b), following Harrison (2010), recommends conducting sensitivity analysis on discount rates using 3 per cent and 10 per cent. Taking these rates as the range of uncertainty around the 'true' discount rate in a period, and assuming (in the absence of any better information) that rates within this range are equally likely, the resulting time-declining discount rate, plotted against time, is displayed in figure 5[11].

Figure A.1: Time declining discount rates



Source: OBPR

[11] The certainty-equivalent discount factors were calculated using each of the discount rates between 3 and 10 per cent inclusive, at 0.5 per cent intervals. The ρ for each rate was assumed to be equal.

A stepped schedule of rates (table A.2) offers a good approximation of the rates displayed in figure 2. Taken together with the OBPR's recommended real discount rate of 7 per cent, these are put forward as a recommendation when undertaking cost-benefit analysis involving discounting of impacts in the very long term.

Table A.2. Declining long term real discount rates for use in regulation impact statements

Period of years	1-30	31-75	76-125	126-200	201-300	301+
Discount rate	7.0%	5.4%	4.8%	4.3%	4.0%	3.7%

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