Environmental Policy Analysis: A Guide to Non-Market Valuation

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Productivity Commission

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The Productivity Commission

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OVERVIEW

Key points

- Government policies aimed at generating environmental benefits almost always impose costs on the community. Weighing up these trade-offs is challenging, in part because environmental benefits are difficult to value, particularly those that are not reflected in market prices (so called 'non-market' values).

- There are several non-market valuation methods that can be used to evaluate such trade-offs, but they are not widely used for environmental policy analysis in Australia.

- There are two main types of non-market valuation methods: revealed preference and stated preference.

  – The validity of revealed preference methods is widely accepted, but there are many circumstances where they cannot provide the estimates needed for environmental policy analysis.

  – Stated preference methods can be used to estimate virtually all types of environmental values, but their validity is more contentious.

- The evidence suggests that stated preference methods are able to provide valid estimates for use in environmental policy analysis. However:

  – there are many elements that practitioners need to get right to produce meaningful results

  – value estimates are likely to be less reliable when respondents are asked about environmental assets that are especially complex or relatively unfamiliar to them.

- Benefit transfer involves applying available value estimates to new contexts. Its accuracy is likely to be low unless the primary studies are of high quality and relate to similar environmental and policy contexts. These seemingly obvious cautions are often not observed.

- Because non-market valuation methods can generally provide objective estimate of the value that the community places on environmental outcomes, they offer advantages over other approaches to factoring these outcomes into policy analysis.

- The case for using non-market valuation varies according to circumstances. It is likely to be strongest where the financial or environmental stakes are high and there is potential for environmental outcomes to influence policy decisions.

- Where non-market valuation estimates are made they should generally be included in a cost–benefit analysis. Sensitivity analysis should be provided, as well as descriptive information about the environmental outcomes of the proposed policy.

- There is a range of steps that could be taken to realise more fully the potential of non-market valuation, including developing greater knowledge about it within relevant government agencies.

Overview

Governments are often faced with decisions about whether to impose costs on the community to improve the condition of the environment (or prevent its deterioration). How such policy trade-offs should be made is a matter of considerable debate. Some stakeholders favour prioritising environmental outcomes above other considerations, while others argue that jobs and economic development should come first. The former approach effectively assigns an infinite value to environmental outcomes, while the latter assigns a value of zero.

The Australian, State and Territory Governments generally favour analysing policy decisions using a cost–benefit framework. In many inquiries, the Productivity Commission has supported this approach because it allows decisions to be informed by the trade-offs that the individuals who make up the community would be prepared to make. By applying a cost–benefit framework (ideally through cost–benefit analysis), governments can endeavour to make decisions that make the community better off overall.

However, applying a cost–benefit framework to environmental policy is not easy. Some of the costs, such as the budgetary cost of investing in environmental programs, are straightforward to determine. Others, like the costs to business of restricting development or applying environmental regulations, can be somewhat more challenging to assess. But estimating the benefits can be harder still.

There are two parts to estimating benefits. First, information is needed on how the condition of the environment will be changed by the policy. Such information can be hard to obtain because of incomplete understanding of ecological processes and behavioural responses to policy. Second, a value needs to be placed on the change in condition. This can be particularly difficult where values are not reflected in market prices (so called 'non-market' values). For example, while it is clear that many people value the experience of bushwalking in a national park or knowing that particular ecosystems are being maintained in a healthy condition, there are no market prices that directly reflect these values.

Over the last few decades several non-market valuation methods have been developed to estimate such values, but to date they have not been widely used in policy analysis in Australia. These methods appear to have considerable potential for improving environmental policy and so their limited use is a puzzle. Either the

potential is illusory because the methods cannot reliably do what is claimed, or the reluctance to use them is a lost opportunity that should be rethought.

This paper examines these issues by:

- assessing the validity and reliability of various non-market valuation methods

- reviewing the case for using non-market valuation in environmental policy analysis

- offering suggestions on how best use can be made of non-market valuation in developing environmental policy.

The validity of non-market valuation methods

There are two main types of non-market valuation methods: revealed preference and stated preference. In addition, benefit transfer is a technique that can be used to apply existing value estimates to new contexts.

Revealed preference methods

Revealed preference methods use observations of purchasing decisions and other behaviour to estimate non-market values. For example, the:

- travel-cost method uses recreation expenditure and travel time to impute the value people place on visiting a specific site (such as a national park)

- hedonic pricing method attempts to isolate the influence of non-market attributes (like proximity to parks or landfills) on the price of goods (such as houses).

The ability of revealed preference methods to produce valid non-market value estimates is widely accepted. However, there are many circumstances where these methods cannot provide the estimates needed for environmental policy analysis. Because they rely on values leaving a 'behavioural trace', they cannot be used to estimate so called 'non-use' values (for example, the value people derive from the existence of a species or ecosystem). The methods also focus on what has happened, which can limit their usefulness for valuing prospective changes. For example, the travel-cost method might be able to provide an estimate of the recreational value of an area of native forest, but not the change in value from a proposed program to eradicate pest plants and animals from the forest. More generally, the main limitation lies in the lack (or inadequacy) of data sets that contain traces of non-market values for environmental outcomes.

Stated preference methods

In principle, stated preference methods (including contingent valuation and choice modelling) could be used to estimate virtually all types of values, but their validity is more contentious. These survey-based methods typically impute values by asking people to make choices between policy options, in which better environmental outcomes are associated with higher costs (such as higher taxes and the loss of economic uses of environmental resources).

Ever since contingent valuation was used to estimate damages from the Exxon Valdez oil spill in Alaska in the early 1990s, there has been a lively and sometimes heated debate about the validity of stated preference methods among economists and others. In Australia, an estimate of the environmental cost of proposed mining near Kakadu National Park also proved contentious in the early 1990s.

More recent evidence is that stated preference estimates:

- are often broadly similar to revealed preference estimates

- have been found to be consistent with binding referendums on environmental policies

- often conform to predictions derived from economic theory (while there are exceptions, these can frequently be explained by either poor survey design or behavioural influences that can also affect market transactions).

This suggests that stated preference methods are able to provide valid estimates of non-market values for use in environmental policy analysis. However, there are many different elements that practitioners need to get right for stated preference surveys to produce meaningful results. One of the most important is that participants should be made to feel that their responses could influence outcomes that they care about (for example, that they would be required to pay the amount they state in order to achieve an improved environmental outcome). Much of the debate about stated preference surveys has been about their hypothetical nature, but there is now broad agreement that they can be designed to appear consequential and not purely hypothetical.

It is also crucial that surveys provide clear and specific information about the environmental outcomes that people are being asked to value. Outcomes should be expressed in terms of endpoints that people directly value and should align with the expected outcomes from proposed policies. People will often answer survey questions even if they do not understand or approve of the questions and so there is an important role for follow-up questions that can be used to filter out unreliable

responses. Knowledge about how to improve stated preference estimates has increased over the last 20 years and useful new tools have been developed.

How well stated preference methods perform can depend on how familiar respondents are with the environmental assets in question. For example, people surveyed at a recreation site about their willingness to pay to visit are likely to be able to provide well-informed answers based on their knowledge and feelings about the site, and possibly also knowledge about substitute sites they might prefer if the cost of visiting changed. By contrast, when people are asked about environmental assets that are relatively unfamiliar to them (and which they may never visit) they rely more on the information presented to them and may have to construct their preferences during the survey. While this can be done, insights from behavioural economics suggest that people are more likely to be prone to cognitive biases in such low-experience situations. For example, the focus of a survey on a particular environmental asset may cause people to elevate its significance relative to a situation where it was considered as one asset among many.

Two conclusions follow from this. First, survey design, including the information provided to respondents and techniques for weeding out unreliable answers, is of particular importance when valuing less familiar (or more complex) outcomes. Second, value estimates may be less accurate for unfamiliar outcomes, even with careful attention to survey design. Such problems are more likely to occur for non-use values and so stated preference methods may be less effective in estimating the very type of value for which other valuation methods cannot be used.

Benefit transfer

The evidence suggests that transferring value estimates from one site to another is likely to be very imprecise (and possibly misleading) unless there is a high degree of similarity between the 'study' and 'policy' contexts (in terms of the environmental features, policy outcomes and population characteristics). These seemingly obvious cautions are often not observed. A shortage of suitable primary studies in Australia is likely to mean that benefit transfer can only reliably be used in a limited range of circumstances. However, if even a very imprecise value estimate is potentially of use, benefit transfer may be worth considering even when the available primary studies are less than ideal. A strategic approach to conducting non-market valuation studies could be used to build up an evidence base that could support wider use of benefit transfer.

What role should non-market valuation play?

A finding that non-market valuation methods can provide estimates that are valid and reliable still leaves open questions about when and how they should be used in policy analysis. For example, non-market valuation studies can be expensive and in some cases this will outweigh their potential benefit. In deciding when the methods should be used it is necessary to compare them to potential alternative ways for factoring environmental outcomes into policy analysis.

Comparison with alternatives

Four main insights emerge from comparing non-market valuation with various alternatives, such as expert valuation, deliberative valuation and qualitative assessment of non-market outcomes within a cost–benefit analysis.

First, all of the alternatives to non-market valuation, including those that are commonly used, have major deficiencies. Some effectively ignore the trade-offs inherent in environmental policy, and most do not even attempt to factor the community's preferences into the analysis. None matches non-market valuation in providing objective and valid estimates of non-market environmental values. Accordingly, it is important to consider when non-market valuation is the best available approach, not whether it is ideal in all respects.

Second, the case for using non-market valuation is likely to be strongest where the financial or environmental stakes are high and there is potential for non-market outcomes to influence the choice of policy option. These conditions are most commonly found in regulatory contexts (such as deciding between different regulatory standards), but may also arise for major government investments in environmental improvement.

Third, conducting a cost–benefit analysis that describes, but does not value, non-market outcomes is likely to be a reasonable approach in some situations. In some cases, this approach is able to identify the policy option that maximises net benefits to the community. In others it assists by making trade-offs clear, but does not indicate whether differences in environmental outcomes tip the scales in favour of a particular option. This difficult judgment is left entirely to the decision maker.

Finally, considerable effort has been put into developing expert (usually science-led) approaches to environmental policy analysis. Sometimes these explicitly incorporate expert valuations of environmental outcomes. More commonly, they focus on how to achieve particular objectives or criteria, but this nonetheless can implicitly value outcomes. Because these approaches rely on values

that are not based on community preferences, they are likely to be a poor alternative to non-market valuation for evaluating trade-offs between environmental and other outcomes (such as reducing taxes). Scientists can inform the community about the consequences of different choices, but not necessarily which choice to make.

That said, expert approaches have considerable potential for improving the cost effectiveness of environmental policy, and so can have an important role to play. But expert approaches are not all equal. Those that are broadly consistent with cost–benefit analysis (such as the Investment Framework for Environmental Resources) offer advantages over those that are not (such as multi-criteria analysis).

Using non-market valuation

Non-market valuation should be used in combination with good practice policy principles, such as those set out in the Australian Government's *Best Practice Regulation Handbook*. For example, valuation should only be used in situations where a sound reason for considering government action (such as the existence of market failure) has been established. Where non-market valuation estimates are made they should generally be included in a cost–benefit analysis. The likely accuracy of all components of the analysis should be explained and sensitivity analysis used to demonstrate how the results change under alternative assumptions. It is important to describe the non-market outcomes (what the policy would achieve relative to what would have occurred in its absence) as well as providing their estimated value.

Cost–benefit analysis is an information aid to decision making, not a substitute for it. The analysis needs to be presented clearly to allow for proper scrutiny, including of the basis for non-market valuation estimates.

Realising the potential

There are several barriers to non-market valuation achieving its potential to improve environmental policy. One important barrier is that a cost–benefit framework is often not applied. Where this occurs, non-market valuation is unlikely to gain traction. Concepts such as sustainability and the precautionary principle understandably play a major role, and it is often thought that these are incompatible with applying a cost–benefit framework. But the extent of any incompatibility is unclear. For example, some interpretations of the precautionary principle are fully consistent with cost–benefit analysis, while others are not. Greater guidance on how to apply these concepts could help to resolve these issues.

There are also proactive steps that could be taken to realise more fully the potential of non-market valuation, including:

- paying greater attention to the quality of studies and developing a more widespread understanding of what constitutes a high-quality study

- better aligning the research effort into non-market valuation with policy needs, including building up a bank of value estimates to support benefit transfer

- developing greater knowledge about non-market valuation within relevant government agencies.

1 Introduction

Human life depends on the environment: on clean air and water, soil and the wide diversity of plants and animals that provide food, fibre and much else besides. For this reason, attempts to measure the total value of the environment are unlikely to be useful. In fact, it has been suggested they can only produce underestimates of infinity (Bateman et al. 2011).

Government decisions about the environment, however, involve smaller-scale trade-offs between environmental outcomes and other things that benefit the community. For example, investing in environmental improvements (such as cleaner rivers) takes resources that could have been used for other desirable purposes (such as funding for schools or hospitals). Similarly, allowing the use of an environmental asset (such as logging of a native forest) could put pressure on the habitat of a threatened species, but provide benefits (such as timber to build houses).

Valuing environmental outcomes in these types of situation, while difficult and sometimes contentious, may assist with making trade-offs in a more considered way. Dollar values are used, not to 'commodify nature', but rather to help decide whether having more of one good thing is preferable to having more of some other good thing in situations where a choice must be made.

Over the last few decades several 'non-market' valuation methods have been developed for this purpose, but to date they have not been widely used for policy analysis in Australia. This paper examines the potential for these methods to provide a better understanding of environmental trade-offs, and contribute to policy decisions that better reflect community preferences.

1.1   Understanding environmental values

There are different meanings of the word 'value'. It is used in this paper to refer to the monetary amount that reflects the worth of one good or service relative to others. This section looks at the various types of environmental values before focusing on non-market values.

Types of environmental values

People value the environment in a range of different ways. Classifying the different types of values is useful because it can help to make sure that no values are overlooked or double counted. Figure 1.1 presents a widely used classification system.

Figure 1.1　**Classification of environmental values**

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                        ┌──────────────────────┐
                        │ Total economic value │
                        └──────────────────────┘
                 ┌─────────────────┴─────────────────┐
                 ▼                                     ▼
        ┌─────────────────┐                  ┌─────────────────┐
        │    Use value    │                  │  Non-use value  │
        └─────────────────┘                  └─────────────────┘
          ┌──────┴──────┐                      ┌──────┴──────┐
          ▼             ▼                      ▼             ▼
 ┌────────────────┐ ┌────────────────┐ ┌──────────────────────┐ ┌────────────────┐
 │ Direct use value│ │Indirect use value│ │Altruism/bequest value│ │ Existence value│
 └────────────────┘ └────────────────┘ └──────────────────────┘ └────────────────┘
```
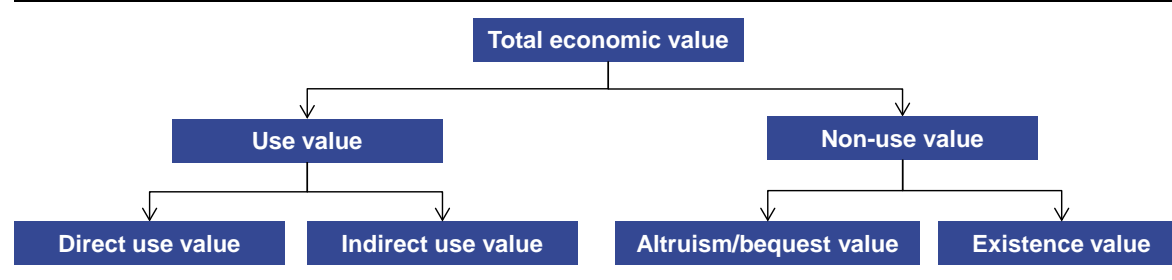
Direct use values encompass consumptive uses, such as for crops, livestock and fisheries, and non-consumptive uses, such as recreation. Indirect use values are the values people hold for the services provided by species and ecosystems. Examples include pest control, pollination and water cycling.

Altruism value is a type of non-use value (sometimes called passive-use value) that derives from the satisfaction of knowing that other people have access to nature's benefits. Bequest value is similar, but relates to future generations. Existence value relates to the satisfaction of knowing that a species or ecosystem exists. Existence values may derive from altruism towards biodiversity and be associated with people's ethical position on the importance of other species. While some analysts have questioned the relevance of some aspects of non-use values to community welfare (Diamond and Hausman 1994), there is now widespread acceptance that non-use values are a legitimate component of total economic value (Arrow et al. 1993; Atkinson, Bateman and Mourato 2012; Kumar 2010).

These categories help to illustrate that there is an important distinction between value and price. A crucial issue for this paper is that non-use values may be considerable, but they are generally unpriced in markets (that is, they effectively have a price of zero, which does not reflect their value). Even where prices exist, as they sometimes do for environmental goods and services that have a direct use value, some people may be willing to pay more than this price (for example, for access to clean water). In both cases, there is a gap between value (as expressed by

willingness to pay) and price. This gap is sometimes referred to as consumer surplus.[1]

Another important distinction is that between the value of environmental assets and environmental services. For example, a wetland might provide a range of services that are valued, including recreation opportunities, habitat for threatened species, and regulation of water flows that help prevent downstream flooding. The wetland itself can be thought of as an asset with a value that depends on the future services it can provide.

Thinking in terms of environmental assets raises important valuation issues. One of these is that an ecosystem may provide services in the future that we are presently unaware of. An often cited example is the potential medicinal value that might be derived from a plant species. Such possibilities mean that there may be value in preventing irreversible damage to an environmental asset. This is known as an option value. Option values are not a separate element of total economic value; rather, they may make up a component of other types of value (Hanley and Barbier 2009). So the option value associated with possible future medicines would be a component of direct use value.[2]

What are non-market values?

Another way of classifying environmental values is between market values and non-market values.

The environment plays an important role in supporting the production of goods and services that are sold in markets. For example, soil, pollinating insects and other environmental inputs support food production. Accordingly, aspects of the environment give rise to 'market values'. Some environmental assets, such as land, and services, such as honeybee pollination, are traded in markets and so have an explicit price that reflects their market exchange value. The value of others, such as rainfall or native pollinators, can be estimated based on the contribution they make to market production using production function methods (whereby the value of

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[1] Similarly, the price in excess of the cost of supply is known as producer surplus. In a competitive market with no distortions, the price and the quantity supplied/consumed are such that consumer surplus plus producer surplus is maximised. At this equilibrium, the price equals the value obtained from the last unit consumed (marginal benefit), which also equals the marginal cost of supply.

[2] Where medicines are made possible by knowledge gleaned from the analysis of plants this would give rise to indirect use values.

environmental inputs can be inferred from the contribution they make to the value of the marketed final product).

For example, where clearing native vegetation is expected to lead to greater salinity on nearby agricultural land, hydrologists, agronomists and agricultural economists can estimate the value of the loss of agricultural production. The greatest source of error in making such estimates often arises from incomplete scientific understanding of the impact of environmental changes on production. By contrast, the valuation of the change in production is often reasonably straightforward (at least for small changes), given the existence of market prices (for example, for agricultural produce or agricultural land of different qualities).

The environment, however, also contributes to people's wellbeing in ways that do not directly involve markets. Many people enjoy spending time in natural settings, or derive satisfaction from the existence of wilderness areas or natural ecosystems. This means that people value aspects of the environment, in the sense that they would be willing to give up something else of value to continue to enjoy them, or to ensure they are available for future generations. Economists use the term 'non-market' to denote these types of values. Referring back to figure 1.1, some use values are non-market values (for example, recreation often is) and non-use values are almost always non-market values.

There are a few things worth noting about non-market values. First, they cannot be estimated by any direct reference to market prices, which makes valuation much harder.

Second, there is not always a behavioural trace that is suggestive of these values. For example, if someone often goes bushwalking in the Ku-ring-gai Chase National Park it may be possible to infer the recreational value they place on the park by observing the amount of money and time they devote to visiting it. However, if someone values the existence of Ningaloo Reef but does not visit it they might not exhibit any behaviour from which this value could be inferred. It follows that scientific or other experts may have no genuine capacity to estimate some types of non-market values unless they ask people about them. Or as one analyst put it, the relevant experts are the public itself (Hanemann 1994).

Third, non-market values, as usually conceived by economists, are a human-centred construct. Some commentators raise ethical objections to valuing the environment in this way, as they argue that the environment has 'intrinsic value' that is unrelated to human preferences (Spash 1997). Full understanding of the concept of non-market value may remove some of these objections. This is because where people's ethics lead them to be willing to altruistically forgo some of their resources

for the sake of environmental improvement, this does get counted as non-market value.

However, those who believe decisions concerning the environment should be settled through debating ethical perspectives, rather than taking each individual's preferences as given, are likely to remain opposed to economic approaches to valuation. That said, it is not clear how the concept of intrinsic value could be satisfactorily applied. One problem is that once one environmental asset is assigned intrinsic value, it is difficult to see how unavoidable trade-offs with other environmental, cultural or social assets that are also afforded intrinsic value could be resolved.

Finally, while many non-market values relate to the environment and these are the focus of this paper, non-market values arise in other areas as well. For example, people value good health and shorter travel times.

1.2 Why do non-market values matter for policy?

There are many cases where environmental non-market values are relevant to policy analysis — table 1.1 provides some examples. In most of these, there are conflicting uses of the environment, which give rise to a trade-off between market outcomes and non-market outcomes. Valuing outcomes can be useful to inform decisions about these trade-offs.

Non-market values are often associated with 'market failures', such as the existence of public goods or negative externalities (box 1.1). In these cases, markets do not adequately take account of the outcomes — both market and non-market — that people value. For example, a factory might pollute a river because it bears no cost from doing so (a negative externality) and this could affect recreational users of the river (a decrease in non-market values) and production by irrigators (a decrease in market values).

Table 1.1 **Some policy areas where non-market values are relevant[a]**

Policy area	Some of the market and non-market values at stake
Air quality	Air pollution, particularly in cities, can cause irritation, illness and loss of visual amenity. Policies that reduce pollution can reduce these negative effects, thereby producing non-market benefits. The trade-off is that these policies can also impose market costs, such as those associated with fitting pollution control devices, switching to more expensive fuel sources and banning particular industries from urban areas.
Water quality	Stormwater from agricultural and urban areas, and water discharged from factories and treatment plants, can pollute rivers, which can degrade valued ecosystems and reduce recreational enjoyment of them. There is a trade-off between the market costs associated with meeting more stringent water quality targets (such as the cost of upgrading water filtration systems and funding government programs to improve water quality) and the non-market benefits from less polluted water bodies.
Water allocation	Choices must be made about the proportion of water resources to allocate to consumptive uses (such as irrigation and household use) and to environmental uses (such as flushing pollutants or maintaining the health of wetlands). There is a trade-off between the market value of consumptive uses and the non-market value of environmental uses.
Mining	Mining can require native vegetation to be cleared, affect the health of wetlands through the extraction of groundwater, cause land subsidence and have amenity impacts on local communities. Increasing the stringency of mining regulations can reduce these non-market costs (to zero, if a mine is disallowed). The trade-off is that this can also reduce the profits of mining companies, the incomes of mining workers, and the flow of royalties and taxes to governments.
Native forest logging	Logging of native forests can cause loss of biodiversity and reduced recreational enjoyment. Therefore, there can be a non-market benefit from banning (or limiting) logging, but this comes at the cost of not having access to logs that are valued by wood processing facilities (and ultimately consumers). Both the non-market costs and market benefits of logging vary markedly from one area of forest to another, meaning that it may be sensible to ban logging in some forests but not others.
Waste management	Improper disposal of waste can have negative effects on human health, visual amenity and ecosystems. Reducing these effects can have non-market benefits, but also market costs associated with upgrading landfills, anti-litter programs and recycling.

a In most of these examples there can also be market benefits associated with 'pro-environment' actions. For example, improving water quality can reduce treatment costs for downstream consumptive users. There may also be non-market benefits from 'pro-development' actions. For example, maintaining or increasing the water allocations to irrigators can reduce non-market social costs from declining employment in irrigation areas.

> **Box 1.1   Some types of market failures**
>
> **Public goods** exist where provision for one person means the product is available to all people at no additional cost. Public goods are non-rivalrous (consumption by one person will not diminish consumption by others) and non-excludable (it is difficult to exclude anyone from benefiting from the good). Some examples include the conservation of biodiversity, flood-control dams, national defence and street lights. Given that exclusion would be physically impossible or economically infeasible, the private market is unlikely to provide sufficient quantities of these goods. The nature of public goods makes it difficult to assess the extent of demand for them, while the marginal cost of supply beyond the first consumer is zero. Hence the optimal supply of public goods is fraught. Moreover, even if ideal supply is known, non-excludability leaves no incentive for private provision.
>
> **Externalities (or spillovers)** occur where an activity or transaction has positive (benefits) or negative (costs) effects on the welfare of others who are not direct parties to the transaction. An example of a positive externality is disease immunisation, which protects the individual, but also lowers the general risk of disease for everyone. Examples of negative externalities include pollution and large buildings that block sunlight to their neighbours.
>
> These market failures (or the lack of a solution) arise from problems with property rights. For example, if the right to clean air was adequately defined and defended, polluters and those affected by pollution could negotiate efficient outcomes, provided the costs of negotiation (or 'transaction costs') were low.
>
> *Sources*: Bennett (2012); PC (2006).

Taking account of non-market values

Australian governments have developed processes and guidance material with the aim of ensuring that all expected outcomes (or impacts) of policy options to address market failures (or other problems) are considered. For example, a regulation impact statement (RIS) is mandatory for all decisions made by the Australian Government and its agencies that are likely to have a regulatory impact on business or the not-for-profit sector (Australian Government 2013). The *Best Practice Regulation Handbook* provides guidance on preparing a RIS, including the need to assess the market and non-market costs and benefits of policy options, ideally through a formal cost–benefit analysis. The aim is to assist decision makers to maximise net benefits (benefits minus costs) to the community, although equity and other considerations can also influence the choice of policy option (box 1.2).

| Box 1.2 | **Cost–benefit analysis** |

Cost–benefit analysis (CBA) is a method that can be used to evaluate whether an investment project or a policy makes the community better off overall compared to the status quo. That is, whether it is expected to produce a 'net benefit', and if so, the extent to which benefits exceed costs. This evaluation should be broad, taking into account economic, social and environmental outcomes.

In CBA, benefits are valued according to the willingness of individuals to pay for them, which is often more than they would actually need to pay. For example, the price of the water supplied to a household is often less than willingness to pay.

Similarly, costs are valued according to the willingness of others to pay for the resources involved and, therefore, reflect the best alternative forgone (this is called 'opportunity cost'). To illustrate, while a painter who paints their own house does not have to pay for labour, their labour still has an opportunity cost as they could have been doing something else in the time taken.

A financial analysis only takes into account the market price (and total revenue) of supplying the service relative to its cost of production. A CBA takes into account the value of the service to consumers beyond the price paid, and the cost beyond what is paid to the factors of production. A CBA should also take into account any externalities — other costs and benefits — that fall on people outside those involved in the transaction.

The costs and benefits of projects and policies often accrue over a considerable length of time. To take account of people's preference to receive benefits now rather than later, future values are discounted to a present value.

Usually, costs and benefits are aggregated across individuals without regard to winners and losers from the policy. Governments and others may be concerned about how particular groups, such as low-income households or rural communities, are affected, and so may not think it appropriate to base decisions purely on a cost–benefit rule. There may also be concerns about impacts on future generations, particularly for policies that seek to promote sustainability. Such distributional (or equity) concerns can be addressed in CBA by presenting disaggregated results showing the effects on particular groups. Decision makers can then make judgments about the need for any particular response to distributional issues.

Further information about CBA can be found in Commonwealth of Australia (2006) and Boardman et al. (2010). Pearce (1998) and Pearce, Atkinson and Mourato (2006) deal specifically with CBA and the environment.

There are good reasons for the emphasis on applying a cost–benefit framework that is found in publications such as the Australian Government's *Best Practice Regulation Handbook* and state equivalents, such as the *Victorian Guide to Regulation*. This framework provides a means for weighing up the gains and losses from policy proposals in a way that is consistent with the concepts of welfare

economics. In essence it involves extending the approach that individuals take to making economic decisions (such as which products to buy and how much to work) to the community-wide level. However, when a cost–benefit framework is applied at the community-wide level it needs to take into account the (sometimes conflicting) wishes and wants of the people that make up the community, and recognise that the actions of one group can have impacts on the wellbeing of others (Dobes and Bennett 2009).

There are alternative frameworks that can be applied, such as subjective weighting of outcomes, as applied in multi-criteria analysis. While such approaches can sidestep the difficult issue of valuing non-market outcomes, this can greatly compromise the quality of the analysis. For example, while multi-criteria analysis is often used to avoid valuation, it can implicitly assign values that bear no relationship to community preferences. Another approach is to identify certain environmental outcomes that are to be achieved regardless of the cost. There are many instances where this is likely to serve the community poorly because it ignores the trade-offs that must be made. These issues are discussed in more detail in chapter 3.

While cost–benefit analysis provides a framework for considering trade-offs between environmental and other outcomes, the task becomes more difficult when there are policy outcomes that have non-market values. Scientific and market data can be used to identify the most cost effective way of achieving particular environmental outcomes. However, if the value of the non-market outcomes is not included in the analysis there is no basis to conclude that the policy option chosen maximises net benefits to the community. This can lead to important environmental outcomes being ignored (effectively assigning a zero value to these policy impacts) or regulatory bans being placed on particular activities to achieve an environmental outcome that the community may not value highly (effectively assigning an infinite value).

Figure 1.2 illustrates this situation using a hypothetical example. The outcome without government action is shown as P0. Cost-effectiveness analysis can be used to estimate the minimum feasible costs of achieving increasingly better environmental outcomes (P1, P2 and P3). These points map out a minimum cost curve. The policy question is which point on the curve is optimal from the point of view of the community. At least notionally, we can think in terms of there being a latent benefits curve (shown as a dashed line), determined by the values placed on different environmental outcomes by the individuals in the community.[3] Net

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[3] For simplicity, in this example all the environmental benefits from the policies relate to non-market values.

benefits are maximised at the point where the vertical distance between benefits and costs — the net benefit — is greatest.

In figure 1.2, P1 has the highest net benefit. By contrast, P3, the option that produces the best environmental outcome, is worse (in net benefit terms) than the other policy options and also worse than doing nothing. In other words, the value created for the community by P3 is less than what it costs. Of course, in some other cases the policy option delivering the best environmental outcomes will be the one with the highest net benefits.

In the absence of non-market value estimates, consultation can provide some information about stakeholder views, but the strength of people's preferences is difficult to gauge. Expert scientific opinion can be sought, but there is no reason to suppose that the trade-offs scientists would choose to make reflect those of the general public. It is easy for debate to become polarised between those with strong interests in the decision. Ultimately, judgment must be exercised and a decision made, even though very little may be known about the actual position of the benefits curve.

Figure 1.2   **Total costs and benefits of a hypothetical policy**
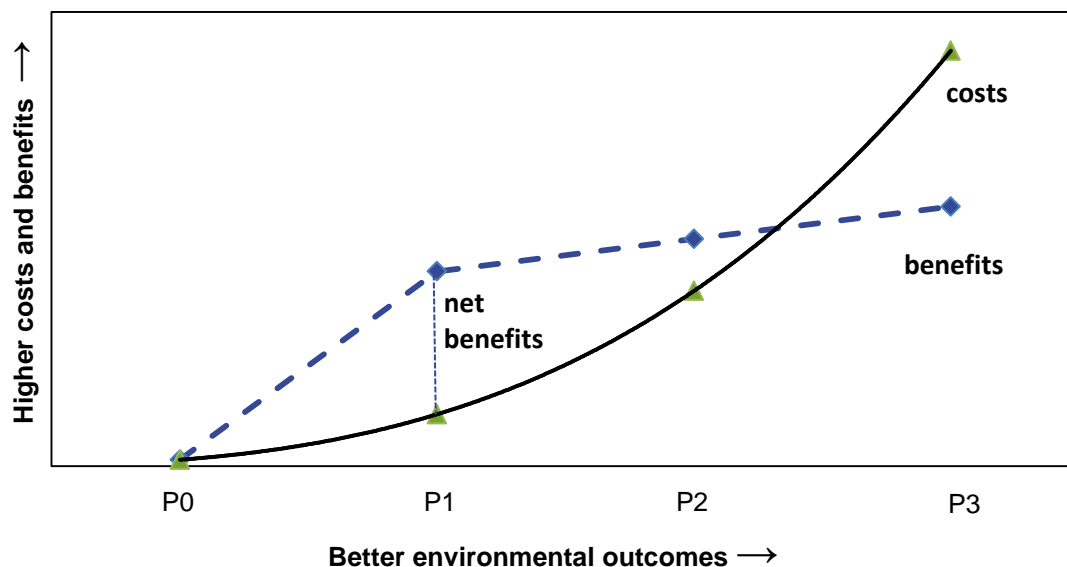
If reliable assessments of the likely non-market outcomes of a policy are available, several methods can be used to quantify (in dollar terms) their value to the community. The claim made by practitioners of these non-market valuation methods is that they can trace out the missing benefits curve, or at least give reasonably reliable estimates of some points along it. Where this is true, it would enable decision makers to be much better informed about the costs and benefits of

policy proposals. Importantly, non-market valuation can incorporate the preferences of a representative sample of the population into policy analysis, which could reduce the potential for decisions to be unduly influenced by vested interests. Even quite imprecise estimates, or upper or lower bound estimates, would be of use in many cases. So this is a claim well worth examining.

1.3   About this paper

There is a disconnect between the seemingly large potential that non-market valuation has to improve environmental policy and its limited use to date. Either the potential is illusory because the methods cannot reliably do what is claimed, or the reluctance to use them is a lost opportunity that should be rethought. This paper examines these issues.

Specifically, this paper:

- assesses the validity and reliability of various non-market valuation methods and of benefit transfer (where valuations from secondary sources are used)

- reviews the case for using non-market valuation — in particular, stated preference techniques — in the analysis of environmental policy (and other policies that have environmental consequences)

- offers suggestions on how best use can be made of non-market valuation in developing such policy.

The paper is about valuation for policy analysis (broadly defined to include analysis used in developing regulations, and making regulatory and government investment decisions) and not for other purposes, such as environmental accounting or legal compensation.

The paper also does not consider the institutional arrangements within which policies are determined, but it is acknowledged that these are important. A premise underlying this paper (and much of the literature on which it draws) is that government institutions could use estimates of non-market values to devise policies that improve overall community welfare. Whether this is likely to occur depends not only on the validity of the estimates, but also on the design of institutions. For example, a government agency with poor accountability might pursue ends favoured by its own staff rather than seek to improve community welfare. There is a risk that such an institution would use non-market valuation in a selective or biased way to justify predetermined decisions.

A further institutional question is the role that private organisations should play in the provision of environmental goods. One means through which this could occur is governments creating markets (such as pollutant trading schemes or conservation tenders). An important point to recognise is that government-created markets are a complement, not an alternative, to non-market and other valuation. Unless valuation methods are used in the design of such a market, there is little reason to expect that the resulting price will reflect the value that the community places on the environmental outcomes achieved. This is because, without valuation, there is no reliable way to calibrate the demand side of the market to community preferences. In other words, government-created markets can put a price on environmental outcomes, but non-market valuation may be able to help ensure that this is close to the 'right' price.

2 When can non-market valuation provide good estimates?

Key points

- Stated preference methods ask survey respondents about the choice they would make over environmental outcomes that come with a price.
  - Contingent valuation can value an outcome as a whole. It usually involves asking people whether or not they would pay a set amount of money for the outcome.
  - Choice modelling estimates implicit prices for the attributes of a non-market outcome. This is done by asking people to choose between options that are described by different levels of attributes and any costs they would have to pay.
- Revealed preference methods use observations of behaviour to impute a value for non-market outcomes that are linked to market goods. These methods include:
  - travel-cost models that use recreation expenditure and travel time to estimate the value people place on visiting a specific site
  - hedonic pricing techniques that decompose the price of a multi-attribute good (such as a house) to value individual non-market attributes (such as environmental amenity).
- The validity of stated preference methods has been widely debated. On balance, the evidence suggests that the methods are able to provide valid and reliable estimates. However, the techniques may provide less reliable estimates when people have a low understanding of, or familiarity with, the good being valued.
- Revealed preference methods are grounded in actual behaviour and can provide reliable estimates of non-market environmental values when suitable data are available. However, the methods cannot take account of non-use values that have not been reflected in observed behaviour.
- Non-market valuation methods have the potential to provide a good indication of the value the community places on non-market outcomes.
  - A well-designed stated preference survey has several key features. Most importantly, the non-market environmental outcome and policy context are clearly explained, outcomes are expressed in terms that are relevant to participants, and participants feel that their responses will have consequence.
  - Revealed preference methods require good data on all relevant factors, and are most informative when key assumptions are clearly set out and tested.
- Benefit transfer involves applying available estimates (from other studies) to a new context. Accuracy is likely to be low unless the primary studies are of high quality and relate to similar environmental and policy contexts.

Non-market values can be quantified in several ways through the use of stated and revealed preference methods. There is a substantial academic and applied literature documenting these methods and examining whether they can provide valid and reliable estimates. This chapter outlines some of the main approaches and reports on the evidence on their validity and reliability. It also sets out criteria to help assess the quality of non-market valuation studies.

2.1   Methods for valuing non-market outcomes

There are two broad ways to estimate the monetary value of a non-market outcome. Stated preference methods use surveys to estimate how much money people would be willing to pay to obtain a non-market outcome, such as a specific environmental improvement due to a policy. Revealed preference methods analyse observed behaviour to impute the dollar value that people place on non-market outcomes such as recreation or amenity. 'Benefit transfer' is not a valuation method in itself, but rather a technique for applying available estimates of non-market values to new policy contexts.

Stated preference

Stated preference methods involve asking people how much they value a particular non-market outcome. This is done by surveying a sample of people that is considered to be representative of the population. There are two main approaches (box 2.1).

- Contingent valuation involves asking people to make choices about environmental outcomes and payments that can be used to estimate how much they are willing to pay for a non-market outcome to be provided. This outcome, or 'good', is valued as a whole (for example, the amount of money people would be willing to forgo through additional taxes for improvements in vegetation along a river). Typically, people are asked whether or not they would be willing to pay a set amount of money for the environmental outcome to occur.

- Choice modelling (sometimes called choice experiments) involves offering people choices between different options that are made up of sets of attributes or characteristics that describe a policy outcome. For example, attributes might indicate numbers of birds and fish, an area of vegetation, and the cost to the individual or their household. 'Implicit prices' are then estimated for each attribute, reflecting average willingness to pay for an additional unit. The value placed on a particular policy option is the sum of the value of its attributes (the implicit price multiplied by the change in the attribute).

These methods typically provide average per-person or per-household estimates for the survey respondents, which can be extrapolated to the wider population to provide an indication of the total non-market benefits or costs of a policy option. This requires making assumptions about the extent of the population that will be affected by the policy change, and whether people who chose not to respond to the survey would also value the outcomes.

Box 2.1   What do stated preference methods do?

Contingent valuation and choice modelling both use surveys to estimate how much individuals are willing to pay for a non-market good. Participants are typically asked to make selections from a set of alternatives ('discrete choices'). Both methods use statistical models, based on random utility theory, to analyse survey data. This includes estimating average willingness to pay for non-market outcomes or specific attributes, and examining how willingness to pay is influenced by income, attitudes or other factors (such as age, gender and education).

Contingent valuation uses surveys to estimate the highest amount that people would be willing to pay for a non-market 'good' (which may be a single outcome or a complex set of outcomes). When this method was first used, surveys typically asked people to simply state their maximum willingness to pay. It has since become more common to present people with a set amount of money and ask whether or not they would be willing to pay that amount for the non-market outcome to be achieved (this could be an annual payment or one-off amount). The amount is varied across participants in a way that allows statistical models to be used to calculate average willingness to pay. Another approach involves presenting participants with 'payment cards' and asking them to select a maximum dollar amount from a list.

Choice modelling is a more sophisticated technique that was originally developed by marketing researchers, partly to overcome some of the drawbacks with contingent valuation. Individuals are asked to choose their most preferred option from a set of alternatives, each of which consists of a bundle of attributes that comprise the non-market outcome (or, in some cases, asked to rank or rate the options). One of the attributes is the cost to the survey participant, and each choice set contains an option representing the status quo (no policy change). By varying the levels of the attributes and presenting people with several choice sets, statistical methods can be used to quantify the trade-offs that people make between attributes (including implicit prices).

Stated preference methods are built upon several key assumptions. One is that people know how much they would be willing to pay (in terms of forgone income) for higher levels of a non-market good, and that this is constrained by their wealth and preferences to consume market goods. Another assumption is that people answer the survey questions honestly and rationally with these constraints in mind. Like other economic methods, it is also assumed that people are best able to know their own preferences.

Sources: Bateman et al. (2002); Hanley and Barbier (2009); Whitehead and Blomquist (2006).

Other methods have also been developed, such as using life-satisfaction survey data to value air quality or local amenity (box 2.2). Such techniques are not widely used, and are not discussed further in this paper.

Box 2.2   **Valuing non-market outcomes with life-satisfaction surveys**

Surveys of life satisfaction or subjective wellbeing have been used to estimate monetary values for non-market outcomes. This involves using econometric techniques to estimate the relationship between environmental factors (such as air or water quality) and the level of life satisfaction or wellbeing that people report. The relationship between income and wellbeing is also examined, allowing the analyst to quantify the trade-off that people implicitly make between income and the environmental outcome of interest.

The approach has been used in several countries to value water pollution, noise, natural hazards and air quality (Welsch and Kühling 2009). For example, Luechinger (2009) combined life-satisfaction survey data with air-quality observations to estimate how much German households are implicitly willing to pay to reduce concentrations of sulphur dioxide. Ambrey and Fleming (2011) estimated how much households in south-east Queensland are implicitly willing to pay for increases in scenic amenity.

This is a relatively new field of research. As wellbeing surveys are deployed more widely, there may be opportunities to examine how the community's values for non-market outcomes change over time or are influenced by distributional outcomes. However, the approach has limitations. General surveys about wellbeing may not be well suited to providing information on the values associated with a specific environmental policy option.

Stated preference methods are based on the notion that there is some amount of market goods and services (which people buy with their income) that people would be willing to trade off so they can benefit from a non-market good (which might be provided by governments). This is often measured in terms of *willingness to pay* for a non-market outcome, although the methods have also been used to assess how much compensation people would be *willing to accept* to give up a non-market good they already benefit from.

The use of surveys allows a wide range of non-market outcomes to be valued, capturing both use and non-use values (chapter 1). This gives stated preference methods the flexibility to evaluate potential policy outcomes for which there is little historical experience. Choice modelling, in particular, may be useful when a range of policy options, with different environmental outcomes, are being compared. The downside with stated preference studies is that they require significant effort, time and resources to be done well (section 2.3), and the validity of the methods is not universally accepted (section 2.2).

Stated preference studies have influenced environmental policy in Australia in several cases (appendix B). The methods have been more widely used in a policy context in the United States (typically in areas concerning outdoor recreation and air and water quality) and the United Kingdom. Stated preference methods have also been used in a number of countries to value non-market policy outcomes relating to health, transport and water provision.

Revealed preference

Revealed preference methods use data on people's behaviour to examine the trade-offs they make between money (or market goods) and non-market goods, such as recreation, amenity or improved health outcomes. There are two widely used approaches (box 2.3).

- The travel-cost method imputes the value that people place on visiting a recreation site by examining how much they spend to visit (including costs of transport, accommodation and park entry) and the cost of their time. These data are used to estimate the consumer surplus that people derive from visiting — a measure of the non-market benefit less the costs they incur.

- Hedonic pricing deconstructs the price of market goods that are influenced by non-market outcomes. It involves estimating implicit prices for a number of characteristics that make up the good (in the case of housing, these could be the number of rooms, bushland views or proximity to a landfill). The method has often been used to estimate environmental amenity values by analysing house prices. It has also been used to estimate the value of a statistical life by analysing wages across jobs with different levels of risk.

Box 2.3 **What do revealed preference methods do?**

The **travel-cost method** uses the 'price' (or cost) that people pay to travel to a particular site (such as a national park) to estimate the value they obtain from visiting. Surveys are used to collect data on the costs people incur, and these data are used to estimate a 'trip generation function' that relates travel costs to visit rates (visits per person or visits from a particular region, depending on the model used). A demand curve is then constructed using several assumptions, including that people would respond to the cost of travelling in the same way that they would respond to a site entry fee, and that the marginal (highest-cost) visitor derives no benefit from visiting in excess of the cost they incur. The demand curve is used to estimate the amount of consumer surplus associated with visiting the site, or to examine how visit rates and consumer surplus might change if entry fees were increased.

(Continued next page)

> **Box 2.3** (continued)
>
> Several assumptions are often made in applying the travel-cost model. One relates to the cost associated with travel time, which is generally not observed. Some studies use a fixed fraction of the wage rate, while others omit time costs from the analysis. Another complication is that people might travel for multiple reasons (such as to visit friends or other recreational sites), making it difficult to attribute costs to the site of interest. Some researchers do this based on the proportion of trip time spent at that site, while others use multi-site models that allow choices between recreation sites to be modelled explicitly, taking into account the fact that some sites may be substitutes.
>
> **Hedonic pricing** exploits the fact that some market goods comprise a bundle of attributes that include non-market elements. Most environmental applications use regression analysis to decompose house prices into the contributions that come from key characteristics, including house features (such as size or number of bathrooms), location (such as proximity to schools) and non-market environmental attributes (such as air quality or local amenity). This provides estimates of the implicit 'price' of each attribute, which indicates how much house buyers would be willing to pay for one additional unit of the attribute. Welfare measures such as consumer surplus and willingness to pay for a larger change in the attribute have rarely been estimated because of statistical complications and the strength of assumptions required.
>
> The hedonic pricing method is based on the theory that housing attributes have implicit prices and house buyers seek out higher or lower levels of a particular attribute such that the implicit price equals their marginal willingness to pay. Several assumptions are required to estimate these implicit prices. One is that all attributes are fully capitalised into house prices. Another is that house buyers are fully aware of the environmental attributes and weigh these up against the prices of all available houses in the market.
>
> *Sources*: Bateman (1993); Hanley and Barbier (2009); Randall (1994).

Other revealed preference methods have also been used, but less widely (and are not discussed further in this paper).

- The averting-behaviour (or avoided-cost) method infers the value that people place on non-market outcomes by examining what they pay to avoid or mitigate negative impacts. However, this method has been criticised for using price to proxy for economic surplus, and because it can understate non-market values (if the averting behaviour cannot fully offset non-market costs) or overstate them (if there are offsetting benefits that arise from the behaviour). For example, the amount of money that people spend on double glazing windows could proxy for the costs of traffic noise, but this may not be a reliable proxy if the double glazing does not fully mitigate the noise or if people also double glaze to save on heating costs (Pearce, Atkinson and Mourato 2006).

- The travel-cost and hedonic pricing methods have been extended to draw on stated preference data. This includes the analysis of stated and revealed

preference data in a single statistical model, as well as 'contingent behaviour' methods that survey people about future travel or house purchases then analyse the data using travel-cost or hedonic pricing models (Hanley and Barbier 2009).

Revealed preference methods cannot be used in every case where non-market values are needed for policy analysis. This is because these methods:

- can only be used where the value people place on a non-market outcome can be deduced from their behaviour — this generally rules out using the methods to quantify non-use values

- often require data to be collected for a large number of transactions, in which there is sufficient variation of the non-market characteristic of interest

- reflect the total value that people place on a non-market outcome in their actual behaviour, which can limit the usefulness of revealed preference methods to value future policy changes (especially where the changes go beyond past experience).

In the environmental area, revealed preference methods have mostly been used to value outdoor recreation and housing amenity (Hanley and Barbier 2009). Hedonic pricing has also been used examine the value placed on different aspects of a workplace environment by comparing the wages of jobs with different characteristics (OECD 2012). There are few recent instances where the methods have had a direct influence on environmental policy in Australia (appendix B).

Benefit transfer

Non-market outcomes can also be valued by drawing on estimates from available stated or revealed preference studies through benefit transfer. As a new primary study can be costly and time consuming, benefit transfer can provide considerable savings. However, it requires comparable estimates to be available, in terms of similar environmental goods, the extent of the policy change and the populations affected. For example, the non-market costs of limiting recreational access to a river might be assessed by drawing on a travel-cost study for another river of similar size and proximity to population centres. Contingent valuation estimates of the value people place on improving the health of an ecologically significant wetland (which could encompass non-use values) might provide some guide to how the community would value improvements to a similar wetland elsewhere.

There are two main approaches to benefit transfer, each of which involves making assumptions about the similarity of the current policy context (where an

environmental policy decision needs to be made) and the past study context (where a study valued non-market outcomes).

- Unit transfer involves transferring an available estimate of willingness to pay to the policy context on the assumption that the value is likely to be similar to that in the study context.

- Function transfer involves modelling willingness to pay as a function of specific variables (such as the size of a wetland or proximity to a population centre), allowing estimates to be adjusted for differences in these characteristics. Sometimes this involves drawing on the results of multiple studies (meta-analysis) to identify factors that influence willingness to pay across studies.

Benefit transfer is the most common way in which non-market valuation has been incorporated into policy analysis.

2.2   Can estimates be valid and reliable?

Non-market valuation has been the subject of much debate by economists, policy makers and others. The lack of markets for non-market goods makes it difficult to assess how well the methods perform. Nevertheless, there are other ways to assess their validity (that is, the extent to which the methods can accurately value what they intend to value). This can involve comparing the estimates of one method with those derived from another, testing whether the estimates are consistent with the assumptions that underpin economic theory, or examining the effect of different assumptions in the analysis. Reliability can be tested by replicating studies.

There is a substantial amount of evidence on how well non-market valuation methods perform. The question is whether this evidence is sufficient to conclude that the methods are able to provide estimates that are valid and reliable enough to usefully contribute to policy analysis.

Stated preference

Stated preference methods have been highly contentious, especially when used to estimate non-use or existence values (box 2.4). Evidence on the validity and reliability of stated preference valuation methods is summarised below, with more detail provided in appendix C.

Box 2.4   **Stated preference methods have been contentious**

Debate about the validity of stated preference methods gained prominence following the use of contingent valuation to value the damage caused by the Exxon Valdez oil spill in Alaska's Prince William Sound in 1989. This study generated a lower-bound estimate of US$2.8 billion, associated almost entirely with non-use values (measured by asking a sample of people about their willingness to pay to avoid a similar incident) (Carson et al. 2003). The findings were widely scrutinised, with debate focusing on whether people have well-formed preferences over non-use environmental outcomes and, if so, whether these can be accurately elicited by a survey. Subsequently, the oil company Exxon paid over US$3 billion in damages and to fund restoration.

Following these controversies, the US National Oceanic and Atmospheric Administration set up a panel of prominent economists to study the efficacy of the contingent valuation method. The panel gave qualified support, concluding that 'contingent valuation studies can produce estimates reliable enough to be the starting point of a judicial process of damage assessment, including lost passive-use values' (Arrow et al. 1993, p. 4610). The panel also set out guidelines for the use of contingent valuation, which proved influential.

However, contingent valuation remained subject to strong criticism. Diamond and Hausman (1994, p. 62) argued that 'contingent valuation is a deeply flawed methodology for measuring non-use value, one that does not estimate what its proponents claim to be estimating'. More recently, Hausman (2012, p. 54) contended that 'despite all the positive-sounding talk about how great progress has been made in contingent valuation methods, recent studies by top experts continue to fail basic tests of plausibility'.

Others dispute these negative conclusions. Carson (2012, p. 40) claimed that 'contingent valuation done appropriately can provide a reliable basis for gauging what the public is willing to trade off to obtain well-defined public goods'. Kling, Phaneuf and Zhao (2012, pp. 21–22) examined evidence that has emerged to support the validity of stated preference methods, arguing that:

> The past two decades have seen the coming of age of experimental economics, new theoretical developments, accumulating insights from behavioral economics, and a general maturing of the non-market valuation literature. … Those who formulated their beliefs about contingent valuation two decades ago, whether positive or negative, should update their beliefs based on the research agenda that has unfolded.

Australia has experienced its own controversies over the use of these methods, most notably following the use of contingent valuation by the Resource Assessment Commission in 1990 to estimate the environmental costs from proposed mining at Coronation Hill, adjacent to Kakadu National Park (RAC 1991). These costs were estimated to be in the order of 60 times greater than the economic surplus from mining. The study was strongly criticised in the media and elsewhere. It has been suggested that the ensuing debate led to the study becoming discredited in the view of many Australian policy makers, and that subsequent application of contingent valuation did little to improve the method's standing in Australia (Bennett 1996).

Do the estimates match real payments?

A natural starting point is to compare stated preference estimates with other measures of value that are widely accepted as being valid. If the estimates align closely, this would provide evidence for the validity of the methods (this is often termed 'criterion validity'). Prices in competitive markets are the most widely accepted indicator of the economic value of a good. Other indicators include values derived from economic experiments and voting outcomes.

Market prices can only be compared to stated preference estimates for *private* goods (such as consumer products), since many non-market outcomes are *public* goods that lack a competitive market (or a market at all). Accordingly, some researchers have sought to test how well stated preference methods (especially contingent valuation) can value private goods, such as new products that are about to be brought to market. The intention is usually to test how well the methods perform in a context where the goods are relatively familiar to consumers, and where value estimates can be compared to demand curves derived from market data (taken to represent the true values).

The assumption has generally been that this is a relatively easy test compared to valuing environmental goods that are less familiar to survey participants, and for which market estimates of value are rarely available. Such tests using contingent valuation have often found that the stated preference estimates are somewhat higher than market-derived values (Carson and Groves 2007). These results led some analysts to conclude that stated preference estimates are invalid, while others have explored ways to 'calibrate' the estimates (for example, by halving them) (Diamond and Hausman 1994).

However, more recent developments in the theory of non-market valuation suggest a different interpretation. Carson and Groves (2007) argue that the results are due to the nature of private goods. Because survey participants are not compelled to purchase the good, they might act strategically by overstating their willingness to pay if they believe that this would encourage a new good to be made available on the market. Actual purchase decisions would be made later.

However, public goods are provided in a different context. The government can provide public goods (such as improvements in biodiversity) to all and compel everyone to pay (for example, through levies or changes to tax rates). If survey participants believe that they may be compelled to pay based on their responses (and consider the payment mechanism to be acceptable), they may have less incentive to answer strategically. It is therefore possible that stated preference

methods can provide valid estimates for public goods, but not necessarily for private goods, when people are asked about their willingness to pay.

Absent relevant evidence from real markets, researchers have turned to other tests of validity. One approach has been to use experiments based on constructed markets. In one type of experiment, a referendum-style vote is used to determine whether or not all participants receive a good for which they will all be made to pay a set amount. The results of this real payment mechanism (estimates of total willingness to pay) are then compared to results from a stated preference survey of the participants, conducted prior to the experiment. A common finding is that values are similar when participants feel that their survey responses would have consequences (by influencing outcomes that affect them) (Landry and List 2007; Vossler, Doyon and Rondeau 2012; Vossler and Evans 2009).

Another source of evidence comes from comparisons with voting outcomes. A referendum — for example, on whether an environmental program funded by increased taxation should be introduced — is generally considered to be 'incentive compatible'. That is, people that would prefer to pay the extra tax and have the program proceed have an incentive to vote yes (and vice versa for no votes). Therefore, such referendums provide an opportunity to test the validity of stated preference surveys that ask essentially the same question. Several researchers that have taken up this opportunity have found that referendum results tend to align well with results from an earlier survey on the same issue (Johnston 2006; Vossler and Kerkvliet 2003; Vossler and Watson 2013). As with experiments, the alignment is strongest when participants consider the survey to be consequential and are encouraged to answer honestly.

The evidence from experiments and referendums generally supports the validity of stated preference methods, but it is not definitive on its own. Experiments are often based on providing participants with a tangible good, and are not well suited to eliciting non-use values (such as for biodiversity or natural heritage). Referendums can only be held in particular circumstances, and voting has not been compulsory in cases where researchers compared the outcomes to stated preference estimates (mostly in the United States). This may bring into question the representativeness of the results. Accordingly, other sources of evidence are desirable to assess how well stated preference methods can perform.

Do the estimates align with revealed preference measures?

Comparisons have also been made between stated and revealed preference estimates (often termed 'convergent validity'). This can be done where there are sufficient data to allow both techniques to be applied, such as when valuing recreation or housing amenity (but generally not non-use values). Statistical analyses of the available literature (meta-analyses) have typically found that stated and revealed preference estimates are correlated and broadly similar in magnitude, with the stated preference estimates usually tending to be somewhat lower (Brander, Van Beukering and Cesar 2007; Carson et al. 1996).

However, the gap between revealed and stated preference estimates varies widely across studies. Some studies have found that the two sets of measures match closely (Grijalva et al. 2002), or that stated preference estimates are higher (Azevedo, Herriges and Kling 2003; Woodward and Wui 2001). Others have found closer convergence when steps are taken to improve the quality of estimates. For example, Rolfe and Dyack (2010) found that excluding 'uncertain' contingent valuation responses from their analysis led to convergence with travel-cost estimates. Loomis (2006) found convergence after controlling for multi-destination visitors in his travel-cost analysis.

Some differences may also be due to the way estimates are calculated. For example, travel-cost studies generally estimate the average surplus associated with visiting a site, whereas stated preference studies estimate the value of an additional or marginal unit of an environmental good. In addition, revealed preference estimates can be sensitive to assumptions made in the analysis and the quality of available data (discussed below).

Overall, there is evidence that stated preference estimates are often reasonably close to their revealed preference counterparts (for use values and where a good can be valued using both approaches). But this depends on how well each study is conducted and whether the same underlying values are being measured. The possibility that revealed preference estimates could be subject to errors in their construction means that stated preference estimates are not necessarily invalid when they do not align closely. The fact that estimates from both methods tend to be broadly similar and are correlated suggests that stated preference estimates are consistent with other measures of value.

However, this literature has focused almost exclusively on *use* values, for which revealed preference estimates can be derived. It says little about the validity of stated preference methods for estimating *non-use* values, for which corresponding revealed preference estimates are generally not available.

Are the estimates consistent with economic theory?

Another source of evidence relates to whether stated preference methods provide results that are consistent with the economic assumptions that underpin the methods (often termed 'construct validity'). The methods are based on welfare economics, which assumes that people have well-formed and stable preferences over outcomes (market or non-market) that are relevant to their wellbeing. Stated preference methods seek to discover these preferences based on how people respond to survey questions.

The predictions made by economic theory can be tested in stated preference data. If the methods can pass these tests, it would be evidence that the value estimates they provide can be consistent with those derived from competitive markets. Key testable predictions include that:

- people are willing to pay more for a greater quantity of a non-market good (such as for a larger environmental project)

- the underlying preferences people have over non-market outcomes do not depend on the survey instrument used to elicit them

- there is a close alignment of measures of willingness to pay and willingness to accept compensation.

Testing these predictions has been a key focus of the literature (appendix C).

Invariance to scale

There has been considerable debate about whether stated preference estimates respond plausibly to the scale of environmental goods. Critics of the methods have pointed to contingent valuation studies that found that willingness to pay did not increase significantly with the scale of the good. For example, Desvousges et al. (1992) found little difference in willingness to pay for preventing the death of 2000, 20 000 or 200 000 waterbirds. This led to claims that stated preference surveys do not measure willingness to pay for a specific non-market outcome but, rather, a 'warm glow' that reflects the moral satisfaction of supporting environmental causes generally (Diamond and Hausman 1994; Kahneman and Knetsch 1992).

However, many studies have found that estimates of willingness to pay are sensitive to the scale of the good described in the survey (for example, Carson 1997; Ojea and Loureiro 2011; Smith and Osborne 1996). Some instances of invariance to scale have been associated with poor survey design, such as an unclear description of the environmental good (Carson 1997) or when changes in low-level risks are not explained in a way that is tractable to participants (Corso, Hammitt and

Graham 2001). Economists have also pointed out that economic theory gives little indication of how much willingness to pay should increase with the scale of the good. While theory suggests that the *increase* in willingness to pay should fall as the level of the good gets larger, this increase could be very small after a particular level of the good (a threshold) has been obtained (Bateman 2011), or when the good is provided as part of a larger package of goods (discussed further below).

Sensitivity to the survey instrument

Researchers have found that stated preference estimates can be sensitive to the way a survey is designed. Small changes in the design or layout of a survey can have a large influence on the resulting estimates of willingness to pay. The evidence points to several patterns in how people respond to surveys, including that:

- estimates of willingness to pay tend to vary depending on the type of valuation question asked, with a single 'yes/no' contingent valuation question (asking whether or not people would pay a given amount) providing higher estimates than other question types (Carson and Groves 2007; Champ and Bishop 2006)

- estimates can be sensitive to the specificity and detail of information provided about the environmental outcome and broader environmental context (MacMillan, Hanley and Lienhoop 2006; Munro and Hanley 1999)

- the type of payment mechanism used (such as a compulsory levy versus a rise in existing taxes or consumer prices) can have a significant impact on willingness to pay, or can imply a very high or low discount rate (based on comparisons of one-off charges to annual payments) (Kovacs and Larson 2008; Rolfe and Brouwer 2011)

- people sometimes appear to 'anchor' responses to numbers seen earlier in a survey — especially when asked several valuation questions — and may answer 'yes' to questions even when they are uncertain (Day et al. 2012; Green et al. 1998; Loomis, Traynor and Brown 1999)

- willingness to pay for a good falls the later it is valued in a sequence of goods (Carson and Mitchell 1995; Clark and Friesen 2008).

Such findings have sometimes been interpreted as evidence that people do not have well-formed and stable preferences for the underlying non-market outcomes, and that stated preference methods do not provide valid estimates of the value people place on these outcomes (Diamond and Hausman 1994). However, much of the research cited above has involved closely examining how variations in survey design can influence the results (typically by using two versions of a survey, each for a separate sub-sample of people). Overall, the findings indicate that people

generally respond to a survey in a rational and predictable way, given the specific circumstances of the survey.

Strategic bias can explain why different kinds of valuation questions can give different results. When a survey is consequential (that is, participants believe their responses will affect policy decisions that they care about), participants may seek to answer in a way that influences policy decisions in their favour (Carson and Groves 2007). This could involve misrepresenting their true preferences. For example:

- a participant might overstate their willingness to pay if surveyed about a voluntary contribution (which they could then choose not to make once the good is provided), or select an option in a choice set that is not their most preferred because they believe that the outcomes could be provided at lower cost (based on alternatives in earlier choice sets)

- a participant might answer strategically if they do not consider the payment mechanism to be specific to their circumstances, such as may occur in jurisdictions where specific levies are rarely used and tax rates differ across taxpayers.

However, economists have identified ways to minimise scope for such strategic responses. Asking a single 'yes/no' valuation question can help to avoid biases arising from the type of question asked (Carson and Groves 2007). Ensuring that the payment mechanism is perceived as credible and applicable to each individual survey participant can further encourage honest responses.

The way that people respond to surveys will also depend on whether they have a good grasp on what they are being asked to value. The evidence suggests that people will answer survey questions even if they do not understand the questions or material provided. In the absence of clear and unambiguous information, they might make their own assumptions to fill in the gaps (Hanemann 1994; Johnston et al. 2012). This may be especially likely where the policy outcomes being described are not expressed in terms that are directly valued by participants, but are instead proxies for the ultimate environmental outcomes that they care about — in which case they may draw on prior knowledge or make erroneous assumptions to make the relevant connections (Collins 2011; Johnston et al. 2012). Estimates of willingness to pay can also be biased when some important elements of the policy outcome are not mentioned in the survey (such as social impacts or how the policy will be implemented) and participants respond based on their own understanding of what these elements would likely be (Johnston and Duke 2007).

Moreover, when faced with an unfamiliar context or decision, people may fall back on behavioural rules-of-thumb, such as anchoring responses to numbers seen previously in the survey or answering 'yes' even when uncertain (Bateman 2011). By making people focus on a particular issue, a survey could give them an exaggerated sense of its relative importance (Kahneman and Sugden 2005). Participants could also submit 'protest' responses if they disapprove of the questions being asked or disagree with the information provided. For example, there is evidence that responses are sensitive to whether participants consider the type of payment mechanism or distribution of costs to be 'fair' (Cai, Cameron and Gerdes 2010; Jorgensen and Syme 2000). Another possibility is that participants who are not convinced that the policy would be implemented in the way described modify their responses (for example, based on their perception of how likely it is that the outcome would be achieved).

These behavioural responses suggest that stated preference methods may be more likely to generate biased estimates when survey participants have low familiarity with the non-market good being valued (which may be more likely for non-use values), or when the good is not described in a way that they find credible or can easily relate to. In such cases, they may not have a prior sense of their willingness to pay and could construct their valuation of the good during the course of the survey, with little time for reflection (Bateman 2011).

However, there is evidence that these kinds of biases can be minimised through survey design. More specific and detailed information about the non-market outcome and its context (including visual aids) can reduce the need for participants to make their own assumptions (Blomquist and Whitehead 1998; Munro and Hanley 1999). The use of community focus groups and consultation with key stakeholders can ensure that the information is objective and credible, and the outcome is described in terms that are relevant to participants (section 2.3). Clearly marked practice questions can familiarise people with the exercise before the formal valuation question is asked (Bateman 2011), and asking detailed questions on discretionary expenditure can encourage people to pay more attention to their income and budget constraints (Li et al. 2005). Further, follow up questions can identify when participants are highly uncertain or did not understand the information provided, allowing their responses to be treated differently in the analysis (for example, by excluding them) (Loomis, Traynor and Brown 1999).

Finally, the context in which environmental outcomes are provided matters for the value that people place on them. This may explain why willingness to pay appears to fall depending on where a good is placed in a sequence, or whether people have considered the available substitutes. Economic theory predicts an 'embedding' or 'part–whole' effect, where the value of a good can be higher when it is provided on

its own rather than as a package of goods. The goods might be substitutes, reducing the benefit of providing an additional good, or the provision of a larger number of goods (funded through taxes) could reduce the income people have available for other uses (Carson, Flores and Hanemann 1998; Hoehn and Randall 1989).

This implies that the value people place on non-market outcomes depends on the available substitutes. People tend to value an environmental outcome more highly the scarcer it is thought to be (although they may not be fully aware of the availability of substitutes prior to completing a survey). It also suggests that valuing one non-market outcome in isolation will not be accurate when a specific policy change is associated with a set of non-market outcomes.

In sum, the sensitivity of estimates to changes in the survey instrument is largely consistent with economic and behavioural predictions. Moreover, estimates should be expected to vary with differences in survey instruments, to the extent that these differences alter how the environmental 'good' is perceived by participants.

Divergence between willingness to pay and accept

Economic theory predicts that an individual's willingness to pay to obtain a good should be very similar to their willingness to accept compensation to give up the good. However, many studies have found that stated preference methods provide estimates of willingness to accept that are substantially higher than estimates of willingness to pay (Horowitz and McConnell 2002). Critics have suggested that this indicates that neither estimate is valid (Hausman 2012).

There are several explanations for the divergence. Behavioural economics predicts an 'endowment effect', where people place a greater value on a good because they have a property right over it (Kling, Phaneuf and Zhao 2012). This interpretation is supported by experimental and market data, where a similar divergence has been observed when people are faced with an unfamiliar situation (Knetsch and Sinden 1984; List 2011). Alternatively, the lack of a budget constraint when survey participants are asked about accepting compensation (as opposed to paying a particular sum) could reduce the incentive to answer honestly (Arrow et al. 1993). Another explanation comes from economic theory — economists have shown theoretically that a large gap between willingness to pay and willingness to accept can arise when public goods are provided at a fixed quantity to everyone and are not perfect substitutes for private goods (Hanemann 1991).

These arguments suggest that stated preference methods may accurately reflect real behaviour and provide theoretically consistent results, at least when participants are

familiar with the good they are being asked to value. However, testing these theories can be difficult and so the evidence remains inconclusive on this point.

Can the results be replicated?

Reliability offers another perspective on the quality of stated preference methods. This refers to whether the results can be replicated. The evidence is limited, but provides a strong indication that willingness to pay estimates are similar when a different sample of people is surveyed, or when the same people are surveyed twice (Carson et al. 1997; Loomis 1989; McConnell, Strand and Valdés 1998).

While this suggests that stated preference methods can provide consistent results, in some cases there may be little reason to expect that values will be constant. Some values may vary over time and across populations (a general characteristic of economic value that will also affect revealed preference and market-based estimates). Indeed, one study found that changes in attitudes towards the environment and government explained differences in value estimates from surveys (of random samples of the population) conducted five years apart (Whitehead and Hoban 1999).

Stated preference methods can be valid and reliable

On balance, the evidence suggests that stated preference methods can provide estimates of non-market values that are sufficiently valid and reliable to use in policy analysis. This is especially so where other quantitative data on the value of non-market outcomes are not available. While there is no compelling evidence that stated preference methods *in general* are invalid, the literature does point to several conditions that are necessary for a study to provide valid results (even though none of these alone can guarantee validity). Criteria for a good study are set out in section 2.3.

One important question is whether stated preference methods can provide valuations for non-market environmental goods when familiarity is low or survey participants have a low level of understanding of the good. For example, when people are asked about environmental assets that are relatively unfamiliar to them (and which they may never visit) they rely more on the information presented to them and may have to construct their preferences during the survey. While this can be done, insights from behavioural economics suggest that people are more likely to be prone to cognitive biases in such low-experience situations.

Low familiarity may be a problem when non-use values are concerned. Because stated preference techniques are usually the only way to estimate such values, it is hard to obtain evidence on how well the methods can do this (especially in relation to criterion and convergent validity). As Atkinson, Bateman and Mourato (2012, p. 30) note:

> … while stated preferences may provide sound valuations for high experience, use-value goods, the further we move to consider indirect use and pure non-use values, the more likely we are to encounter problems. Paradoxically, then, where [stated preference] techniques are most useful is also where they have the potential to be less effective.

The lack of alternative estimates of non-use values means that the validity of stated preference methods for quantifying such values cannot be established as robustly as it can for use values. However, the literature does show that stated preference methods can, under certain conditions, provide reliable and internally consistent estimates of non-use values that conform to the predictions of economic theory. While problems related to the lack of familiarity are more likely to occur for non-use values, there would seem no good reason to conclude that estimates of non-use values are meaningless.

The evidence demonstrates the importance of clearly specified policy outcomes and good survey design for minimising potential sources of bias in stated preference estimates. In particular, estimates are more likely to be valid when participants are presented with credible choices that are expressed in terms of environmental outcomes that they ultimately care about, and when they perceive their responses to be consequential. It is also important that sufficient information is provided to allow for well-informed responses, without adding unnecessary complexity or over-burdening participants (such that some choose not to complete the survey or submit protest responses, which can bias estimates).

Revealed preference

This section provides a much shorter examination of the validity of revealed preference methods than that provided for stated preference methods. The main reason for this is that the overall validity of revealed preference methods is generally accepted by economists. This is because they apply well-established econometric techniques to data derived from people's actual behaviour. Therefore, rather than examining the fundamental validity of revealed preference methods, this section focuses on the factors that may influence validity in particular situations.

Researchers have generally assessed the performance of revealed preference methods by focusing on how sensitive estimates are to key assumptions and data

sources. For the travel-cost method, there is evidence that estimates of consumer surplus can be sensitive to a number of factors, including:

- the way that the time cost of travel is valued (Smith and Kaoru 1990)

- how multiple-purpose travellers are taken into account (Clough and Meister 1991; Loomis 2006)

- how substitute sites are taken into account (Rosenthal 1987; Smith and Kaoru 1990)

- whether fixed or marginal costs of travelling are used (Bateman 1993)

- the functional form selected to model the demand curve (Lansdell and Gangadharan 2003).

Different studies have used different techniques to deal with these factors.

For hedonic pricing, there is evidence that implicit price estimates can be biased when:

- data on relevant factors are not included in the statistical model, but are correlated with the environmental attribute of interest (Graves et al. 1988; Leggett and Bockstael 2000)

- households do not have perfect information on local environmental attributes and thus misperceive the levels of these attributes when buying houses (Hanley and Barbier 2009).

There is also evidence that hedonic pricing estimates can be sensitive to the extent of the housing market analysed (Adair, Berry and McGreal 1996; Chattopadhyay 2003) or the functional form used in the analysis (Cropper, Deck and McConnell 1988; Smith and Huang 1995). Further, estimates can be very imprecise when some attributes are highly correlated with each other (multicollinearity) (Hanley and Barbier 2009).

Because they are based on observed behaviours and choices, revealed preference methods are likely to provide reliable estimates of non-market environmental values, provided that good data sources are used and assumptions are clearly justified. The evidence shows that revealed preference estimates — as with stated preference estimates — are sensitive to the quality of data available and assumptions made in the analysis. That said, statistical techniques continue to develop and can sometimes offset deficiencies in data sources, allowing more accurate estimates to be obtained. However, the main limitation of these methods is that they cannot be used unless there is adequate data relating to the non-market outcome.

Benefit transfer

The validity of benefit transfer can be assessed by conducting a new study (at the 'policy' site) and comparing the resulting estimates to those generated from benefit transfer (derived from the 'study site(s)'). Most studies that have done this have found large, statistically significant errors (sometimes well over 100 per cent) when transferring estimates. This is the case for both stated preference (Colombo and Hanley 2008; Kaul et al. 2013; Morrison et al. 2002; Rosenberger and Stanley 2006) and revealed preference methods (Chattopadhyay 2003; Rosenberger and Loomis 2000). In its inquiry into waste management, the Commission found an example of the use of benefit transfer (relating to benzene and other gas emissions from landfills) that likely overstated costs by a factor of 50 to 100 times (PC 2006). Appendix C sets out the evidence for benefit transfer in more detail.

A general finding is that the errors tend to be lower when there is greater similarity between the study and policy contexts, but the evidence on which factors matter most is mixed. For example, some studies have found lower errors when the environmental features of interest are more similar (Johnston 2007). Others have found that differences in attitudes towards the environment across populations have a more pronounced effect (Brouwer and Spaninks 1999; van Bueren and Bennett 2004). Errors can also arise from differences in the scale of environmental change due to a policy, the way the policy is implemented, or the available substitutes for the environmental good (Johnston and Rosenberger 2010).

There is a lack of consensus on which benefit-transfer approaches are associated with the lowest errors. A common finding is that 'function transfer' (where willingness to pay is modelled as a function of variables, drawn either from one or multiple studies) performs better than a simple 'unit' transfer of values (Brouwer and Spaninks 1999; Kirchhoff, Colby and LaFrance 1997). However, some studies have found unit transfer to be more accurate (Bergland, Magnussen and Navrud 2002; Colombo and Hanley 2008). This divergence in findings could partly reflect the specific study and policy contexts in each case, and any assumptions made.

The evidence suggests that transferring estimates of non-market values from one context to another is likely to be very imprecise (and possibly misleading) unless there is a high degree of similarity between the 'study' and 'policy' contexts (in terms of the environmental features, policy outcomes and population characteristics). Because environmental outcomes tend to be heterogeneous, there may be relatively few cases where benefit transfer can be accurately applied to value environmental outcomes. This may not, however, be the case for the value placed on outcomes that are more similar across policy contexts, such as the value

of a statistical life or transport waiting times (Australian Transport Council 2006; OECD 2012; Ready and Navrud 2007).

2.3   What makes a good study?

A good non-market valuation study has several key features. This section sets out criteria that policy makers can consider when commissioning, or assessing the likely validity of, a study. These criteria are drawn from the available evidence on the validity and reliability of methods, set out above and in appendix C. The stated preference criteria also draw on published guidelines commissioned by the US Government (Arrow et al. 1993) and UK Government (Bateman et al. 2002).

While the criteria identified are generic and not an exhaustive set of requirements, they point to several areas where non-market valuation methods and practice have improved over time in response to criticism. A common theme is that the approaches and assumptions used in a study need to be clearly set out and communicated (including by reporting ranges of values), and tested where possible using sensitivity analysis.

In some policy contexts, complying with every criterion may be too costly. It can also be impractical when highly precise estimates are not needed. The potential benefit of having accurate estimates relative to the cost of undertaking a primary study is an important consideration. Chapter 3 further discusses how non-market valuation can be used to inform policy analysis.

Stated preference

Stated preference studies that estimate non-market environmental values should generally have the following characteristics.

- *Participants are given the impression that their answers are consequential* (by influencing policy decisions they care about) and that they may be compelled to pay any amount they commit to in the survey. This gives participants an incentive to answer carefully and honestly. Part of this is ensuring that the payment mechanism by which people would financially contribute — such as higher taxes — is specific and credible, as well as being generally accepted by stakeholders. The choice of payment mechanism can be difficult, for example, because some participants could consider a one-off levy to be unrealistic, or feel they would be immune from increases in taxes they currently do not pay. However, focus groups can help to fine tune the choice of mechanism, and

follow-up questions can be used to detect (and adjust for) problems with credibility.

- *The environmental goods or attributes in the survey are expressed in terms of endpoints that people directly value.* For example, people should be asked about willingness to pay for the environmental improvements brought about by increases in environmental water flows, rather than for increases in environmental water flows themselves. In some cases, difficulty in selecting attributes that relate to endpoints can warrant the use of contingent valuation in preference to choice modelling (discussed further below).

- *There is alignment between the environmental goods or attributes being valued and the likely policy outcomes.* One aspect of this is that the survey should not reflect an overly optimistic or pessimistic view about what the policy would achieve. The best available biophysical information should be used, with any major uncertainties made clear. Another aspect is that descriptors like 'good', 'fair' or 'poor' environmental condition should not be used unless they can be understood by participants and explicitly related to the actual outcomes that may be achieved. All major environmental outcomes associated with the policy should be covered by the survey.

- *The information provided to participants is clear, relevant, easy to understand and objective.* Focus groups and pilot surveys can be useful to ensure that participants clearly understand the survey material, and consider it to be relevant and credible. Where appropriate, maps, images and diagrams should be used to convey key information. Consultation with stakeholders can be useful to ensure that disagreements about what constitutes objective information can be resolved.

- *Participants are encouraged to consider the context of their decisions*, including their income and other expenditures, as well as alternative or substitute environmental outcomes (for example, potential policy changes that would affect similar environmental assets).

- *The valuation questions require participants to make discrete choices* (such as 'yes/no' or selecting options), and include a 'no-answer' option to identify participants that are indifferent, unfamiliar with the environmental good, or object to the question (supplemented by follow-up questions as outlined below).

- *Valuation questions are designed and analysed using appropriate statistical techniques* (box 2.5).

- *Follow-up questions are used to detect potential sources of bias*, including 'protest' answers and cases where participants did not understand the valuation question(s) or the information provided. Where these factors significantly impact results, appropriate adjustments should be made in the statistical analysis. The

study should disclose reasons that participants provide for any protest responses and the method used to identify these responses.

- *Participants are given adequate time to complete the survey.*

Box 2.5   Econometric modelling of stated preference data

Statistical techniques play a key role in the analysis and design of stated preference surveys. Some early uses of contingent valuation involved asking people directly for their willingness to pay, which could be averaged across participants. Methods have since been refined and people are now usually asked to make discrete choices (such as whether or not they would be willing to pay a set amount, or which option they prefer in a choice set). Statistical models are needed to translate answers to these types of questions into estimates of average willingness to pay (usually drawing on random utility theory). Specialised expertise is generally required since the way that stated preference surveys are designed and the models used to analyse the data can have a large influence on the results. Bateman et al. (2002), Haab and McConnell (2002), Hanley and Barbier (2009), and Hensher and Greene (2003) provide greater detail on the econometrics of survey design and analysis (summarised briefly below).

Survey design

Statistical considerations are important for survey design. Statistical efficiency can be enhanced when different participants are asked valuation questions with different levels of payments and attributes (provided that these levels are realistic and credible). Selecting the right number of levels, and their values, is key.

In contingent valuation surveys, payment levels need to cover the likely range of amounts that participants would be willing to pay. A rough indication can be obtained by pre-testing surveys. Techniques of optimal-bid design can be used to fine-tune the payment levels.

In choice modelling, the efficiency of statistical models can be increased by offering each participant multiple choice sets. The attributes used need to be 'orthogonal' (can be varied independently of each other), although ways around this are being developed. No option should dominate all others within a choice set (in terms of having 'better' levels of all attributes). Fractional-design techniques can be used to select the most efficient combinations of attributes. It is also important that participants are not burdened with too many choice sets.

Data analysis

Econometric models that estimate willingness to pay are based on assumptions that can have a large influence on results. In particular, estimates can be highly sensitive to assumptions about the distribution of willingness to pay (such as normal, lognormal or Weibull) (Alberini 2005).

(Continued next page)

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Box 2.5 (continued)

Other assumptions are implicit in the choice of model used, and there are trade-offs. For example, conditional and multinomial logit models are usually the most straightforward way to analyse choice modelling data. However, these are based on an assumption that the probability of choosing between two options is independent of all other available options. This is not always the case in discrete choice experiments. Mixed logit and random-parameter models that allow for individual-specific randomness can avoid this assumption, but are more difficult to compute and require further choices to be made by the analyst (such as which parameters are set as random).

- *The sample of people surveyed is representative of the broader community* (in terms of location, income, age and other characteristics), and large enough to permit robust data analysis. The study should clearly set out how people were selected for the survey, the number of participants and the response rate. While the scope of the relevant population (for example, across a region or state) can be difficult to determine and a matter of judgment, it should be clearly set out and justified.

- *Estimates of average willingness to pay are supplemented with confidence intervals to indicate the precision of the estimates.*[1] Per individual (or household) estimates of willingness to pay should lie within the range of values presented to survey participants. The impact of relevant variables on willingness to pay should be analysed so that economic predictions can be tested (such as higher willingness to pay for a higher quantity of a good).

- *Population-wide estimates of the benefits or costs of a policy are calculated in a transparent and appropriate way.* Potential reasons for non-response to the survey should be identified. Sensitivity analysis should be used to demonstrate how aggregate estimates change depending on assumptions about the values held by non-respondents and the extent of the population affected by the policy (box 2.6).

- *A copy of the survey instrument is attached to the study report*, along with a list of all payment levels and attributes used in different versions of the survey. Ideally, the underlying data should be made available, so that other researchers can replicate the statistical analysis.

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[1] It is important to note that confidence intervals are a statistical construct, based on the range of responses from participants and assumptions made about the distribution of willingness to pay. This means that true willingness to pay could lie outside the range of a confidence interval when the estimates are biased (for example, because participants did not answer honestly).

Box 2.6   **Aggregating willingness to pay estimates**

Non-market valuation studies provide average per person (or per household) value estimates that need to be aggregated across the relevant population (which may be a region, state or country) to produce a total figure that can be used in cost–benefit analysis. For the total figure to be valid, the survey should target a representative sample of the population. However, even when this is done an assumption must be made about the willingness to pay of non-respondents.

At one extreme, it could be assumed that those who chose not to participate did so because they do not care about the issue and so have a zero willingness to pay. At the other extreme, non-respondents could be assumed to have similar preferences to those that did respond. The assumptions made can have a large impact on total value estimates, especially when response rates are low — for example, estimates could differ by a factor of four or more when response rates are lower than 25 per cent.

Several techniques have been used to address non-response biases. These include:

- imputing willingness to pay for non-respondents using available socioeconomic data and estimates of how socioeconomic factors influence willingness to pay

- using distance–decay functions that assume willingness to pay declines with the distance from an environmental feature

- assuming that a particular proportion of non-respondents have similar preferences to survey participants but the remainder do not value the outcome (Morrison 2000; Whitehead and Blomquist 2006).

To support the use of the latter technique, Morrison (2000) used a follow-up survey to estimate that around 30 per cent of non-respondents are likely to share similar values to survey participants. Some other practitioners have followed this lead and also used the 30 per cent figure.

Given the sensitivity of total value estimates to the assumptions made, further research on this issue may be warranted. Such research could separately examine in-person, mail and email based surveys, as reasons for non-response may differ for each.

Revealed preference

Revealed preference studies that estimate non-market environmental values should generally have the following characteristics.

- Reliable data on all relevant variables that influence the behaviour of interest (such as travel decisions or house purchases) are used.

- Key calculations and assumptions are clearly set out (including the choice of functional form). Sensitivity analysis is used to demonstrate the effect that these assumptions have on the results. Data are analysed using the most appropriate statistical models and techniques (Haab and McConnell 2002).

Travel-cost studies

- Data and assumptions relating to the costs people incur when travelling are clearly set out. Attempts are made to determine what proportion of these costs can be attributed to the site of interest, based on responses to the survey.

- Substitute sites are taken into account in the statistical model.

- The treatment of multiple-purpose and international visitors in the analysis is clearly specified.

- Justification is provided for the value placed on the time cost of travel.

- The information and questions in surveys are clear and unambiguous. Sampling techniques are explained and response rates identified.

- A copy of the survey instrument is attached to the study.

Hedonic pricing studies

- Data sources and any transformations of data are clearly specified.

- The market is characterised by a large number of transactions, and any regulatory distortions to prices are taken into account in the analysis.

- Justifications are provided for the extent of the market used in the analysis (such as the geographic scope of a housing market), and alternative definitions are tested where appropriate.

- Where data on all relevant attributes are not available, the potential impact of any omitted variables is discussed.

- Variables used in the statistical model are carefully chosen to reduce multicollinearity.

- Implicit price estimates are only used to value small or marginal changes in attributes.

Benefit transfer

Benefit-transfer studies that estimate non-market environmental values should have the following characteristics.

- The primary study (or studies) is selected so that the differences between the current policy context and the context in which the primary study was undertaken are small. In particular, the environmental good, the type and extent of environmental change due to policy and the characteristics of the affected population are similar (for example, estimates of the value of improvements to

specific wetlands are not extrapolated to cover an entire river basin). These factors are set out and compared for both contexts.

- The primary study is of high quality, and aligns with the criteria for stated and revealed preference studies set out above.

- Any adjustments made to estimates to reflect differences between the study and policy contexts are clearly set out and justified, including the choice of unit or function transfer. Sensitivity analysis is used to demonstrate the impact that these adjustments have on the transferred estimates.

Selecting the right methods

In circumstances where it would be useful to undertake a non-market valuation study (chapter 3), a remaining consideration is which method to use. This will largely depend on the type of non-market outcome, available data and the information required for policy analysis. Figure 2.1 sets out some initial questions to consider (intended as a broad guide to selecting a method — in practice, the most appropriate methods to use will depend on the specific circumstances).

Where suitable data are available to support the use of revealed preference methods, these can provide estimates derived from actual economic behaviour. However, when such data are not available (such as when non-use values are thought to be significant), stated preference methods may offer a useful alternative.

A key consideration with stated preference methods is which technique to use. Choice modelling can be more appropriate where values for particular attributes would allow for more flexible formulation of policy, and where people value these attributes separately from one another. Contingent valuation is better suited to valuing the outcomes of a policy change as a whole.

There are also situations where the flexibility of choice modelling would be desirable, but the nature of the non-market values make contingent valuation a more straightforward approach in practice. These include cases where it is not possible to select attributes that can be varied independently of one another, such as when environmental processes are interdependent or there are complex interactions between them. Contingent valuation may also be preferable when the endpoints that people care about are broader than a collection of specific attributes (for example, overall wetland health rather than numbers of fish or hectares of wetland).[2]

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[2] Where this is the case and the policy decision concerns a particular type of intervention (such as setting water extraction limits), one option is to use contingent valuation to value 'packages' of environmental outcomes associated with different policy settings (for example, different water extraction limits).

Sometimes both stated and revealed preference methods may be worth using (such as for estimating the value of a statistical life). At other times, a primary non-market valuation study may be too costly or unlikely to influence the choice of policy option (chapter 3). In these cases, benefit transfer may offer a practical alternative, or environmental outcomes may be considered in other ways.

Figure 2.1   **Selecting a non-market valuation method — initial questions**
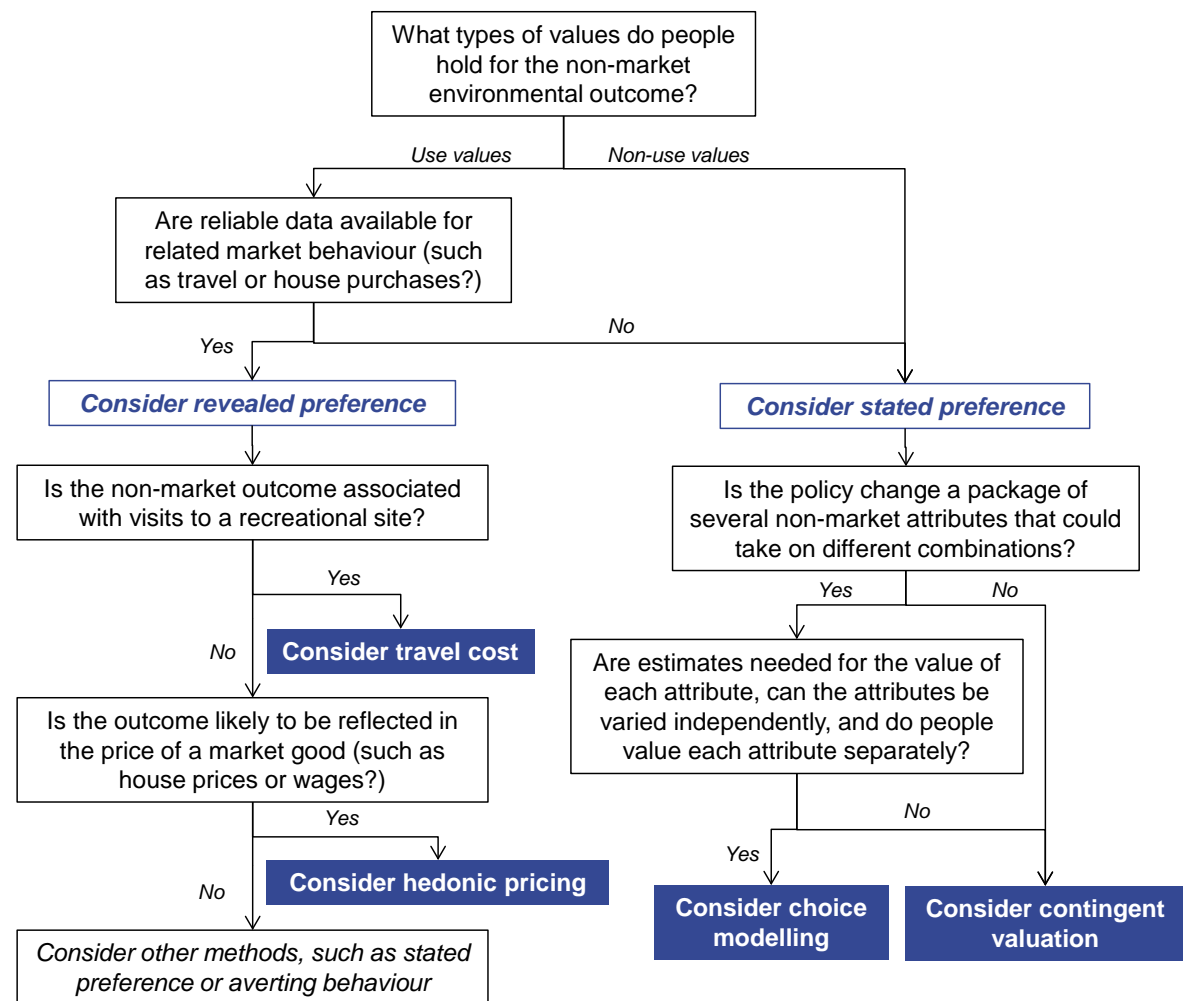
3   Use in environmental policy analysis

Key points

- Environmental policy analysis should use a cost–benefit framework that considers both market and non-market outcomes. In some cases a cost–benefit analysis that values market outcomes but provides only a qualitative description of non-market outcomes will be sufficient for identifying a preferred policy option.

- In other cases, the preferred policy option will depend crucially on trade-offs between market and non-market outcomes. Non-market valuation methods have advantages over alternative approaches to assessing these trade-offs.

  – Some alternatives, such as multi-criteria analysis, are deficient in the way they deal with non-market outcomes and can also be inconsistent with a cost–benefit framework.

  – Approaches involving expert valuation have considerable potential for improving the cost effectiveness of policy. However, they are not able to shed light on what trade-offs the community would be prepared to make between dissimilar outcomes (such as reduced taxes and improving the condition of a wetland).

- Given cost considerations, the case for non-market valuation is likely to be strongest where the financial or environmental stakes are high and there is potential for non-market outcomes to influence the choice of policy option.

- The development of comprehensive sets of environmental non-market values would assist in incorporating non-market outcomes into policy analysis. At present this is either not being done or expert-led valuation approaches are used. There is merit in considering the use of strategic approaches to conducting non-market valuation studies supplemented by benefit transfer.

- There is considerable academic interest in non-market valuation, but its use in policy analysis in Australia is limited. It is more widely used in the US and the UK.

- Where non-market value estimates are made they should be included in a cost–benefit analysis. Results should be presented with and without the non-market values, the likely accuracy of all components explained and sensitivity analysis provided. Non-market value estimates should be accompanied by information about the underlying non-market outcomes.

- One of the main barriers to increased use of non-market valuation is failure to apply a cost–benefit framework. If this and other barriers could be overcome, steps could be taken to build confidence and make the most of non-market valuation, including:

  – paying greater attention to the quality of studies

  – developing knowledge and capacity in government departments

  – refocusing research effort on policy needs.

The previous chapter concludes that non-market valuation methods, when conducted well, can provide estimates that are valid and reliable. On its own, this does not settle the question of when and how they should be used in environmental decision making. In this chapter, non-market valuation is compared to other approaches to factoring environmental outcomes into policy analysis. This leads into a discussion of when non-market valuation is most likely to be useful, and how it should be used. The chapter concludes by suggesting ways that barriers to the use of these methods might be overcome and confidence in their application improved.

3.1   Comparison with alternatives

Non-market valuation methods have strengths and weaknesses. While they can tap into preferences that are representative of the community, considerable expertise, time and money are needed to produce reliable estimates, and even then they are likely to be imprecise. However, all methods for factoring non-market outcomes into policy analysis have shortcomings, as discussed below. The task is to identify the best approach for a given context.

Not explicitly considering the value of non-market outcomes

Because non-market outcomes are difficult to value they are sometimes overlooked altogether (implicitly assigning them a zero value) or given precedence over all other considerations (implicitly assigning them an infinite value). Carson (2012, p. 28) argued:

> Those working on benefit–cost analysis have long recognized that goods and impacts that cannot be quantified are valued, implicitly, by giving them a limitless value when government regulations preclude certain activities, or giving them a value of zero by leaving certain consequences out of the analysis. Contingent valuation offers a practical alternative for reducing the use of either of these extreme choices.

There are many instances where particular activities that are thought to produce environmental benefits are promoted by government policies without explicit consideration of whether the benefits outweigh the costs. For example:

- Pannell (2013b, p. 5) reported that it is common for agri-environment programs (such as environmental stewardship programs) to 'invest in the promotion of what are perceived to be environmentally favourable practices, with faith that these practices will benefit the environment, but no real knowledge of the extent or value of those benefits'.

- The Commission has reported that environmental offset policies (which can require developers to protect and enhance the condition of an area of native

vegetation to compensate for clearing native vegetation elsewhere) are commonly based on a 'no net loss' principle. The Commission found that this 'prioritises … the impacts of the project on environmental matters over all other impacts (including economic and social impacts)' (PC 2013, p. 236)

- There are a range of policies that encourage or require the installation of rainwater tanks with the aim of producing environmental and other benefits to the community. The Commission found evidence that these benefits were generally small relative to the cost of rainwater tanks (PC 2011). In specific circumstances the benefits were substantially higher, but most policies did not vary according to circumstance.

- The Commission found that many State and Territory Governments set targets for waste diversion (through avoidance, reuse or recycling), with some going so far as to aim for zero waste to landfill (PC 2006). These targets were usually set based on a general presumption that waste disposal was the least-preferred option and on technical considerations of what was feasible. Costs were generally not weighed up against benefits.

The value of non-market outcomes is not explicitly considered in these and in many other cases. However, it is worth noting that it is also common that the environmental outcomes themselves are not assessed or quantified. In other words, decisions are made without knowing how the condition of the environment will be changed by the policy, let alone what the value of that change might be.

Policy analysis that does not consider the costs and benefits of environmental protection ignores the trade-offs involved in achieving desirable non-market outcomes. For government investment in environmental improvement, this can result in poor project selection, such that less environmental benefit is achieved from a given budget than was possible (Pannell (2013b) suggests up to 330 times less, based on data from potential environmental investments in Australia). For environmental regulation, it can result in high costs being imposed on the community for a small (or nonexistent) environmental benefit. On the other hand, pro-development actions that ignore non-market outcomes can result in the degradation of environmental assets that are highly valued by the community for small commercial gain.

Partially quantified cost–benefit analysis

Sometimes a cost–benefit analysis describes and discusses non-market outcomes but does not value them. This description can include quantification in non-value terms, such as the expected increase in bird breeding from additional environmental water, or the area of different vegetation types to be cleared in a development project. Information can also be provided about how significant the outcomes are to the community relative to the quantified costs and benefits.

The Australian Government's *Best Practice Regulation Handbook* endorses the inclusion of qualitative discussion of some policy outcomes (or impacts) in assessments of regulatory proposals, stating that:

> Where it is not possible to quantify impacts [in dollar terms], the cost–benefit analysis should recognise this and include a qualitative discussion of these impacts so that they can be compared with other impacts that can be more easily quantified. (Australian Government 2013, p. 96)

However, the handbook also specifies that an attempt should be made to quantify (in monetary terms) all highly significant costs and benefits, and refers to both revealed and stated preference methods as ways of doing this.

Undertaking a cost–benefit analysis that describes but does not quantify non-market values is likely to be a reasonable approach where the cost of a non-market valuation study is not warranted and suitable studies to support benefit transfer are not available. In some cases a partially quantified cost–benefit analysis can clearly identify a preferred option. This is because the option showing the highest net benefit based on quantified costs and benefits may also be the one that produces the best non-market outcomes.

For example, the market costs of logging an area of native forest (including road construction, logging supervision and regeneration costs) could be larger than the value of the logs harvested. Where this is the case, an incomplete cost–benefit analysis would identify not proceeding with logging as the preferred option. Any non-market impacts of logging, such as loss of biodiversity or reduced visual amenity, could be described, but provided these were predominantly negative, failure to quantify them would not affect the usefulness of the analysis. Box 3.1 outlines an example where investing in environmental improvement was the preferred option because of the *market* benefit this provided.

In other cases, a partially quantified cost–benefit analysis leaves the decision maker to judge whether differences in non-market outcomes tip the scales in favour of an option that would not be preferred based on market costs and benefits alone. This can be a very difficult task, but at least the trade-off can be made clear. For

example, the cost–benefit analysis could spell out explicitly that one option would have higher net benefits than another if the difference in non-market outcomes between the two is judged to have a value to the community in excess of a specified amount. For example, Bennett (1998) estimated such 'threshold values' for forest preservation for regions of New South Wales.

Box 3.1   **Management of catchments supplying New York City**

Historically, the Catskills catchments supplied New York City with high quality water with little contamination due to the natural filtration processes of the ecosystems on the banks of streams, rivers, lakes and reservoirs. However, increasing housing development and pollution from vehicles and agriculture threatened water quality in the region. By 1996, New York City faced a choice: either it could build water filtration systems to clean its water supply or the city could protect the Catskills catchments to ensure high-quality drinking water.

A decision was taken to protect the Catskills catchments. It was estimated that the total cost of building and operating a filtration system was in the range of US$6 to $8 billion. In comparison, the total cost for protecting the water-provision service of the Catskills through land purchases and regulations to control development and land use in the catchments was estimated at US$1 to $1.5 billion. The catchment protection option also produced non-market environmental benefits, but quantifying them was unnecessary for reaching a decision.

Sources: Barbier and Heal (2006); PC (2011).

Consultation with stakeholders is important for informing judgments. However, the absence of non-market value estimates means that the decision maker may not be informed in an objective and unbiased way about the strength of preferences across the community. This brings the danger that decisions may be unduly influenced by lobbying from prominent stakeholders. The cost of a poor decision should be weighed up against the cost of a non-market valuation study.

A final point is that something can be learnt from the non-market valuation literature about describing non-market outcomes well. Carson argued:

> Much of the usefulness of doing a contingent valuation study has to do with pushing scientists and engineers to summarise what the project would do in terms that the public cares about. (Carson 2012, p. 31)

Even when non-market values are not estimated, it would seem worthwhile to 'push' scientists and engineers to do this. Such a description would ideally include quantitative elements, such as the area of native vegetation that would be expected to be in an improved condition as a result of a policy, and a measure of the degree of improvement (relative to what would have occurred in the absence of the policy).

Alternative valuation approaches

Chapter 2 describes non-market valuation methods that are based on aggregating the preferences of the community in a way that is consistent with economic theory. There are various alternative ways of estimating non-market values that involve deliberation by expert or community groups. Sometimes these values are expressed in non-monetary terms, such as an environmental benefits index.

Expert valuation

Some critics of non-market valuation methods advocate the use of valuation by scientific and/or policy experts (Hausman 2012). In practice, experts (such as environmental scientists and environmental managers) often do make environmental policy decisions based explicitly or implicitly on their own valuations. Sometimes this is done using analytical tools and models that incorporate values (or weights). For example:

- multi-criteria analysis has been used in Queensland, Western Australia and some other jurisdictions to weigh-up environmental and other outcomes of policy options (a process that can effectively place an implicit value on non-market outcomes)

- the Investment Framework for Environmental Resources (INFFER) has been used by a number of Catchment Management Authorities and other regional bodies in several states to prioritise environmental investments

- Marxan is a decision-support tool that has been applied to a range of conservation planning problems in Australia and elsewhere (for example, it was used to assist in the rezoning of the Great Barrier Reef)

- EnSym is a decision support tool designed to help prioritise natural resource investment, which has been used by the Victorian Department of Environment and Primary Industries (DEPI) to evaluate the relative cost effectiveness of bids in environmental tenders, and other policies and programs

- NaturePrint is a model that integrates and analyses information about biodiversity values, threatening processes and ecosystem function that is used by DEPI to evaluate the relative biodiversity value of locations across Victoria.

These approaches differ from one another in important ways. Box 3.2 explains how approaches that are broadly consistent with cost–benefit analysis (such as INFFER) can offer advantages over those that are not (such as most forms of multi-criteria analysis).

Box 3.2   **Two analytical tools that incorporate expert valuation**

Multi-criteria analysis (MCA)

MCA is an alternative to cost–benefit analysis that is often used where non-market outcomes are important. MCA is simpler to apply than cost–benefit analysis. There are many variants of MCA, but it usually involves defining policy objectives, determining a set of criteria to measure performance against each objective and assigning weights to criteria. Typically, some criteria relate to market factors (such as the cost of funding a project) and some to non-market outcomes. Each policy or project option is given a score for each criterion and these are weighted and added up to give an overall score.

A range of analysts have argued that MCA is seriously flawed (Dobes and Bennett 2009; Pannell et al. 2013). Criticisms include that:

- while a major motivation for choosing MCA is to avoid assigning dollar values to environmental outcomes, the method usually does implicitly assign dollar values
- implicit values from MCA are a consequence of the framing of the policy problem and the way that a particular MCA is done, meaning that two analyses may produce very different values for the same outcome
- the adding up of weighted scores can lead to errors because there are situations where they should be multiplied (for example, 'benefit if successful' should be multiplied by, not added to, 'probability of success')
- implicit values are usually determined by a single individual or small expert group, and therefore do not represent community preferences.

Investment Framework for Environmental Resources (INFFER)

INFFER is a tool for developing and prioritising projects to address environmental issues, such as reduced water quality, biodiversity, and land degradation. Like MCA, it is designed to be simpler to use than cost–benefit analysis. However, unlike MCA, it is based on the principles of cost–benefit analysis (Pannell et al. 2012).

INFFER involves scoring the value of the environmental asset in question relative to a table of well-known environmental assets. This score is converted to a dollar value, based on estimates of the value of the well-known assets. A range of other inputs are then used to estimate the change in value expected to result from the project. These include estimates of the impact of the project on the asset's value (if successful) and information on various types of risks, including those related to technical feasibility and adoption of desired practices by landowners. The change in value is used in calculating the benefit–cost ratio of projects.

Like MCA, INFFER is open to criticism because it uses values that are not based on community preferences (although informal means, such as community workshops, have sometimes been used to inform value estimates). However, it has a range of advantages compared to MCA, including that it avoids logical errors in determining project rankings and explicitly considers relevant sources of risk.

For all methods, however, it is important to consider whether experts are qualified to value non-market outcomes. Scientific expertise may be used, but science has no normative content about what should be done. Rather it provides information about the consequences of different choices (PC 2010). This was recognised, for example, by an expert panel on environmental flows and water quality requirements for the River Murray system.

> It was not the role of the Expert Reference Panel, or scientists in general, to decide upon the compromise between the competing values of production, ecosystem services and the natural environment. (Jones et al. 2002, p. 4)

Scientists and others working in environmental areas are likely to have their own views about the value of achieving particular environmental outcomes. However, they are unlikely to reflect the same characteristic mix as people drawn randomly from the population. In addition, where experts provide a valuation they are essentially stating what the community as a whole should be prepared to pay to achieve an outcome. But there are many alternative uses for these funds, for example in education, health or for tax cuts, and environmental experts are not well placed to evaluate these trade-offs on behalf of the community. Therefore, the use of expert valuation to evaluate trade-offs between environmental and other types of outcomes would seem to be a poor alternative to non-market valuation.

A more justifiable role for experts is to seek to improve the cost effectiveness of policies directed at achieving environmental objectives set by governments. Governments clearly do have a role in making trade-offs on behalf of the community and, in a democratic society, are accountable for their decisions through elections. Where governments have adopted clear objectives and priorities it would seem appropriate to use these as the basis for determining values for some types of policy analysis.

Sometimes cost-effectiveness analysis does not require environmental outcomes to be valued. For example, if the objective is to lower salinity in a river to a certain level, all that is required is information on the costs and salinity-lowering potential of the available policy options. However, very often cost-effectiveness analysis requires judgments to be made about the relative merits of achieving different environmental outcomes. For example, whether it is more beneficial to improve the water quality in one river or a different river, or to prevent the clearing of one type of native vegetation or another. Making these sorts of judgments requires some sort of value or benefit metric, although not necessarily a monetary one.

Expert-determined value metrics can help to improve the cost effectiveness of environmental policy in several ways. First, where environmental objectives have been pursued with little attention to the value of environmental outcomes, there may

be much that can be done to improve cost effectiveness by applying common sense combined with a systematic approach. For example, experts might identify that high costs were being incurred to achieve an improvement in the condition of a wetland, when there were lower-cost opportunities to achieve equivalent improvements to wetlands that, based on any reasonable criteria (such as size, biodiversity, and number of visitors), were of higher value. Developing and consistently applying simple value metrics based on fairly uncontroversial criteria can be worthwhile in such circumstances.

Second, experts can apply more sophisticated analysis to estimate the relative contributions of different environmental assets (or actions) to high-level environmental objectives, and this can be used to develop sets of values. For example, DEPI's NaturePrint, reportedly:

> … brings together large amounts of information collected about species presence, habitat quality and connectivity, to determine relative environmental value across the landscape. (DSE 2012, p. 23)

NaturePrint has been incorporated into Victoria's native vegetation clearing regulations and it appears that it has enabled the regulations to more cost effectively contribute to the Victorian Government's biodiversity conservation objectives (PC 2013). One way that it does this is by allowing environmental offsets to be determined on a 'value for biodiversity' basis rather requiring offsets to be of the same vegetation type. This can substantially reduce the cost to developers of providing offsets, without compromising biodiversity outcomes.

Third, it may be that some people favour the relative value of different environmental outcomes being evaluated by experts that understand ecological processes, rather than specifying these values themselves (Clark, Burgess and Harrison 2000).

However, there is potential for the relative environmental values set by experts to depart substantially from those of the community. For example, the community may place a substantially higher value on an environmental asset that is close to a population centre because of the recreational opportunities it provides, but this proximity may not be factored into expert valuations that focus on environmental condition. This potential may be lessened to some degree through informal approaches to factoring community preferences into expert valuations. For example, some applications of INFFER have used community workshops for this purpose (Pannell et al. 2012).

Overall, there are both advantages and disadvantages in using expert valuation rather than non-market valuation in cost-effectiveness analysis of environmental

policies. Experts may be able to develop consistent sets of values at relatively low cost compared to using non-market valuation, but these values may depart significantly from those held by the community. At present, there are many circumstances in which it is not feasible (for cost reasons) to use non-market valuation, and so expert valuation has a role to play. In time, if a strategic approach was taken to conducting non-market valuation studies so as to support benefit transfer, this role might be diminished (some steps have already been taken in this direction — for example, by van Bueren and Bennett (2004) and Greyling and Bennett (2012)).

Whether or not expert valuation is used, scientific and other expertise is essential for predicting the environmental and other outcomes of policy proposals. This involves specifying the likely condition of environmental assets over time, with and without the proposed policy. In doing this, policy risks need to be considered — for example, INFFER pays particular attention to various risks that, if not accounted for properly, could cause outcomes to be greatly overstated (box 3.2). Careful estimation of outcomes is an important complement to non-market valuation. There would be little point in devoting considerable time and expense to obtaining a robust non-market value estimate that is calibrated to a substandard outcome estimate, which might be wrong by several orders of magnitude.

Deliberative valuation

A range of deliberative processes can be used to incorporate community attitudes and values into environmental decision making. These include consensus conferences, deliberative polls and citizens' juries (box 3.3). Such approaches can be used to arrive at either a policy recommendation, statement of views or a non-market value estimate.

While there is considerable interest in deliberative processes that involve representatives of the general community, there are only a few instances of them being used in Australia. A consensus conference was held in Canberra in 1999 on the topic of gene technology in the food chain (Blamey et al. 2000). More recently a citizens' jury was conducted to estimate willingness to pay for improvements in water quality in the Bremer River in south-east Queensland (Robinson et al. 2009).

> **Box 3.3   Deliberative methods**
>
> Several deliberative methods have been developed to incorporate community attitudes and values into policy decisions. The foundations of these methods are in deliberative/discursive democracy.
>
> **Citizens' juries**
>
> Citizens' juries are similar in form to legal juries and require a group of citizens to meet for several days to discuss, consider and report on a matter of public policy. The jury is given a specific 'charge' to address, most commonly requiring it to select the preferred of several presented options. The jury is addressed by and questions witnesses, and reaches conclusions regarding the charge.
>
> **Consensus conferences**
>
> Consensus conferences are typically conducted over several days. A panel of around 12 participants, selected by stratified random sampling, is given a topic to consider on which it formulates key questions. The panel is addressed by and interrogates expert witnesses, and deliberates on the topic. The panel then prepare a document containing their views, opinions, stances and recommendations. This document is discussed with policy and decision makers.
>
> **Deliberative polls**
>
> A deliberative poll involves surveying a random sample of the population before and after collective discussion on the issue under consideration. The selected (usually several hundred) group members spend several days listening to and questioning experts and politicians and discussing the issue, and are then subject to a final poll.
>
> *Source*: Blamey et al. (2000).

Deliberative approaches can be used to arrive at a non-market valuation in four main ways.

- The group decides on a value that society should be prepared to pay.

- The group decides on a value that individuals should be prepared to pay.

- Following the deliberative process, individual group members express a value for what society should be prepared to pay (which are then averaged).

- following the deliberative process, individual group members express a value for what they would be prepared to pay themselves (Fish et al. 2011).

Only the last of these approaches is likely to be able to provide estimates that could potentially be used in cost–benefit analysis. This is because the others encourage participants to think about costs and benefits in aggregate societal terms that are unlikely to fit well with the notion of individual willingness to pay (Blamey et al. 2000).

Two main advantages are claimed for deliberative valuation over economic valuation methods (particularly over stated preference methods).

First, that they allow preferences to be formulated based on better information and understanding. Stated preference methods, it is sometimes argued, often require respondents to construct preferences during the survey, but inadequate information, lack of time, and absence of an opportunity to discuss issues undermines this process (Blamey et al. 2000). In contrast, deliberative processes allow much more time (sometimes several days, compared to a survey that may only take 20 minutes or so), the opportunity to listen to and question experts, and group discussion.

Second, some advocate deliberative processes because they can promote consensus, or at least resolve some differences through discussion and debate. Under this view, individual preferences concerning social choices should be debated, rather than being taken as given. It is argued that deliberative processes (unlike stated preference surveys) treat people as citizens rather than consumers (Fish et al. 2011).

While there would appear to be some validity to the first advantage, there is a trade-off between having better informed participants and having more of them. Deliberative processes typically involve one to two dozen people and this severely limits any conclusions that can be drawn from them about overall community preferences.

Further, deliberative processes do not always change individuals' valuations substantially. Participants in the citizens' jury referred to above completed a choice modelling survey before and after deliberations (Robinson et al. 2009). Willingness to pay estimates for three water quality improvement scenarios in the final survey were all within 4 per cent of the equivalent estimates for the preliminary survey. However, it is not valid to conclude from this single study that deliberative processes do not generally change people's valuations much.

The second claimed advantage is more contentious, as it could be argued to be a disadvantage in some circumstances. This is because a narrowing of differences may result from a few participants dominating the discussion, rather than from genuine agreement. In addition, non-market valuation surveys are now often framed as seeking citizens' preferences about proposed government policy. As Blamey et al. (2000, p. 14) explain:

> … instead of asking 'how much do you value the blue whale?' or 'how much are you willing to pay to protect it?', respondents are now asked 'How would you vote at a referendum if the following two options were available?'.

The adoption of these sorts of questions means that it is not legitimate to characterise non-market valuation surveys as necessarily casting participants as

purchasers of environmental outcomes and contrasting this with deliberative valuation treating them as citizens. The point is that if treating people more as citizens than consumers is seen as a good thing, it does not necessarily follow that deliberative valuation is clearly better than non-market valuation surveys in this regard. Moreover, while those involved in the process experience the benefits of this consultative approach, those not involved and who the policy decision still affects, do not.

Marshall, McNeill and Reeve (2011) discuss a range of challenges that, if not properly managed, can compromise the use of deliberative valuation. For example, discussion may dwell excessively on preoccupations of the group members, while important issues can be overlooked. Also, it may prove difficult to engage a representative sample of citizens in a deliberative process due to lack of interest. In addition, Blamey et al. (2000) argue that deliberative processes can be open to strategic behaviour by both participants and witnesses.

Overall, deliberative valuation seems likely to perform poorly relative to non-market valuation in providing value estimates that are reflective of community preferences and can be included in cost–benefit analysis. However, deliberative processes may be useful when run in parallel to a cost–benefit analysis so as to provide a different perspective (Blamey et al. 2000). Deliberative valuation methods may also provide insights on how to best present information on environmental outcomes that are relatively unfamiliar (Atkinson, Bateman and Mourato 2012).

3.2   Non-market valuation: when and how?

When to use

Two important conclusions can be drawn from the previous section. First, economic non-market valuation methods are generally the most objective and valid means for estimating non-market environmental values. Second, whether it is worth conducting a non-market valuation study to factor non-market outcomes into policy analysis depends on the circumstances. Factors to consider include the importance of non-market outcomes to the policy decision, the cost of undertaking a non-market valuation study, and the type of analysis that can be conducted. Figure 3.1 provides a flow chart that suggests how these and other factors might be taken into account.

Figure 3.1   **Dealing with non-market outcomes in policy analysis**
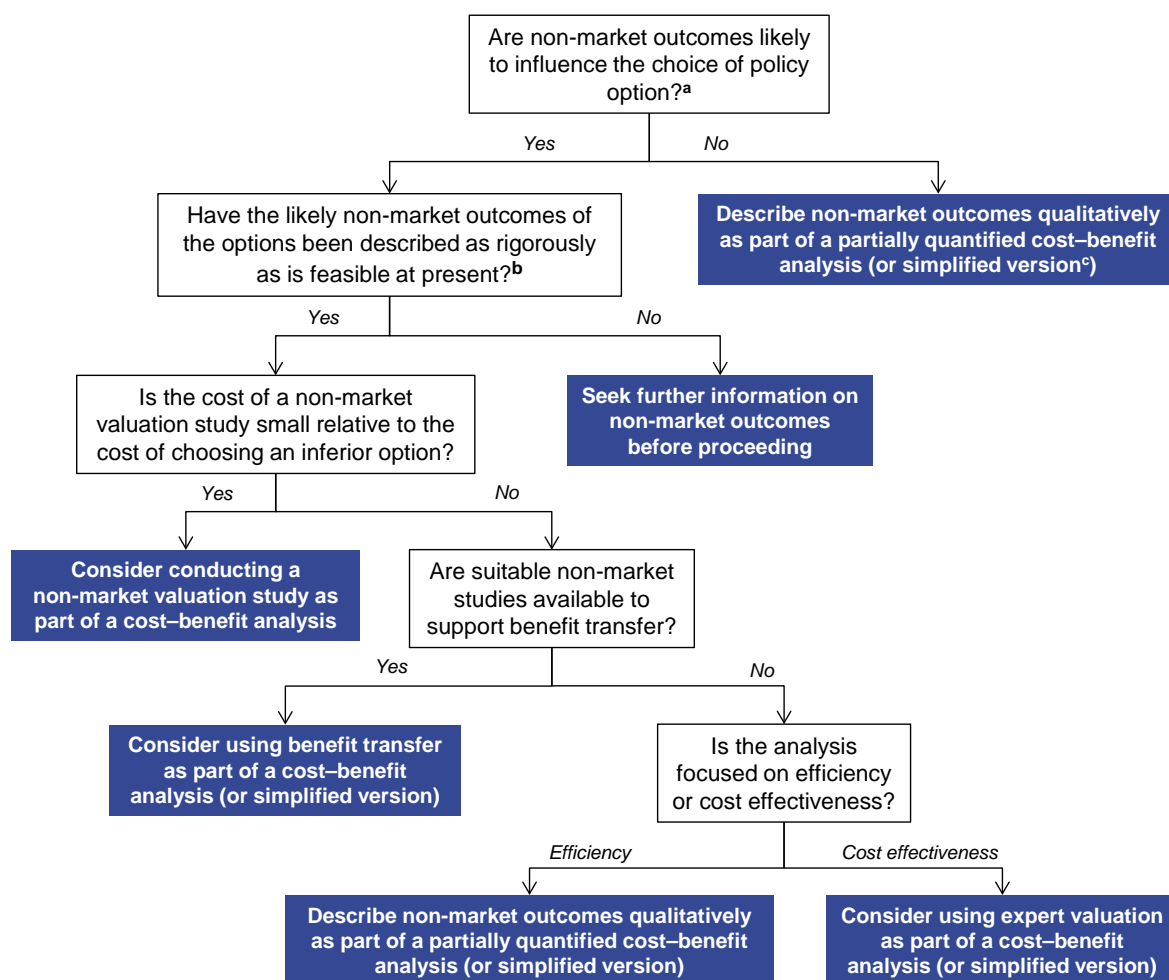
a Preliminary analysis could be useful in establishing whether the value of non-market outcomes is likely to be material in determining which option has the highest net benefit. **b** This does not imply that action should be delayed until uncertainties about the effects of policy options are resolved. Precaution should be applied through risk management frameworks that take account of uncertainty **c** The depth of analysis should be commensurate with the overall effects and in some cases a formal cost–benefit analysis is not justified (Australian Government 2013).

In assessing whether non-market outcomes are important, it should be borne in mind that not all environmental outcomes are non-market in nature (chapter 1). For example, improving the quality of river water might give rise to both market benefits (for example, increased production by irrigators) and non-market benefits (healthier and more sustainable native fish populations). Where market benefits predominate, methods other than non-market valuation are required.

As suggested by figure 3.1, the case for conducting a non-market valuation study is strongest where the cost of a study is low relative to the value of the information it can provide for the policy analysis. In practice, this is most likely to occur where the financial or environmental stakes are high and there is potential for non-market

outcomes to influence the choice of policy option. Possible policy contexts include where:

- different regulatory options are being assessed (for example, different air quality standards)

- a regulatory decision is required on whether, or under what conditions, an action that would have negative environmental consequences should go ahead

- a major government investment in environmental improvement is being considered.

Meaningful value estimates can only be obtained when there is reliable information on what the policy outcomes are likely to be. This will often require scientific assessment of the environmental improvement (or degradation) likely to be brought about by a policy option. In some cases, information about how various groups are likely to respond to the policy is also needed (for example, how farmers are likely to respond to incentives to use more environmentally friendly practices). This information does not need to be precise, but the degree of uncertainty should be documented.

The evidence on benefit transfer presented in chapter 2 suggests that, at present, a shortage of suitable primary studies is likely to mean that this technique can only reliably be used in a limited range of circumstances. This is particularly relevant for areas such as natural resource management and biodiversity conservation, where there are a large number of environmental assets, each with a unique value that can be enhanced (or compromised) to different degrees by proposed policies. However, if even a very imprecise value estimate is potentially of use, benefit transfer may be worth considering even when the available primary studies are less than ideal.

The implication of this, combined with the reasonably high cost of doing non-market valuation studies, is that the bottom portion of figure 3.1 is likely to come into play fairly often. The figure suggests that non-market outcomes should not be quantified in dollar terms where the cost of doing so is high relative to the value of the information it can provide to the policy analysis. Where this applies and the analysis is focused on efficiency, such as whether introducing a new environmental regulation will have net benefits for the community, non-market outcomes should be described. In these cases, the analysis should make it clear that the choice of option comes down to the judgment of the decision maker.

Where analysis is focused on cost effectiveness, expert valuation can have an important role to play. For example, if a budget has been set for funding environmental investments, analysis can help identify the options that are most cost effective in achieving the objectives and priorities of the government.

How to use non-market valuation

Chapter 2 provides information on what makes a good non-market valuation study. In addition, to achieve good results non-market valuation needs to be used in combination with good practice policy principles. For example, the Australian Government's *Best Practice Regulation Handbook* details a number of steps that are to be completed before quantifying the costs and benefits of policy options. One of these is analysing the problem to see whether there is a case for considering new government action (Australian Government 2013). Unless there is a problem requiring action, the development of options, let alone a cost–benefit analysis incorporating non-market value estimates, is unnecessary. Box 3.4 provides an example of the importance of problem identification and analysis.

Where non-market valuation estimates are used it is important that they relate to the change in value resulting from the policy or project. In other words, the correct value to use is 'the difference between the environmental value with the project and without the project' (Pannell 2013a). While this is a simple point, there are reports of the entire value of the asset being used as well as other errors (Maron, Rhodes and Gibbons 2013).

Non-market valuation estimates that are sufficiently reliable should generally be included in a cost–benefit analysis. Results should be presented with and without the non-market values, the likely accuracy of all components of the analysis explained and sensitivity analysis done. It is important to describe the non-market outcomes as well as providing their estimated value (or range of values).

Cost–benefit analysis is an information aid to decision-making, and not a substitute for it. The analysis needs to be presented clearly to allow for proper scrutiny, including of the basis for non-market valuation estimates. There is invariably a role for judgment concerning a range of social, ethical and political considerations, as well as those relating to residual measurement uncertainties (Commonwealth of Australia 2006).

Box 3.4 **Non-market valuation and waste policy**

A Productivity Commission inquiry into waste management found that some Australian governments had adopted unrealistic and potentially very costly targets for reducing waste and increasing recycling.

The Commission argued that the main problem that governments should address was not so much that there was too much waste being produced and not enough recycling, but rather that waste disposal could cause pollution and loss of visual amenity. The market failure rationale for government intervention is because these problems impose costs on people in the community. Accordingly, the Commission argued that waste management policy should be refocused on the environmental and social impacts of waste collection and disposal (including those associated with poorly engineered landfills, types of waste that pose particular hazards, and litter).

The inquiry report acknowledged that waste is the end product of a life-cycle process that can have 'upstream' environmental impacts. That is, if the full-life cycle of a product is not considered, too many resources could go toward its production. However, it argued that these impacts could be much more effectively and efficiently addressed using direct policy instruments, rather than by using waste management policies. For example, it is likely to be more effective to address environmental damage from mining by requiring mining operations to meet specified standards, rather than trying to reduce mining (and hence any associated environmental impact) by placing a levy on waste disposal or subsidising recycling.

Since the Commission's inquiry, a number of non-market valuation studies have been commissioned to try to estimate the benefits of increasing recycling rates. For example, choice modelling was used to estimate the amount that households would be willing to pay for government intervention to increase the percentage of electronic waste (such as televisions and computers) that is recycled (appendix B).

It is not clear that the commissioning of these studies and their use in policy analysis proceeded from a sound analysis of the problem. This is because the focus was on recycling rates (that would not seem to be directly affected by market failure), rather than on the environmental or social benefits from reducing waste disposal.

A related issue is that these studies ask households to value a process rather than an environmental endpoint, which brings the non-market value estimates into question (Collins 2011). This is because people's willingness to pay for recycling rates to be increased may be based on a poor understanding of the likely environmental and other benefits.

Sources: Collins (2011); PC (2006).

3.3 Building confidence in non-market valuation

Current use

Internationally, there is a considerable academic interest in environmental valuation and, in particular, non-market valuation methods including contingent valuation, choice modelling and hedonic pricing. Adamowicz (2004) documented a steep rise in publications on these topics from the late 1980s through to the early 2000s. This activity continues, and in more recent years comprehensive reviews of efforts to value ecosystem services have been published, such as Bateman et al. (2011) and Kumar (2010).

The extent to which academic activity has been mirrored by increasing use of non-market valuation estimates in environmental policy making varies across countries and policy areas.

In the United Kingdom, the use of non-market valuation in environmental policy analysis has increased since the case for it was made in the report *Blueprint for a Green Economy* (Pearce, Markandya and Barbier 1989). For example, it has been used to help develop policy in the:

- design of environmental stewardship schemes and other agri-environment policies

- establishment of marine protected areas

- assessment of water quality improvement schemes and review of water quality regulations

- design of taxes relating to environmental damage (Hanley 2012).

In addition, the *UK National Ecosystem Assessment* raised awareness of environmental valuation and led to the development of a white paper on environmental policy (UK Government 2011). The white paper outlined the UK Government's commitment to valuing nature in its policy making as part of its approach to mainstreaming sustainable development across government.

Non-market valuation methods are also widely employed in the United States in areas such as outdoor recreation, water quality and air quality. For example, the US Environmental Protection Agency provides guidelines on estimating the benefits of proposed policy changes (USEPA 2010). The guidelines give a value of statistical life figure that the Agency recommends be used in analysing environmental policies that impact on mortality. They also provide guidance on the use of revealed and stated preference methods for estimating the value of ecological improvements.

In Australia, qualified support for the use of non-market valuation can be found in various official publications, including the Australian Government's *Best Practice Regulation Handbook* and *Handbook of Cost–Benefit Analysis*, and the Victorian Government's *Victorian Guide to Regulation*. Despite this, information from a variety of sources, including a workshop held by the Commission, suggests that non-market valuation is used less in environmental policy analysis in Australia than it is in the United Kingdom and the United States. That said, its use does appear to have increased over recent years. Appendix B provides some examples of where it has been used in developing policy.

Rogers et al. (2013) surveyed Australian and New Zealand researchers about the use of non-market valuation in environmental policy making. They report there was:

> … little evidence of NMV [non-market valuation] studies making a difference to environmental decision-making in Australia. The great majority of decisions in this sector are made without the use of information from NMV studies. The majority of environmental NMV studies do not get used by decision makers. Where they are used, they tend not to be used to make decisions, but rather to justify existing decisions. (Rogers et al. 2013, pp. 10–11)

Barriers to use

Based on the limited consultations undertaken for this paper, the available literature, and the Commission's experience with environmental policy issues, the main barriers to the use of non-market valuation to inform the development of policy appear to be:

- failure to apply a cost–benefit framework
- scepticism about stated preference methods
- opposition from vested interests
- lack of familiarity with the methods among decision makers
- time and cost requirements.

Failure to apply a cost–benefit framework

There is little prospect of non-market valuation becoming influential where a cost–benefit framework is not applied. This framework requires consideration of the value of the policy outcomes, including non-market outcomes, which is not required in multi-criteria analysis or most other approaches to decision making.

Environmental policy is very much concerned with concepts such as sustainability and the precautionary principle. There are good reasons for this given the irreversibility of some types of environmental degradation and the uncertainty surrounding the impact of many human activities. It is sometimes thought that properly taking these concepts into account is incompatible with applying a cost–benefit framework and this can work against acceptance of non-market valuation.

However, the extent of any incompatibility is unclear, and may be perceived to be greater than it actually is. This is because cost–benefit analysis can demonstrate the merits of governments taking action to address market failures that result in excessive damage to the environment. Through this process, the application of a cost–benefit framework can promote better environmental outcomes and the objective of sustainability (Markulev and Long 2013).

That said, there is an equity dimension to sustainability that may not be fully captured in a cost–benefit analysis that incorporates non-market values. Accordingly, governments may contemplate taking greater action than is indicated by such analysis. For example, while cost–benefit analysis can incorporate values that individuals place on future generations having access to particular environmental assets, governments may consider that they should place additional emphasis on the wellbeing of future generations. However, even here cost–benefit analysis can be useful to inform decision makers of the cost of pursuing intergenerational equity objectives.

A range of analysts have demonstrated that concerns about precaution can also be accommodated within a cost–benefit framework (Peterson 2006; VCEC 2009). For example, Hahn and Sunstein (2005, p. 6) argued:

> … cost–benefit analysis can and should incorporate concerns about precaution. For example, a problem characterized by irreversibilities … can be modelled using standard techniques in cost–benefit analysis. Uncertainties about both benefits and costs can also be incorporated, perhaps by specifying a range of possible outcomes, perhaps by seeking to preserve specified options, or perhaps by identifying the worst-case scenario and showing a degree of risk aversion with respect to that scenario.

Much depends on how terms are defined. There are many definitions of both sustainability and the precautionary principle, some of which are more compatible with a cost–benefit framework than others. This has led some analysts to call for greater guidance from Australian governments as to how these concepts should be applied (Peterson 2006; VCEC 2009). Such guidance might assist in overcoming this barrier to the use of non-market valuation.

Scepticism about stated preference methods

While some economists are supportive of non-market valuation being used to a greater extent, some remain sceptical, particularly about stated preference methods. For non-market valuation to gain greater traction economists would need to explain its use and demonstrate when and how it can be used effectively. The influence of economists is particularly important given their prominence within central agencies at both the Commonwealth and state and territory levels of government. That some economists remain unconvinced of the validity of the most widely applicable methods works against this process.

This paper concludes that, despite significant caveats, stated preference methods can produce valid estimates of value and have potential to improve environmental policy. The appropriate test is not whether they provide estimates that are as accurate as those based on market transactions, but whether they perform better than the available alternative ways of factoring in non-market outcomes. It is important to recognise that no method of estimating values is without flaws (including market demand and supply estimation).

That said, the evidence is not so clear cut that there is no room for disagreement about the merits of stated preference methods, particularly when applied to relatively unfamiliar environmental 'goods'. There is clearly an ongoing role for economists to scrutinise the application of these methods and identify problems.

Opposition from vested interests

As discussed in chapter 1, a potential benefit of non-market valuation is that it can reduce the potential for policy decisions to be unduly influenced by vested interests. Accordingly, it would be expected that some interest groups that have a degree of influence on policy decisions would oppose the use of methods that may provide evidence that does not support their preferred policy. Policy makers who have a preference for particular options may also be uncomfortable about the use of non-market valuation.

Lack of familiarity with the methods among decision makers

Rogers et al. (2013) conducted interviews with Australian decision makers, including staff from government and natural resource management bodies that were involved in decision making processes relating to the environment. They found that many of those interviewed had a 'profound lack of knowledge about non-market valuation' (Rogers et al. 2013, p. 11). Most were unable to name any non-market

valuation methods and only 37 per cent said that they had ever been exposed to non-market valuation in the course of making environmental decisions. This lack of familiarity is clearly a barrier to the use of non-market valuation. Rogers et al. (2013) suggested that non-market valuation researchers consider doing more to communicate their results and demonstrate how they could be used in decision-making frameworks.

Time and cost requirements

Non-market valuation studies can be reasonably expensive to conduct ($50 000 to $100 000, or more) and can take several months to complete. While it is appropriate that time and cost are taken into account in decisions about whether to commission studies, these factors also need to be weighed up against the potential benefits. There is broad acceptance that considerable time and money should be put into collecting environmental baseline data and other biophysical data that can be used in analysing environmental policies. By contrast, there would seem to be less appreciation of the gains available from generating valuation data.

It is also possible to reduce costs by using internet-based survey techniques, but this advantage needs to be weighed up against potential disadvantages. For example, face-to-face surveys may be preferable when it is necessary to convey information about unfamiliar or complex environmental outcomes.

The use of benefit transfer can substantially reduce the time and cost of obtaining value estimates. However, the range of circumstances where this can currently be done reliably is limited. Development of a broader set of value estimates would be needed to enable benefit transfer to become a viable option in a wider range of situations.

Realising the potential

There are a number of steps that could be taken to more fully realise the potential of non-market valuation to contribute to better environmental policy.

First, greater attention could be given to the quality of non-market valuation studies. As demonstrated in chapter 2 and appendix C, there are many potential problems that need to be avoided for a study to produce reliable estimates. Because of this, there is a danger of poor quality studies damaging the credibility of non-market valuation generally. Developing a more widespread understanding of what constitutes a high-quality study would help. Academic experts can play a useful role

in promoting high standards by reviewing studies undertaken by consultants and providing input on their design.

Second, there could be a better alignment of the research effort into non-market valuation with policy needs. As discussed, there is considerable academic research into non-market valuation. However, at present the incentives faced by researchers do not necessarily promote research that is aligned with policy needs. For example, there can be a strong incentive for academics to explore methodological innovations that may be of little practical importance for policy. Areas of research that might be worthy of greater attention include:

- how a strategic approach could be taken to conducting non-market valuation studies so that a policy-relevant evidence base is built up to support the use of benefit transfer

- whether there is merit in modifying current expert valuation methods to incorporate results from non-market valuation studies.

Finally, it may be worthwhile to develop greater knowledge about non-market valuation within relevant government agencies. This could assist in achieving the focus on study quality and the re-alignment of research effort discussed above. It might also assist in more effectively communicating the potential of non-market valuation to contribute towards better environmental policy.

A   Workshop participants

The authors wish to acknowledge the following people who participated in a workshop held at the Commission's Canberra office on 9 April 2013.

Participant	Organisation
Rosalyn Bell	Productivity Commission
Jeff Bennett	Australian National University
Marc Carter	Department of Sustainability, Environment, Water, Population and Communities
Drew Collins	BDA Group
Larry Cook	Productivity Commission
Siobhan Davies	Murray–Darling Basin Authority
Lisa Elliston	Australian Bureau of Agricultural and Resource Economics and Sciences
Noel Gaston	Productivity Commission
Jenny Gordon	Productivity Commission
Quentin Grafton	Bureau of Resources and Energy Economics
Jared Greenville	Productivity Commission
Lisa Gropp	Productivity Commission
Darla Hatton MacDonald	CSIRO
Jordan Louviere	University of Technology Sydney
Mark Morrison	Charles Sturt University
David Pearce	The Centre for International Economics
John Rose	University of Sydney
Peter Saunders	Department of Finance and Deregulation
Chris Toyne	Office of Best Practice Regulation

B   Australian studies

This appendix outlines several environmental non-market valuation studies that have been undertaken in Australia. For each study, it provides a summary of the environmental outcomes assessed, the methodology (including key assumptions), the results, and how the study influenced policy processes or outcomes. The studies in this appendix have been selected to illustrate how non-market valuation has been used in Australia, and do not form a comprehensive list. A larger number of Australian studies (over 100) are listed in the Environmental Valuation Reference Inventory (www.evri.ca) and the Envalue database (www.environment.nsw.gov.au/envalueapp).

B.1   River red gum forests

In 2005, the Victorian Environmental Assessment Council (VEAC) commenced an investigation into the management of river red gum forests across northern Victoria (VEAC 2008). This included consideration of the benefits of different land uses in the area (including agriculture, forestry, mining and conservation). One input to the investigation was a choice modelling study that estimated non-market values for environmental attributes relevant to these forests (Bennett et al. 2007). The study also estimated values relating to forests in East Gippsland (Victoria), which were not part of VEAC's investigation and so are not discussed in this appendix.

What environmental outcomes were assessed?

Bennett et al. (2007) estimated values for non-market environmental outcomes relating to river red gum forests in Victoria. These were intended to provide information on the benefits of improving environmental outcomes, which could then be compared with the associated costs (including changes in alternative land uses, such as forestry, mining and grazing). The analysis focused on the area of healthy forest, numbers of threatened parrots, and numbers of fish (including Murray Cod and other threatened species). These were selected following consultation with experts (including scientists, land managers and government agencies) and focus groups comprising members of the general public. The study also estimated the value that people place on the presence of campsites with

facilities (such as toilets and rubbish bins), which were considered a way to reduce the environmental impacts of visitors by discouraging camping elsewhere.

What methods were used?

The choice modelling was based on a survey that provided information about river red gum forests and their management (including maps and photographs), and which asked participants various questions about their attitudes towards the forests, their socioeconomic background, and which outcomes they would choose in choice sets. The choice sets comprised combinations of the attributes and levels set out in table B.1 (there were five separate survey designs, each of which contained five choice sets). Information in the survey stated that more frequent flooding of the forests, along with 'other changes to land uses', could improve their ecological condition. Additional questions were included to assess how participants interpreted the material in the survey, including questions about how well participants understood the information provided and whether they found the choice sets confusing.

Table B.1   **Attributes and levels — river red gum forests**

Attribute	Description	Levels
Healthy forests	Area of healthy river red gum forest (in hectares)	54 000; 67 000; 74 000; 80 000
Threatened parrots	Number of breeding pairs	900; 1200; 1500; 1800
Murray Cod and other threatened native fish	Percentage of numbers that existed before European settlement	10; 20; 40; 60
Recreation facilities	Number of campsites with facilities	6; 9; 12; 18
Cost	Compulsory annual payment (in dollars)	0; 20; 50; 100

Source: Bennett et al. (2007).

Bennett et al. (2007) distributed surveys to randomly selected households. Printed surveys were administered in November 2006 (using a 'drop off, pick up' approach) to households across several regions:

- Echuca, Mildura and Wodonga (situated near river red gum forests)

- Bairnsdale (in the Gippsland region)

- Melbourne.

Surveys in regional areas covered both urban and rural households. The Melbourne and Bairnsdale samples were used to elicit values for households living outside the region, in a major city and in a different regional area, respectively. A total of 1045 completed surveys were collected, with response rates of 81–88 per cent for

households within the region, 73 per cent for households in the Bairnsdale area, and 58 per cent for Melbourne households (Bennett et al. 2007).

Econometric models (conditional logit and nested logit) were then used to examine how the different levels of attributes affect participants' willingness to pay. These models included socioeconomic variables collected from the survey (such as gender and education). Implicit prices were then estimated for each environmental attribute.

What were the results?

The implicit prices that Bennett et al. (2007) calculated are estimates of the average participant's marginal willingness to pay, over a 20 year period, for a one-unit increase in the attribute (table B.2). There was some variation in values across the three samples, particularly for the value placed on healthy forests. The estimate for Bairnsdale households ($3.29 per 1000 hectares) is over twice that for Melbourne households ($1.45 per 1000 hectares), while the estimate for households within the same region as the forests was found not to be statistically different from zero. Other estimates varied across the samples, and in all cases the value that households place on campsites was found not to be statistically different from zero.

Table B.2   **Implicit price estimates for River Red Gum forest attributes**
2006 dollars per household per year

Attribute	Local sample (Echuca, Mildura and Wodonga)	Bairnsdale sample	Melbourne sample
Healthy forests (per 1000 hectares)	0.07	3.29 **	1.45 ***
Threatened parrots (per 100 breeding pairs)	3.96 ***	8.39 ***	4.39 ***
Murray Cod and other threatened native fish (per 1 per cent increase)	1.09 ***	1.37 ***	1.02 ***
Recreation (per number of campsites with facilities)	-0.24	-0.85	-0.11

** Significantly different from zero at the 95 per cent level. *** Significantly different from zero at the 99 per cent level.

Source: Bennett et al. (2007).

How has the study been used?

VEAC used the choice modelling results provided by Bennett et al. (2007) to assess the environmental benefits of its recommendations for land-use changes in northern Victoria. These concerned establishing new national parks and reducing the amount of land classified as state forest (which can be logged).

Two scenarios were examined.

1. Changes to land use without changes in environmental water flows to the forests.

2. The same changes to land use with increased environmental water (VEAC 2008).

These scenarios were compared to a baseline scenario of no policy change. The associated environmental benefits were valued using the estimated implicit prices, predictions of how environmental outcomes would change, and assumptions about how applicable these values are to the broader population.

- Changes in environmental outcomes were based on a projection that the area of healthy forest would rise from 54 000 hectares in the baseline to 64 000 hectares under scenario 1 and 80 000 hectares in scenario 2.

- It appears that VEAC calculated population-wide aggregates by assuming that non-respondents to the survey would not benefit from either policy scenario (similar to an illustrative aggregation by Bennett et al. (2007)). Per-household estimates were multiplied by the number of households in the broader population (for regional urban and rural areas, and for Melbourne) then multiplied by the survey response rate.

This led to estimates of the community's willingness to pay for the associated environmental benefits of $37.9 million and $107.4 million (per annum) for scenarios 1 and 2 respectively (VEAC 2008). Most of these benefits (around 96 per cent) were projected to accrue to households outside of the region.

These values were used as part of a broader cost–benefit analysis. This evaluated the costs and benefits of VEAC's recommended land-use changes, and is set out in table B.3 (further details of the assumptions and calculations used are provided by Gillespie Economics, DCA Economics and Environmental & Resource Economics (2008)). Some of these figures were based on 'benefit transfer' from past non-market valuation studies. The figures are presented in terms of annual costs and benefits (average values over the following 20 years) rather than net present values. Annual net benefits were estimated at around $37 million for scenario 1 and $107 million for scenario 2, excluding the cost of providing additional environmental water.

Table B.3   **Cost–benefit analysis of VEAC recommendations**

Measure of change under policy	Estimated value, scenario 1	Estimated value, scenario 2	Comments
	$m per year	$m per year	
Benefits			
Non-market environmental benefits (forests, birds, fish)	37.90	107.42	Based on the choice modelling study by Bennett et al. (2007)
Increased protection of wetlands	0.60	0.66	Based on past contingent valuation and choice modelling studies of Victorian wetlands
Increased protection of riparian areas	2.34	2.34	Based on past choice modelling studies of Victorian riparian areas
Increased tourism and recreation	0.87	0.87	Based on past travel-cost studies and estimates of increased visitation following creation of national parks
Impacts on indigenous and non-indigenous cultural heritage	ne	ne	
Total benefits	*41.72*	*111.29*	
Costs			
Additional park management	1.00	1.00	Based on past estimates of management costs following creation of national parks
Reduction in timber harvest	1.36	1.25	Based on market values obtained from a survey of timber companies
Reduction in grazing in Barmah forest	0.14	0.14	Based on market values obtained from past surveys of graziers
Reduction in grazing in riparian areas	0.76	0.76	Based on market values obtained from past surveys of graziers
Increased costs in riparian areas (fencing, watering points, pest control)	0.87	0.87	Based on market values obtained from past surveys of graziers
Reduction in duck hunting	0.55	0.49	Based on estimates of per-trip consumer surplus transferred from a past travel-cost study
Willingness to pay for maintaining rural communities	0.16	0.16	Based on past choice modelling estimates of the value of maintaining rural populations
Cost of providing additional environmental water	0	ne	
Total costs	*4.84*	*4.66*	
Net benefit	**36.88**	**106.63**	

ne Not estimated.

Sources: Gillespie Economics, DCA Economics and Environmental & Resource Economics (2008); VEAC (2008).

Policy outcomes

VEAC's analysis indicated that adopting its proposed recommendations would have a net benefit for Victoria, once non-market environmental values were taken into account (VEAC 2008). The Victorian Government (2009) supported the recommendations and established four new national parks in the region. In this way, the non-market valuation study was used to support a policy proposal, even though it appears to have been used to value costs and benefits after policy recommendations had been formed, rather than as a direct input to those recommendations.

B.2   The Murray–Darling Basin

Under the *Water Act 2007* (Cwlth), the Murray–Darling Basin Authority is required to develop a Basin Plan that sets out how water resources are allocated across uses in the Basin. A key part of this plan is limits on the volume of water that can be extracted from the Murray–Darling river system ('sustainable diversion limits'), which must be set with regards to the environmental, social and economic implications. Following the release of a draft Basin Plan in 2011, the Authority examined the likely socioeconomic and environmental impacts. This assessment — which informed the final Basin Plan — drew on a number of studies, one of which provided estimates of the non-market values the community would place on environmental improvements in the Basin (Morrison and Hatton MacDonald 2010).

What environmental outcomes were assessed?

Morrison and Hatton MacDonald (2010) examined several environmental attributes across the Murray–Darling Basin, including the area of healthy native vegetation, numbers of native fish, the frequency of colonial waterbird breeding events, and numbers of birds. The study covered 18 regions of the Basin. (Some estimates of recreation values were also reported, but are not discussed in this appendix.)

What methods were used?

The study involved drawing together available non-market valuation estimates across different regions of the Basin to provide a set of values for environmental attributes in each region. Excluding recreation studies, nine primary choice-modelling studies covering various regions of the Basin were used in the analysis (published over the period 2001–2010). Where estimates were not available

from a primary study, benefit transfer was used to impute these values (with the exception of waterbird breeding, which does not occur in all regions).

The process of benefit transfer involved the application of estimates to new areas and the conversion of estimates into comparable units (dollars per household, expressed in present-value terms). Estimates were transferred across contexts in per-unit terms (for example, dollars per percentage point increase in vegetation or fish numbers). Key assumptions reported in the study are listed below. Some of these are based on available evidence; others are closer to 'rules of thumb' where clear evidence was not necessarily available.

- Where available estimates were expressed as annual payments, these were converted to present values using a 28 per cent discount rate (which was selected based on experimental research on rates of time preference).

- A single value was used where primary studies reported separate estimates for different populations (such as urban and rural households). This was an average, weighted by the size of each population.

- Weighted average values were used for attributes of the Macquarie Marshes, for which multiple primary studies had been undertaken.

- Values for the Goulburn River were transferred to the Campaspe, Loddon, Ovens and Moonie regions (for which primary estimates were not available) and reduced by one third (to reflect the greater size of the Goulburn River and its relative proximity to Melbourne) (Morrison and Hatton MacDonald 2010).

Morrison and Hatton MacDonald (2010) also estimated aggregate values for each environmental attribute. This was based on the assumption that 30 per cent of non-respondents in a valuation survey would share the same average values as respondents, with the remainder having a zero valuation (based on a study by Morrison (2000)). For the Murray River, the relevant population (that values the environmental attributes) was the total number of households in Australia. For other regions, the number of households in the same state was used.

What were the results?

The implicit prices (per household) are reported in table B.4 for each region. Some estimates are the same across multiple regions because the one primary study was used.

Table B.4   Implicit price estimates — Murray–Darling regions
2010 dollars per household, present value

Region	Native vegetation	Native fish	Colonial waterbird breeding	Waterbirds and other species
	per 1% increase in healthy native vegetation	per 1% increase in native fish populations	per 1 year increase in frequency of breeding	per unit increase[a] in number of waterbirds and other species present
Barwon–Darling	2.26	0.46	13.87	2.25
Border Rivers	2.19	0.46	ne	1.10
Campaspe	5.69	5.06	ne	3.89
Condamine–Balonne	2.63	0.46	13.87	1.10
Mt. Lofty Ranges	5.69	5.06	ne	3.89
Goulburn–Broken	5.69	5.06	ne	3.89
Gwydir	2.19	0.46	13.87	1.10
Lachlan	2.19	0.46	13.87	1.10
Loddon–Avoca	5.69	5.06	ne	3.89
Macquarie–Castlereagh	2.19	0.46	33.08	1.10
Moonie	2.63	0.46	ne	1.10
Murray	13.72	12.80	65.11	3.43
Murrumbidgee	2.26	0.46	13.87	2.25
Namoi	2.19	0.46	ne	1.10
Ovens	5.69	5.06	ne	3.89
Paroo	2.63	0.46	13.87	1.10
Warrego	2.63	0.46	ne	1.10
Wimmera	2.19	0.46	ne	1.10

a 'Unit increase' is not specifically defined in the study. **ne** Not estimated.

Source: Morrison and Hatton MacDonald (2010).

Aggregated values (across all relevant households) are reported in table B.5, based on the assumptions outlined above. Morrison and Hatton MacDonald (2010) also calculated aggregates under different sets of assumptions (not reported in this appendix).

Table B.5 **Aggregate values — Murray–Darling regions**
Thousands of 2010 dollars, present value

Region	Native vegetation	Native fish	Colonial waterbird breeding	Waterbirds and other species
	per 1% increase in healthy native vegetation	per 1% increase in native fish populations	per 1 year increase in frequency of breeding	per unit increase[a] in number of waterbirds and other species present
Barwon–Darling	3 594	667	24 693	3 578
Border Rivers	2 437	414	ne	1 086
Campaspe	3 363	2 990	ne	2 299
Condamine–Balonne	2 926	414	15 337	1 086
Mt. Lofty Ranges	1 494	1 329	ne	1 022
Goulburn–Broken	5 019	4 463	ne	3 431
Gwydir	3 482	667	24 693	1 749
Lachlan	3 482	667	24 693	1 749
Loddon–Avoca	3 363	2 990	ne	2 299
Macquarie–Castlereagh	3 482	667	58 802	1 749
Moonie	1 961	277	ne	728
Murray	79 098	73 794	375 369	12 203
Murrumbidgee	3 594	667	24 693	3 578
Namoi	3 482	667	ne	1 749
Ovens	3 363	2 990	ne	2 299
Paroo	2 598	414	15 337	1 086
Warrego	2 598	414	ne	1 086
Wimmera	2 660	509	ne	1 336

a 'Unit increase' is not specifically defined in the study. **ne** Not estimated.

Source: Morrison and Hatton MacDonald (2010).

How has the study been used?

The Murray–Darling Basin Authority (MDBA 2011) drew on the research by Morrison and Hatton MacDonald (2010) in its socioeconomic analysis of the draft Basin Plan. This was one input among many to the final Basin Plan, in which an overall surface-water sustainable diversion limit was set to allow for approximately 2750 gigalitres of additional water to be set aside for the environment each year (from 2019). Estimates of non-market values were used, to some extent, in the socioeconomic analysis and subsequent regulation impact statement for the final Basin Plan (MDBA 2012). These included estimates of the environmental use values of the Murray–Darling Basin (including to tourism, recreation, fishing and agriculture) along with the non-use (or existence) values provided by Morrison and Hatton MacDonald (2010). This assessment was conducted by the CSIRO (2012).

The CSIRO (2012) combined these estimates of non-use values with an analysis of the likely change in ecological outcomes due to the Basin Plan to provide figures for the total environmental benefits. For simplicity, this was based on the assumption of a linear relationship between ecological responses and economic values. This yielded an estimate of $3.4 billion (in present-value terms) for non-use environmental benefits under a scenario with an additional 2800 gigalitres of water set aside for the environment each year (close to the final level decided), relative to a baseline scenario where water management arrangements were not changed.

The CSIRO's analysis also included estimates of the value of improving the quality of waterbird habitat in the Coorong wetland. This drew on a choice-modelling study by Hatton MacDonald et al. (2011) that estimated a value of $4.3 billion (in present-value terms) for improving the quality of waterbird habitat in the wetland from 'poor' to 'good'. This figure was scaled downwards by the CSIRO — based on the proportion of years that the wetland is likely to be in a 'good' state — to obtain an estimate of the incremental value of improving the Coorong under the 2800 gigalitre scenario of $0.48 billion (CSIRO 2012).

Policy outcomes

The regulation impact statement for the Basin Plan noted that non-use benefits are likely to be significant, but treated estimates of these as illustrative. Concerns about the reliability and applicability of estimates provided by Morrison and Hatton MacDonald (2010) and the CSIRO (2012) were raised. The Authority stated that:

> The levels of improvement in environmental condition that underpin the estimates have been derived from a small number of sites; they make simplifying assumptions about links between hydrological changes and ecological outcomes; and many value estimates are 'transferred' from other studies — which were not designed to value the changes associated with the Basin Plan. Given these limitations, the estimates are best considered as indicative only, and should be considered together with other measures (for example, environmental outcomes) of the benefits of the Basin Plan. (MDBA 2012, p. 47)

However, the Authority concluded that:

> The evidence on the value of the use and non-use environmental benefits … suggests that even if only those examples of benefits of the Basin Plan that can be estimated in monetary terms are considered, and allowing for uncertainty inherent in the estimates, these benefits are of a comparable scale to the costs of the Basin Plan. (MDBA 2012, p. 77)

Quantifiable costs consisted of an estimated $160 million in forgone profits each year (from water-using activities) and administrative costs of around $100 million. Overall, the Authority concluded that 'the benefits of the Basin Plan are likely to

outweigh the costs' (MDBA 2012, p. 77), in terms of environmental, economic and social impacts that the Authority was required to consider under the *Water Act*.

B.3 Television and computer recycling

In 2008, the Environment Protection and Heritage Council (which has since been replaced by the COAG Standing Council on Environment and Water) commissioned a choice-modelling study to examine the value that the community places on recycling electronic waste ('e-waste'). Specifically, this study estimated the amount that urban households would be willing to pay for government intervention to increase the percentage of televisions and computers that are recycled rather than disposed in landfill (URS 2009).

What environmental outcomes were assessed?

The study sought to estimate the non-market value that people place on the knowledge that waste (of discarded televisions and computers) is avoided and materials are re-used, excluding any market value that the recovered material may have. This was framed in terms of recycling rates, rather than environmental or health outcomes associated with waste.

What methods were used?

URS (2009) designed a survey to elicit the community's willingness to pay for increased rates of recycling. This included information on recycling processes, along with questions on attitudes towards recycling, household recycling practices and demographics. It also contained choice sets based on the attributes and levels set out in table B.6. The cost attribute was the additional cost incurred when purchasing a new television or computer (on the assumption that the costs of the scheme would be reflected in higher consumer prices).

The survey provided only limited information about the number of televisions and computers disposed in landfills and the materials contained in them. The survey did not describe the environmental and health risks of landfills (or the likelihood that increased recycling rates would reduce such risks). This information was not provided partly to keep the survey short and partly because it was considered to be controversial (URS 2009).

Table B.6	**Attributes and levels — electronic waste recycling**	
Attribute	*Description*	*Levels*
Recycling rate	Percentage of disposed material recovered	1; 50; 70; 90
Cost	Additional cost per item purchased (in dollars)	0; 10; 20; 40; 60
Collection method	How televisions and computers are disposed of by households	Kerbside (items collected from households); drop-off (households take items to a recycling facility); none (status quo)

Source: URS (2009).

The survey was administered online in January 2009 to households in Melbourne, Sydney, Adelaide, Perth and Brisbane. A total of 2623 surveys were collected, of which 2105 were used for the analysis (out of 24 508 survey invitations that were sent out). Quotas were applied for the number of surveys accepted from each age, gender and income group to achieve a sample that was broadly representative of the population. Incomplete or inconsistent surveys were dropped (for example, where participants completed the survey in less than 6 minutes). Overall, completed surveys were collected from 10.7 per cent of people that were invited to participate.

Responses to a follow-up question (that asked participants who always selected the status quo option in choice sets) suggested that some people were submitting 'protest' responses in the survey. This may have been the case for participants that indicated they objected to paying for increased recycling, or believed that governments should pay (URS 2009). These responses were retained in the sample after analysis showed that they did not have a statistically significant impact on estimates of average willingness to pay.

Econometric methods (a random-parameter logit model) were then used to estimate the impact of various factors on participants' choices and their willingness to pay for increases in the recycling rate and for different collection methods. The analysis assumed a linear relationship between increases in the recycling rate and willingness to pay, based on tests for non-linearity in the range of possible future recycling rates (50–90 per cent) used in the surveys.

What were the results?

URS (2009) estimated willingness to pay for an average household using two metrics.

- The additional amount that a household would be willing to pay per new television or computer (in higher prices) to increase the recycling rate by *one* percentage point. This was estimated at $0.50 (in 2009 dollars).

- Willingness to pay (in higher appliance prices, per household) for increases in the recycling rate to 50, 70 or 90 per cent (table B.7).

Table B.7 **Implicit price and aggregate value estimates — recycling**
2009 dollars

Scenario	Implicit price	95% confidence interval	Aggregate
	$ per item	$ per item	$m over 5 years
One percentage point increase in recycling rate	0.50	0.43 – 0.56	3.6 – 4.2
Increase in recycling rate to 50 per cent	21.14	18.18 – 23.68	159.9
Increase in recycling rate to 70 per cent	29.77	25.60 – 33.34	225.2
Increase in recycling rate to 90 per cent	38.40	33.02 – 43.01	290.5

Source: URS (2009).

These estimates concern large changes in the recycling rate, from 1 per cent in the status quo (with no policy change) to 50–90 per cent in the choice sets. URS (2009) cautioned that it would only be appropriate to use the estimates to assess policy changes where the recycling rate exceeds 50 per cent.

In total, 85 per cent of participants expressed some willingness to pay for additional recycling. The variation in estimates across cities in the sample was found not to be statistically significant.

Estimates were also aggregated over the population of the five cities where surveys were distributed (Melbourne, Sydney, Adelaide, Perth and Brisbane). Two scenarios were used to do this.

1. Assuming that all households have the same average willingness to pay as survey participants.

2. Assuming that some households have the same average willingness to pay as survey participants, but others have a zero willingness to pay. The proportion of households in the latter category was set equal to the percentage of people who started the survey but did not complete it (13.7 per cent).

Estimates based this second scenario are reported above in table B.7. It was estimated that the population of the five cities would be willing to pay between $3.6 million and $4.2 million per a one percentage point increase in the recycling rate. These amounts are over the following five years, and are based on the number of televisions or computers that participants said they expected to purchase over that period.

In addition to these scenarios, sensitivity analysis was carried out based on income and education levels, which in the survey results were not as representative of

census data as other socioeconomic variables. This involved estimating average willingness to pay for each of three income groups and three education levels in the sample, weighted by the proportion of the overall population in each category. (Results of this analysis are not replicated in this appendix.)

URS (2009) also estimated that households would be willing to pay, on average, an additional $3.55 per item to have discarded televisions and computers collected from the kerbside rather than having to take them to a recycling facility. This estimate largely reflected a relatively high value placed on kerbside collection by households in Sydney and Perth (estimates for households in the other cities were not statistically significantly different from zero). The aggregate value placed on kerbside collection over all households was estimated at $23.2 million over 5 years (using the second aggregation scenario described above) (URS 2009).

How has the study been used?

This was one of two choice-modelling studies commissioned by the Environment Protection and Heritage Council following concerns about resource conservation, litter and the amount of waste sent to landfill. (The other focused on a container-deposit scheme, but did not lead to a change in government policy after it was found that the costs were likely to exceed any benefits.) The study was followed by a regulation impact statement (RIS) that examined options for increasing the number of televisions and computers that are recycled (PricewaterhouseCoopers and Hyder Consulting 2009).

The cost–benefit analysis in the RIS used the aggregate willingness to pay estimates from URS (2009) as the sole measure of benefits of increased recycling rates. This was done under the assumption that the willingness to pay expressed by survey participants would include the value that they place on any recovered materials, avoided environmental or health impacts of landfills, avoided land costs for landfills, and any change in the amount consumers would pay for rubbish collection (PricewaterhouseCoopers and Hyder Consulting 2009). The analysis was presented in present-value terms, adjusting for projections of future household numbers and appliance purchases. Benefits were only counted for the year 2015-16 onwards, where the recycling rate was projected to exceed 50 per cent.

The RIS examined several policy options. These consisted of various combinations of regulation, industry-run recycling schemes, levies and subsidies. The net benefit of the preferred policy option — where television and computer manufacturers and importers are jointly responsible for collecting and recycling all end-of-life products under an industry scheme — were estimated at $649 million in present-value terms

(2009 dollars) (URS 2009). The total benefits were estimated at just over $1.5 billion, and the costs at $873 million. These costs mainly consisted of the costs to industry of collecting, transporting and processing waste (estimated at $973 per tonne of waste), plus some administrative costs of the scheme.

The choice-modelling estimates were central to the analysis. A separate calculation showed that excluding these estimates and instead using only other available measures of benefits (for example, of the market value of recovered materials and avoided landfill costs) would mean that the costs exceed the benefits for all policy options considered (PricewaterhouseCoopers and Hyder Consulting 2009).

Policy outcomes

Following consideration of the RIS, the Australian Government established the National Television and Computer Recycling Scheme. Under this scheme, importers and manufactures of televisions, computers and computer products (above a threshold) are required to join a co-regulatory arrangement where an industry-established body must recycle e-waste from households and small businesses free of charge, regardless of the brand or age of the equipment (this scheme is similar to the preferred policy option in the RIS) (DSEWPC 2013). The scheme is aimed at increasing the recycling rate for e-waste from around 17 per cent in 2010 to 30 per cent in 2012-13 and 80 per cent by 2022. The scheme commenced in 2012.

B.4 Bulli Seam coal mining

In New South Wales, a change in planning regulations in 2005 meant that existing coal mines that had not already undergone formal development approval were required to do so by December 2010 to continue operating (Gillespie and Kragt 2012). Part of the approval process involved submitting an Environmental Assessment that detailed the likely environmental, social and economic impacts of continued mining.

In several cases, non-market valuation was used to weigh up the environmental and social costs of mining with the economic benefits. This section focuses on the analysis of the Bulli Seam Operations as an illustrative example. A choice modelling study (Gillespie Economics 2009b) was conducted and formed part of the development assessment for whether long-wall coal mining should be permitted to continue for the next 30 years at the Appin Mine and West Cliff Colliery in the Southern Coalfields, near Wollongong.

What environmental outcomes were assessed?

Gillespie Economics (2009b) examined several environmental impacts of underground coal mining, selected by drawing on available evidence and community focus groups. Most of these impacts related to the effects of mine subsidence (the vertical or horizontal movement of the land surface due to underground mining). A key attribute was length of streams affected by stream-bed cracking (due to subsidence), which is associated with the draining of pools in streams, reduced water flows, iron staining of streams and ecological disturbance. These impacts were not separately included in the analysis as the high degree of correlation between them could reduce the precision of the results. Subsidence impacts on upland swamps were not valued in the study as these impacts were considered to be negligible.

The study also assessed the value that the community places on protecting Aboriginal cultural sites (such as grinding groove sites, engraving sites, rock art and artefacts) that could be damaged by mine subsidence, including rock cracking or rock falls. In addition, the non-market costs of clearing native vegetation and the non-market social benefits of employment at the mine facility were estimated (Gillespie Economics 2009b).

What methods were used?

A choice-modelling survey was used to assess the community's willingness to pay to reduce negative impacts of coal mining. This consisted of a description of current mining activities, the potential impacts of subsidence on streams and Aboriginal heritage sites, the area of native vegetation that could be affected by above-ground infrastructure, and employment projections should mining continue.

The survey presented participants with several choice sets — based on the attributes and levels in table B.8 — and asked demographic questions (Gillespie Economics 2009b). Two cost attributes were used (across different versions of the survey). These were described to participants as a one-off or annual environmental levy that must be paid to the NSW Government to replace forgone mining royalties if coal mining were to be curtailed or terminated at the site. The choice sets were followed by questions that asked whether participants understood the information provided or found the choice sets confusing.

Table B.8   **Attributes and levels — Bulli Seam coal mining**

Attribute	Description	Levels
Streams	Kilometres of stream affected by cracking of stream beds	40; 60; 80; 100
Native vegetation	Hectares of native vegetation cleared	240; 290; 330; 380
Heritage sites	Number of Aboriginal heritage sites affected by subsidence	20; 30; 40; 50
Employment	Number of years that mining would directly provide 1170 jobs	1; 11; 21; 31
Cost	A compulsory one-off payment made by households (in dollars)[a]	0; 125; 300; 625

a Some versions of the survey instead used an annual payment (to be made over 20 years). The levels of this version of the cost attribute were not reported in the study.

Source: Gillespie Economics (2009b).

The survey design also reflected criticisms that the NSW Planning Assessment Commission (2009) had made about the use of choice modelling in an earlier coal-mining proposal. In particular, the Commission noted that damage to streams arises from a combination of all mines in the area, rather than from one specific mine. It argued that it would be more appropriate to examine people's willingness to pay for avoiding environmental damage in the wider area (the Southern Coalfield), which 'would lead to higher environmental value estimates because marginal values of goods increase as their supply becomes relatively more limited' (NSW Planning Assessment Commission 2009, p. 110). As such, it recommended that future choice modelling should use 'split sampling' (two samples, each with a slightly different survey) to examine whether results would vary over different levels of the environmental goods.

In response, Gillespie Economics (2009b) prepared a statement about the likely cumulative impacts of all mines in the Southern Coalfield on streams over the following 31 years. Two versions of the survey were developed, one with this statement and one without ('full context' and 'partial-impact context' respectively), to test for any differences in results. Split sampling was also used to test for differences between payment methods, with some surveys using a lump-sum cost attribute in choice sets (as described in table B.8), and others using annual payments over a 20-year period.

The survey was administered online in May–June 2009 to households situated close to the Southern Coalfield (in the Illawarra and Outer South West Sydney regions), as well as to the NSW population more broadly. Due to the split sampling, four subgroups were sampled.

- Households in the region, with the full context description and lump-sum cost attribute.

- NSW households more broadly, with the:

- full-context description and lump-sum cost attribute

- partial-impact context description and lump-sum cost attribute

- full-context description and annual-payment cost attribute.

A total of 4688 surveys were completed, out of 24 966 invitations that were distributed (a response rate of 18.7 per cent). Of the completed surveys, 2917 were used for the analysis (after some were discarded to ensure approximately equal numbers for each subgroup). While Gillespie Economics (2009b) noted that the age and gender distribution of the samples were broadly in line with the NSW population, there were a disproportionately high number of high-income households in the sample, which could bias estimates upwards.

Econometric techniques (conditional logit and random-parameter models) were then used to estimate willingness to pay for each attribute.

What were the results?

Several econometric models were estimated for each subgroup, with random-parameter models considered to be most appropriate following a series of statistical tests. Implicit prices for each attribute were compared across the subgroups. The only statistically significant difference between the regional population and broader NSW population was for the employment attribute (at a 95 per cent level of significance). There were no significant differences between samples given different descriptions of the environmental context (the full-context and partial-impact context versions of the survey). The type of payment method (lump-sum or annual) was found to significantly affect estimates for all attributes.

These comparisons led Gillespie Economics (2009b) to prefer the implicit price estimates for NSW households with the full-context description and lump-sum cost attribute (these were generally the lowest estimates). These estimates were used in the subsequent analysis and are set out in table B.9. It was estimated that the average household would be willing to pay around $4.73 to protect one kilometre of stream from stream-bed cracking, and $0.90 to avoid damage to one hectare of native vegetation. The estimate for avoiding damage to Aboriginal heritage sites was $5.15 per site. The social benefits of employment were estimated at $26.90 per household for each year that the mine directly provides 1170 jobs.

Table B.9 **Implicit price and aggregate value estimates — Bulli Seam**
2009 dollars (lump-sum amounts)

Attribute	Implicit price	95% confidence interval	Aggregate
	$ per household	$ per household	$m
Avoided cracking of stream beds (per kilometre of stream)	4.73	2.81 – 6.65	5.46
Avoided damage to native vegetation (per hectare)	0.90	0.05 – 1.82	1.04
Avoided damage to Aboriginal heritage sites (per site)	5.15	1.52 – 8.84	5.94
Avoided job impacts (per year that the mine would directly provide 1170 jobs)[a]	26.90	na	31.03

a Figures for this attribute are averages reported by the study (the model used the natural log of the number of years). Confidence intervals were not reported. **na** Not available.

Source: Gillespie Economics (2009b).

Aggregate estimates for the whole NSW population were also calculated (table B.9) by multiplying the per-household figures by the number of NSW households and by an 'adjusted response rate'. This rate was 45.8 per cent (that is, it was assumed that 45.8 per cent of NSW valued the outcomes), calculated using the response rate for the survey (18.7 per cent) and an assumption that one-third of non-respondents would have similar values to respondents (based on an earlier study by Morrison (2000)).

How has the study been used?

This study was used in a cost–benefit analysis of allowing mining of the Bulli Seam to continue, also conducted by Gillespie Economics (2009a). This formed part of the environmental assessment for the project. The net benefit of continued mining was estimated at $8.28 billion in present-value terms.

This net benefit comprised several components. The estimated net benefit of production (accruing to the mine operator) was $10.3 billion (in present-value terms), taking account of the estimated value of coal and the opportunity costs of establishing the mine. Externalities associated with mining were also considered, including the impacts of surface operations, underground mining, road transport and employment.

In total, negative externalities were estimated at $2.9 billion, including $368 million in stream impacts (subsidence), $188 million in Aboriginal heritage impacts and $112 million in ecological impacts (Gillespie Economics 2009a). These values were based on the choice-modelling study (Gillespie Economics 2009b). The remaining

costs were for upland swamp impacts and greenhouse gas emissions from mining, which were estimated using secondary sources. Positive externalities were estimated at $870 million, representing the social benefits of employment (drawing on the choice-modelling study).

The cost–benefit analysis also examined alternative project options containing various setbacks of mining longwalls from environmental features (including streams, upland swamps, vegetation and Aboriginal heritage sites). All of these cases were estimated to have lower net benefits than the main proposal discussed above. In particular, any reduction in the value of coal mined from the site was estimated to exceed the value the community places on protecting environmental features (Gillespie Economics 2009a). Estimates of these environmental values, drawn from the choice-modelling study, applied across the area as a whole (as the study did not estimate separate values for different parts of the site, such as streams on one side compared to the other).

The main conclusion of the cost–benefit analysis — an estimated net benefit of continued mining — was robust to changes in key variables, such as operating costs, non-market impacts and discount rates (as demonstrated through sensitivity analysis) (Gillespie Economics 2009a). In other words, the inclusion of non-market value estimates did not suggest that a reduction in mining would have a net benefit for the community.

Policy outcomes

The development application for the Bulli Seam coal mines, including the environmental assessment, were examined by a panel established by the NSW Planning Assessment Commission (2010) to inform the approval process. This panel expressed general support for the use of choice modelling to provide indicative estimates of the environmental and social costs of proposals.

However, the panel also expressed reservations about how non-market valuation had been used in this case. These primarily related to the information presented in the choice-modelling survey (NSW Planning Assessment Commission 2010). The panel noted that the environmental impacts assumed to be associated with the project differed from those predicted by government agencies and other stakeholders (and were generally less severe). It also criticised the study for providing average valuation estimates for the environmental and heritage impacts when these might differ significantly across the project area. In particular, the panel posited that the costs to society are likely to be greater on the relatively pristine eastern side of the site than on the partly developed western side.

The panel argued that these aspects may have resulted in values being underestimated. It further contended that the estimates may be understated because they do not take into account option values (where people may be willing to pay to protect environmental or heritage sites from development if there is uncertainty about future outcomes) or the value of future recreation in the area.

The panel concluded that:

> … it is likely that net social benefits will be achieved by the imposition of selective approval conditions that are designed to protect aggregations of special environmental and heritage features where the biophysical impacts of mining are uncertain. (NSW Planning Assessment Commission 2010, p. 384)

It recommended that mining activities be conditionally approved in the western and northern areas of the site, with a stricter approach for the eastern and southern areas (where any approval would need to be subject to strict criteria being met). In particular, it put forward an approach based on providing 'adequate protection' to 'known aggregations of significant natural features' (NSW Planning Assessment Commission 2010, p. 390). This was based on its conclusion that the benefits of protecting 'significant natural features' in the area are likely to be of similar magnitude to the mining profits that would be forgone to protect these features. The panel also recommended that impacts on sites of special significance should be negligible (including Aboriginal heritage sites, waterways and cliffs).

Approval for the project was granted in December 2011, subject to several conditions. These included the removal of mining operations from three zones in the eastern and southern areas of the site (considered to contain significant natural or heritage features), which comprised around 40 per cent of the original mining proposal (NSW Planning Assessment Commission 2011).

B.5 Underground power supply

In 2011, the Economic Regulation Authority (ERA) of Western Australia conducted an inquiry into the costs and benefits of the State Underground Power Program (SUPP). This program involves replacing existing overground power distribution infrastructure (mainly wires, poles and transformers) with underground power supply in residential areas. From the commencement of the SUPP in 1996 to the time of the inquiry, around 10 per cent of the electricity distribution network in the Perth metropolitan area had been placed underground through the program (ERA 2011).

The ERA commissioned a hedonic pricing study to estimate the impact that underground power supply has on house prices (Marsden Jacob Associates 2011). Although the SUPP is not, strictly speaking, an environmental policy, it is discussed in this appendix as an example of how visual amenity has been considered in policy analysis.

What environmental outcomes were assessed?

The hedonic pricing study sought to quantify the extent that underground power supply increases residential property prices in the Perth metropolitan area (Marsden Jacob Associates 2011). This impact may reflect several non-market benefits to property owners (ratepayers) from moving from overground to underground power supply. For example, the ERA (2011) assumed that the main impact of retrospective underground power would be improved visual amenity of streetscapes and suburbs. It also noted that property owners would benefit from improved reliability and quality of electricity supply, reduced costs of pruning vegetation (that encroaches on overground power lines) and safer street lighting (as new streetlights are also installed under the SUPP) (ERA 2011).

What methods were used?

Marsden Jacob Associates (2011) used data covering a range of attributes to examine changes in metropolitan Perth house prices over the period 2000–2010. These data covered properties with and without underground power supply. However, information on the type of power supply was only available for individual properties that were covered by previous SUPP projects. For the remainder, it was assumed that all houses in 'greenfield' suburbs where most houses were constructed after 1995 have underground power (along with houses in older streets that had been redeveloped after 1995). All other houses (not covered by SUPP projects) were assumed to have overground power.

Several other variables were used in the analysis. These included, for each property, the:

- sale price
- age of the property when it sold
- number of rooms
- land area
- distance from the coast

- distance from Perth central business district

- distance from specific attributes, such as beaches and waterways (estimated using Geographic Information System software) (Marsden Jacob Associates 2011).

These variables were selected to avoid a high degree of correlation between pairs of variables (such as 'number of rooms' and 'number of bedrooms'), which can reduce the precision of estimates (Marsden Jacob Associates 2011). In addition, variables that were not statistically significant in the analysis were removed from the model.

The dataset covered sale transactions for 786 228 residential properties over the period 2000–2010. 'Outliers' were removed to address apparent coding errors and ensure that the modelling results would not be overly sensitive to a small number of very high or very low observations. Several criteria were used to identify and remove outliers. Among other factors, these covered houses reported as:

- having fewer than four or more than 18 rooms

- having sold for under $100 000 or over $2 million

- having a sale price within the highest or lowest 2.5 per cent for their suburb

- being in suburbs with fewer than 100 transactions over the 10-year period (Marsden Jacob Associates 2011).

Marsden Jacob Associates (2011) analysed the data on a quarterly basis over the 10-year period (that is, 40 quarters). Regression modelling was used to examine how the variables of interest affect house prices. The logarithm of house prices was modelled as a function of house characteristics over time (a log-linear model). Specifically:

$$P_{i,t} = \sum_{t=1}^{T} \alpha_t D_{i,t} + \beta_t X_{i,t} + \varepsilon_{i,t}$$

where $P_{i,t}$ is the logarithm of the price of house i at time t, $D_{i,t}$ takes a value of one if the house sold in time period t and zero otherwise, $X_{i,t}$ represents house characteristics, $\varepsilon_{i,t}$ is an error term and β is the vector of coefficients to be estimated. This functional form allowed the model to take account of repeat sales of the same house over the period, which included a significant increase in Perth property prices from 2003 to 2007 (Marsden Jacob Associates 2011).

What were the results?

The study estimated that the average implicit price of underground power was $9962 per property (in 2011 dollars), with a standard deviation of $2613 (Marsden Jacob Associates 2011).[1] This implicit price was equivalent to 1.6 per cent of the average residential property price. It represents the incremental value of underground power (as capitalised in house prices), on average, across residential properties in Perth (including, but not limited to, those that had been covered by SUPP projects).

However, the implicit price depends on other housing attributes (such as the location or age of a property), and may vary considerably across houses or suburbs. To illustrate this, Marsden Jacob Associates (2011) also conducted the analysis for houses in four separate price brackets (table B.10). The implicit price was found to be larger for higher-value properties, indicating that the benefits of underground power are greater for these properties. This could be due to higher levels of overall amenity (such as better views) in areas with high property values, such that the form of power supply would have a greater impact on amenity than in other areas (Marsden Jacob Associates 2011).

Table B.10   **Implicit price estimates for underground power lines**

House sale value	Percentage of properties in sale-value range[a]	Mean implicit price	
$	%	$	% mean sale price
0 – 299 999	1.4	..[b]	..
300 000 – 499 999	37.5	4 840	1.2
500 000 – 699 999	38.0	14 210	2.4
700 000 or more	23.1	29 590	3.5

a Figures in this column are sourced from the ERA's (2011) final report. **b** Reported as 'statistically insignificant'. **..** Not applicable.

Sources: ERA (2011); Marsden Jacob Associates (2011).

How has the study been used?

The hedonic pricing estimates were an input to the ERA's cost–benefit analysis of the SUPP, which was conducted to inform WA Government decisions about the future of the scheme (ERA 2011). The analysis was retrospective, covering the period 1996–2010.

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[1] Marsden Jacob Associates (2011) excluded data for six of the quarterly periods from the reported results on the basis that the estimates for these periods were not statistically significant.

Overall net benefits were estimated at $525 million over the period (in present-value 2010 dollars). This figure reflects:

- estimated costs of $312 million — mainly upfront costs of placing power supply underground

- estimated quantifiable benefits of $817–858 million — mainly improved amenity for households and lower maintenance costs for Western Power (the state-owned electricity distribution company).

Non-market benefits to ratepayers were estimated at $739.4 million. This was calculated by multiplying the average implicit price of underground power from the hedonic pricing study ($9962/property) by the number of properties that had been covered by SUPP projects. The ERA (2011) considered this to be an underestimate since not all benefits to ratepayers would be fully reflected in house prices.

To provide a 'check' on the hedonic pricing estimates, the ERA also drew on past surveys of ratepayers. These asked ratepayers whether they would be willing to pay a specified ratepayer contribution for a particular SUPP project to go ahead in their area (this contribution is usually 50 per cent of the cost of retrospective underground power). Contribution amounts ranged from $3900 to $5100 per household, with acceptance rates ranging from around 62 to 86 per cent across project areas (ERA 2011). While the survey data did not allow for a detailed calculation of willingness to pay, the ERA concluded that these data were broadly consistent with the hedonic pricing study. It considered that the additional costs of conducting a separate stated preference study would not be warranted.

The ERA (2011) noted that a large proportion of the benefits of the SUPP accrued to households in suburbs with higher-value properties. It found that most property owners benefited more from retrospective underground power than the amount they contributed to have it installed. In addition, ratepayers in suburbs with high property values were effectively receiving the largest subsidies, due to the way the costs of the scheme were shared between the WA Government (25 per cent), Western Power (25 per cent) and local governments (50 per cent, collected from ratepayers).

As such, the ERA (2011) recommended changes to the funding arrangements for the SUPP. It proposed that:

- Western Power contribute an amount equivalent to its avoided future maintenance costs (around 15 to 35 per cent of scheme costs, depending on the project area)

- the WA Government's contribution should vary based on property values in each area (ranging from 5 per cent for the areas with the highest house prices to 40 per cent in areas with the lowest)

- ratepayers bear the remaining costs (ranging from 25 to 80 per cent, depending on the above shares).

Policy outcomes

Following consideration of the report, the WA Government decided not to adopt the ERA's proposals for changing funding arrangements for the SUPP (Trenwith 2011). Arrangements for the retrospective installation of underground power under the SUPP were largely maintained.

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C   Validity and reliability of stated preference methods

There has been considerable debate about how well stated preference methods can measure the value that people place on non-market outcomes, especially in the case of non-use values. Key debates have been summarised in detail by Arrow et al. (1993), Carson, Flores and Meade (2001), and Kling, Phaneuf and Zhao (2012).

This appendix examines the evidence on whether stated preference methods can provide results that:

- match values that arise from markets, voting or experiments (criterion validity)

- align with revealed preference estimates (convergent validity)

- are consistent with the assumptions of economic theory (construct validity)

- can be replicated (reliability)

- can be applied to different contexts (benefit transfer).

C.1   Criterion validity

A natural starting point for examining the validity of stated preference methods (contingent valuation and choice modelling) is to compare the estimates with other measures of value that are widely accepted as being valid. These could include prices in competitive markets, values derived from economic experiments, or the outcomes of binding votes. Criterion validity would occur when the estimates align closely, and would provide evidence for the validity of stated preference methods.

Market prices

One way to establish the validity of stated preference methods is to compare the estimates to market prices for the same good. This can only be done for *private* goods (such as consumer products), since many non-market outcomes are *public* goods that lack a competitive market (or a market at all). Accordingly, some researchers have sought to test how well stated preference methods (especially contingent valuation) can value private goods, such as new products that are about

to be brought to market. The intention is usually to test how well the methods perform in a context where the goods are relatively familiar to consumers, and where value estimates can be compared to demand curves derived from market data (taken to represent the true values).

The assumption has generally been that this is a relatively easy test compared to valuing environmental 'goods' that are less familiar to survey participants, and for which market estimates of value are rarely available. Such tests using contingent valuation have often found that the stated preference estimates are somewhat higher than market-derived estimates of total value (Carson and Groves 2007). These results led some analysts to conclude that stated preference estimates are invalid, while others have explored ways to 'calibrate' these estimates (for example, by halving them) (Diamond and Hausman 1994).

However, more recent developments in the theory of non-market valuation suggest a different interpretation. Carson and Groves (2007) argue that the results are due to the nature of private goods. Because survey participants are not compelled to purchase the good, they might act strategically by overstating their willingness to pay if they believe that this would encourage a new good to be made available on the market. Actual purchase decisions would be made later.

However, public goods are provided in a different context. The government can provide public goods (such as improvements in biodiversity) to all and compel everyone to pay (for example, through taxes). If survey participants believe that they may be compelled to pay based on their responses, they may no longer have an incentive to answer strategically. It is therefore possible that stated preference methods can provide valid estimates for public goods, but not necessarily for private goods, when people are asked about their willingness to pay. Thus, comparisons with market values offer little evidence for how well the methods can value public goods.

Voting outcomes

Absent evidence from competitive markets, researchers have looked at whether stated preference methods can accurately predict the outcome of a binding referendum (which asks a similar question to a 'yes/no' contingent valuation survey). A referendum — for example, on whether or not an environmental program funded by increased taxation should be introduced — is generally considered to be 'incentive compatible'. That is, people that would prefer to pay the extra tax and have the program proceed have an incentive to vote yes (and vice

versa for no votes). Therefore, such referendums provide an opportunity to test the validity of stated preference surveys that ask essentially the same question.

Several studies have compared the results of stated preference studies to the outcomes of later referendums on the provision of local public goods, such as parks and connections to piped water systems (Johnston 2006; Vossler and Kerkvliet 2003; Vossler and Watson 2013; Vossler et al. 2003) (box C.1). Most studies have found that estimates of willingness to pay from the stated preference survey match those implied by the referendum outcome. In some cases, this depends on survey participants feeling their answers will have consequences for policy decisions (Johnston 2006; Vossler and Watson 2013), or on 'undecided' responses in the survey being coded as 'no' (Vossler et al. 2003). Where alignment between survey and voting outcomes has not been found, this may have been due to differences in the amount and specificity of information provided in the stated preference survey compared to the referendum (Schläpfer, Roschewitz and Hanley 2004).

Box C.1   Criterion validity: Evidence from a referendum

Vossler and Watson (2013) compared the results of a stated preference survey with the outcome of a binding referendum in Middleborough, Massachusetts (United States). The referendum concerned a proposal to levy a 1 per cent property tax surcharge to fund a scheme to create, acquire and/or preserve open space, historic resources, recreational land and community housing.

Prior to the referendum in November 2010, a stated preference survey was distributed to a sample of households using identical language to the upcoming referendum question. This was done at a time when public knowledge of the referendum was generally low. To reduce the incentive for participants to answer strategically, the survey did not mention the referendum.

The results of the survey closely matched those of the actual referendum. While there were statistically significant differences in three of the six voting precincts, these differences disappeared when the sample was restricted to participants that believed the survey to be consequential. This was assessed using responses to a question on whether participants believed that their survey responses would be taken into consideration by policymakers.

Moreover, the variation in property values (meaning that the dollar amount of tax paid would vary across households) allowed the researchers to estimate average willingness to pay. This measure was significantly lower and more variable for survey participants that did not believe the survey to be consequential. The implication is that stated preference methods are more likely to be valid when participants consider the survey to be consequential.

Source: Vossler and Watson (2013).

However, the evidence from referendums is not definitive on its own. Voting was not compulsory in the above studies (which were mostly conducted in the United States), meaning that if voters hold different values for the environmental good to non-voters, the referendum results may not be representative of the broader population. Moreover, these studies all relate to use values within a local area and say little about non-use values that may be held by people over a wider area.

Nevertheless, the evidence from voting outcomes generally supports the validity of stated preference methods. This gives confidence that stated preference surveys can elicit preferences for non-market environmental outcomes in a similar way to referendums. An additional benefit of stated preference surveys is that — unlike most referendums — the monetary contribution can be varied across people, allowing welfare measures (such as average willingness to pay) to be calculated.

Experiments

Experiments offer another way to observe the value that people place on public goods. This typically involves constructing a market in a laboratory or intervening in real markets in the 'field'. There are three main types of experiment (Kling, Phaneuf and Zhao 2012).

- A referendum-style vote is used to determine whether or not all participants receive a good for which they will all be made to pay a set amount. The results of this real payment mechanism are then compared to results from a stated preference survey of the participants.

- Participants are assigned 'values' for a public good and requested to complete a stated preference survey. The researcher then tests whether the survey accurately elicits the pre-assigned values.

- Stated preference surveys of hunting-licence holders are compared to the outcome of interventions in markets for licences (or other comparable markets).

Several studies have found significant differences between willingness-to-pay estimates from stated preference surveys to those derived from experiments. For example, two meta-analyses have found that stated preference estimates tend to be higher across a large number of studies. One found that average willingness to pay is around three times higher in surveys than in experiments (List and Gallet 2001). Another found a median ratio of survey to experimental estimates of 1.35 (Murphy et al. 2005).

Stated preferences estimates that are significantly higher than values from experiments have also been reported in several other studies (Harrison and

Rutström 2008). A similar conclusion has been drawn by some field experiments that compared actual surrenders of hunting permits to surveys of licence holders' willingness to accept compensation (Bishop and Heberlein 1979). By contrast, other such field experiments have found that stated preference surveys give lower or similar estimates to market data (Loomis, Pierce and Manfredo 2000; Ready, Epp and Delavan 2005).

Other researchers have investigated the causes of these differences, and found that experimental estimates can match those from stated preference surveys when particular conditions are met (Taylor 2006). For example, some studies have found that survey and experimental results converge only when participants perceive their survey responses to be consequential (by influencing outcomes that they care about) (Landry and List 2007; Vossler, Doyon and Rondeau 2012; Vossler and Evans 2009). Others have found convergence where surveys are structured to give people a strong incentive not to strategically misrepresent their responses (Taylor et al. 2001; Vossler and McKee 2006), or where participants are explicitly asked to answer honestly (Cummings and Taylor 1999).

These findings suggest that there may be good reasons for the inflated stated preference estimates found in some studies. For example, studies that value private goods or involve voluntary contributions may be subject to strategic misrepresentation (Carson and Groves 2007). Moreover, where purely hypothetical questions are asked in a survey, participants may not have an incentive to provide careful and honest responses because they are unlikely to consider the outcome to be consequential.

Overall, the experimental evidence supports the validity of stated preference methods, provided that surveys are well designed and considered by participants to be consequential. However, the evidence is not conclusive on its own. For example, the behaviour of volunteers used in experiments (who are often students) may not be representative of the broader community. Further, experiments are often based on providing participants with a tangible good (which is usually not an environmental good), and so may not be well suited to eliciting non-use values.

C.2 Convergent validity

The validity of stated preference methods can also be examined by making comparisons with other non-market valuation methods, such as revealed preference. Convergent validity would occur if the estimates align (when both are expressed in the same units, such as willingness to pay). While such evidence would be consistent with the validity of stated preference methods, it would not be sufficient

to establish validity on its own. Other methods may have a different theoretical basis, or can be subject to measurement errors.

Convergent validity has typically been assessed for recreation, amenity and environmental health risks, where there are sufficient data to allow both revealed and stated preference techniques to be applied. This literature covers environmental use values, the value of a statistical life and 'contingent behaviour' (where travel-cost or hedonic-pricing methods are used to analyse survey responses to hypothetical scenarios). A common finding across these contexts is that stated preference estimates tend to be correlated with revealed preference estimates and broadly similar in magnitude.

However, there is wide variation across studies (table C.1). In a widely cited meta-analysis, Carson et al. (1996) found that stated preference estimates were, on average, around 75 to 94 per cent of the corresponding revealed preference estimates, with a high degree of correlation between them. Some researchers have similarly found that stated preference estimates are lower than their revealed preference counterparts (Brander, Van Beukering and Cesar 2007; Loomis 2006; Rolfe and Dyack 2010), but others have found the stated preference estimates to be higher (Azevedo, Herriges and Kling 2003; Brander, Florax and Vermaat 2006; Woodward and Wui 2001). Further, the two sets of measures have matched closely in some studies, even though statistically significant differences may remain (De Blaeij et al. 2003; Grijalva et al. 2002).

Closer convergence between stated and revealed preference estimates has been found in some cases under particular conditions. For example, Rolfe and Dyack (2010) reported contingent valuation estimates around 22 per cent lower than travel-cost estimates, but the difference was no longer statistically significant after excluding uncertain responses to their contingent valuation question. Loomis (2006) reported contingent valuation estimates around 44 per cent lower than travel-cost estimates, but found convergence after controlling for multi-destination trips in the travel-cost analysis.

Table C.1 Selected convergent-validity studies

Study	Type	Findings
Brander, Florax and Vermaat (2006)	Meta-analysis	Contingent valuation estimates of the value of ecological and recreational services provided by wetlands (across multiple countries) were significantly higher than those from other methods (including travel cost and hedonic pricing estimates), all else equal.
Brander, Van Beukering and Cesar (2007)	Meta-analysis	Contingent valuation estimates of the value of ecological and recreational services provided by coral reefs (across multiple countries) were significantly lower than those from travel-cost and other methods, all else equal.
Carson et al. (1996)	Meta-analysis	There was a high degree of correlation between stated and revealed preference estimates for recreation, amenity and health risks (mostly in the United States and Europe). The median ratio of stated to revealed preference estimates ranged from 0.75 to 0.94, depending how the dataset was treated.
De Blaeij et al. (2003)	Meta-analysis	Stated preference estimates of the value of a statistical life associated with road safety were very close to hedonic wage estimates (after controlling for relevant factors), but slightly higher and with a statistically significant difference.
Woodward and Wui (2001)	Meta-analysis	Contingent valuation estimates of the value of ecological services provided by wetlands (across multiple countries) were significantly higher than travel-cost estimates once the type of ecological service was controlled for.
Loomis (2006)	Travel cost	Contingent valuation estimates of the recreation value of Snake River in Wyoming (United States) were around 44 per cent lower than corresponding travel-cost estimates. The difference was not significant after controlling for multi-destination trips in the travel-cost analysis.
Rolfe and Dyack (2010)	Travel cost	Travel-cost estimates of the recreation value of the Coorong Wetland in South Australia ($149 per adult per day) were significantly higher than corresponding contingent valuation estimates ($116). The difference was not significantly different after uncertain responses to the contingent valuation question were excluded (rather than coded as 'no').
Azevedo, Herriges and Kling (2003)	Contingent behaviour	Travel-cost consumer surplus estimates of future visits to wetlands in Iowa (United States) (contingent behaviour) were around 2.5 times larger than travel-cost estimates based on actual visits.
Grijalva et al. (2002)	Contingent behaviour	Contingent behaviour data could successfully predict changes in travel patterns (to outdoor rock climbing sites in Texas) after site access was restricted.
Jeon and Herriges (2010)	Contingent behaviour	Contingent behaviour travel-cost estimates of the recreation value of lakes in Iowa (if water quality were to be improved) were significantly lower than estimates based on actual visits.
Lanoie, Pedro and Latour (1995)	Hedonic wage	Estimates of the value of a statistical life associated with workplace safety were significantly higher using contingent valuation than hedonic wage analysis. However, the estimates were more similar when highly risk-averse individuals were excluded from the analysis.

More generally, there may be explanations for some of the observed differences. Stated and revealed preference methods usually provide different measures of welfare — for example, consumer surplus in the case of travel-cost models (willingness to pay minus the price paid), implicit prices in hedonic pricing and choice modelling (willingness to pay for one additional unit of an attribute), and total willingness to pay in contingent valuation. Measures from different methods require conversion to a common format to allow comparisons to be made. Moreover, travel-cost methods generally estimate the average surplus associated with visiting a site, whereas stated preference methods estimate the value of an additional or marginal unit of an environmental good. Estimates may also differ when stated preference estimates include some non-use values, which are generally not picked up in revealed preference estimates. Since each technique provides measures that are not strictly comparable, differences are to be expected in the resulting estimates.

Alternatively, measurement errors, biases and assumptions could account for some of the differences. There is evidence that revealed preference estimates are sensitive to assumptions made in the analysis and the quality of available data (chapter 2), and that stated preference estimates can be biased when surveys are poorly designed or implemented (section C.3). Indeed, these complications underpin a strand of the literature that combines data from both stated and revealed preference sources, on the basis that neither is perfect but each can provide a different perspective on behaviour (Adamowicz, Louviere and Williams 1994). The evidence from marketing and transport applications of this approach suggests that differences between the two types of data can in large part be explained by the way that statistical models are constructed (Louviere, Hensher and Swait 2000).

Taken as a whole, the evidence demonstrates that stated preference estimates are often reasonably close to their revealed preference counterparts (for use values and where a good can be valued using both approaches). However, this is contingent on each study being well conducted and measuring the same kind of values. The fact that estimates from both methods tend to be broadly similar and correlated suggests that stated preference estimates are consistent with other measures of value. At the same time, stated preference estimates are not necessarily invalid if they do not align closely with revealed preference estimates when the latter are subject to error.

On the other hand, the convergent validity literature has focused almost exclusively on *use* values, for which revealed preference estimates can be derived. It says little about the validity of stated preference methods for estimating *non-use* values, for which corresponding revealed preference estimates are generally not available.

C.3 Construct validity

Another source of evidence relates to whether stated preference methods provide results that are consistent with the assumptions that underpin the them. This is known as construct validity. Economic theory makes several testable predictions that stated preference methods should pass if they are a valid way to value non-market outcomes. This has been a key area of debate in the literature (Carson, Flores and Meade 2001; Carson 2012; Diamond and Hausman 1994; Hanemann 1994; Hausman 2012; Kling, Phaneuf and Zhao 2012).

What predictions does theory make?

Stated preference methods are derived from welfare economics. Welfare economics is based on the assumption that people have well-formed and stable preferences over all outcomes (market or non-market) that are relevant for their wellbeing. It is also assumed that people make rational decisions based on their preferences and seek to maximise their wellbeing (or utility) at all points in time. Stated preference methods seek to discover the preferences people have based on how they respond to survey questions.

Testable predictions arising from economic theory include that:

- people are willing to pay more for a greater quantity of a non-market good (such as for a larger environmental project)
- the underlying preferences people have over non-market outcomes do not depend on the survey instrument used to elicit them
- there is a close alignment of measures of willingness to pay and willingness to accept compensation
- the income elasticity of demand for an environmental good should be larger than one if environmental quality is a 'luxury' good.

These predictions have been the most debated in the literature and are discussed in the sections that follow. Others have been less controversial, including that willingness to pay should be correlated with income, and the proportion of people that are willing to pay for a non-market outcome should fall as the amount they are asked to pay rises. It is widely accepted that stated preference data generally meet these tests (Carson, Flores and Meade 2001; Kling, Phaneuf and Zhao 2012).

Invariance to scale

Economic theory suggests that if people value a good then they would be willing to pay more for a higher quantity of that good, such as a greater area of land to be conserved or a larger number of birds to be protected. Whether this is the case in stated preference data has been one of the most contentious areas of debate (Diamond and Hausman 1994; Hanemann 1994; Kling, Phaneuf and Zhao 2012).

Some prominent contingent valuation studies found that willingness to pay did not vary much when people were presented with very different levels of a good (tested by surveying two comparable sub-samples of people). For example, Desvousges et al. (1992) found little difference in willingness to pay for preventing the death of 2000, 20 000 or 200 000 waterbirds. Kahneman (1986) found that willingness to pay to improve the environmental health of lakes in one part of Ontario, Canada was only slightly less than willingness to pay to improve all lakes in that province. Kahneman and Knetsch (1992) found that willingness to pay did not increase substantially when people were asked to value a broader set of goods, including environmental conservation, local sporting facilities and scientific research. These results led some researchers to claim that stated preference surveys do not measure willingness to pay for a specific non-market outcome but, rather, a 'warm glow' that reflects the moral satisfaction of supporting environmental causes generally (Diamond and Hausman 1994; Kahneman and Knetsch 1992).

However, many studies (covering both use and non-use values) have found that estimates of willingness to pay are sensitive to the scale of the good described in the survey (Bennett 2011; Carson 1997; Ojea and Loureiro 2011). These include meta-analyses that examined estimates of willingness to pay across multiple studies to find that such estimates are larger for higher levels of a good (such as visibility improvements at national parks, increases in biodiversity or improvements in water quality) (Van Houtven, Powers and Pattanayak 2007; Johnston, Besedin and Wardwell 2003; Ojea and Loureiro 2011; Smith and Osborne 1996).

Possible explanations

Researchers have investigated why estimates sometimes appear to be invariant to scale. One possibility is that inappropriate statistical methods were used to analyse the data (Carson, Flores and Meade 2001; Carson 1997). For example, when sample sizes are small, differences in willingness to pay for different levels of a good might not show up as statistically significant.

Another explanation is that survey participants may have found it difficult to understand or contextualise the information provided in a survey. For example:

- Carson (1997) criticised the survey used by Desvouges et al. (1992) for not making the magnitude of bird numbers (cited above) clear to participants. These were described in terms such as 'much less than 1 per cent' and 'about 2 per cent'.

- Corso, Hammitt and Graham (2001) found that survey participants can appear insensitive to scale when they are asked about changes in low-level risks (such as mortality risk), but this invariance does not arise when risk levels are clearly explained using visual aids.

- Ojea and Loureiro (2011) found sensitivity to scale when the area of land to be conserved is expressed in absolute rather than relative terms.

- Bennett, Morrison and Blamey (1998) found insensitivity to scale when payment amounts were very low, such that participants may have considered the amounts trivial or questioned the credibility of the policy.

Other explanations arise from economic theory. Bateman (2011) argues that economic theory does not predict how *much* willingness to pay should increase for larger amounts of a good. Additional amounts could have a diminishing incremental impact on wellbeing once a certain level has been reached. Amiran and Hagen (2010) argue that limits to how well market goods can be substituted for environmental outcomes can explain low levels of sensitivity to scale. Others have pointed to an 'embedding' effect predicted by theory, where the value of a good is lower when it is valued as part of a broader package of goods, some of which are substitutes (discussed further below).

This research implies that stated preference estimates are sensitive to scale, provided that the good is clearly described and the study is well conducted. There is broad agreement among practitioners of non-market valuation that the degree of sensitivity to scale does not invalidate stated preference methods in general.

Sensitivity to the survey instrument

Researchers have found that stated preference estimates can be sensitive to the way a survey is designed and the information it contains. Small changes in the design or layout of a survey can have a large influence on the resulting estimates of willingness to pay. Researchers have tested this by providing two comparable sub-samples of people with slightly different versions of a survey (for example, with a different description of the environmental good), and examining whether this difference has a significant impact on responses.

Several patterns have been identified in how people respond to surveys.

- Estimates of willingness to pay tend to vary depending on the type of valuation question asked. There is evidence that asking a single dichotomous-choice valuation question ('yes/no') leads to significantly higher estimates of willingness to pay than those obtained through other types of question, such as repeated dichotomous choice, open-ended or payment-card methods (Carson and Groves 2007; Champ and Bishop 2006).

- Estimates can be sensitive to the specificity and detail of information provided about the environmental outcome and broader environmental context (Blomquist and Whitehead 1998; MacMillan, Hanley and Lienhoop 2006; Munro and Hanley 1999). For example, more information about possible uses of the environmental good can increase estimates of willingness to pay (Bergstrom, Stoll and Randall 1990), whereas information about substitutes can reduce estimates (Boxall et al. 1996; van Bueren and Bennett 2004).

- The type of payment mechanism used can influence willingness to pay estimates — for example, the use of compulsory levies may be associated with lower estimates than an increase in a range of taxes and/or consumer prices (Rolfe and Brouwer 2011). There can be differences between estimates when annual or lump-sum payments are used, with the implied discount rate varying greatly over studies — from over 20 per cent (Bond, Cullen and Larson 2009; Kovacs and Larson 2008) to less than 5 per cent (Rolfe and Brouwer 2011).

- Estimates of willingness to pay can be higher and/or less variable when a change in an environmental outcome (such as the number of rare species) is framed as a 'loss' rather than a 'presence' (Kragt and Bennett 2012).

- People sometimes appear to 'anchor' responses to numbers seen earlier in a survey (Green et al. 1998), and may answer 'yes' to questions when they are uncertain or wish to please an interviewer (Carson, Flores and Meade 2001; Loomis, Traynor and Brown 1999).

- Responses in choice modelling surveys sometimes appear to be influenced by questions asked earlier in the survey, and can be sensitive to the order in which choice sets are presented (Day et al. 2012; Scheufele and Bennett 2013).

- Willingness to pay for a good has been observed to fall the later it is valued in a sequence of goods (Carson and Mitchell 1995; Clark and Friesen 2008; Kahneman and Knetsch 1992).

These findings have sometimes been interpreted as evidence that people do not have well-formed or stable preferences for the underlying non-market outcomes, and that stated preference methods do not provide valid estimates of the value people place on these outcomes (Diamond and Hausman 1994). An alternative interpretation is

that people respond to a survey in a rational (and predictable) way given the circumstances. If the observed patterns can be explained and adjusted for, there would be greater confidence that stated preference surveys can provide unbiased estimates of the value that people place on the underlying non-market outcomes.

Incentives to misrepresent preferences

Strategic bias can explain why some kinds of valuation question give different results. When a survey is consequential (participants believe their responses will affect policy decisions that they care about), participants may seek to answer in a way that influences policy decisions in their favour (Carson and Groves 2007). Accordingly, some types of survey could increase the incentive people have to misrepresent their true preferences. For example:

- people might 'free ride' by overstating their willingness to pay if they believe they will not have to pay once the good is provided (either because they are surveyed about a voluntary contribution, or because they do not find the payment scenario to be credible)

- in open ended or payment card questions, people might misrepresent their willingness to pay to increase the influence their responses have on whether the good will be provided — by reporting a very high amount if they value the good more than they expect it will cost to provide, and a very low or zero amount otherwise

- in choice modelling surveys, people might select an option in a choice set that is not their most preferred because they believe that the outcomes could be provided at lower cost (based on the options in earlier choice sets)

- people might also select between the two options in a choice set that they consider the most likely to be adopted by government, even if neither is their most preferred (Carson and Groves 2007).

Carson and Groves (2007) argue that strategic bias can occur in a number of survey formats. They propose that it is only avoidable when a single 'yes/no' dichotomous-choice valuation question is asked, provided other conditions are also met (for example, participants are told that the policy will go ahead if a majority of participants select 'yes'). However, this style of question can be statistically inefficient, suggesting a trade-off between minimising strategic bias and obtaining more information on individuals' preferences (Carson and Groves 2011).

Strategic bias can also arise where participants do not consider the payment mechanism to be specific to their circumstances — for example, as may occur in jurisdictions where specific levies are rarely used and tax rates differ across

taxpayers. In these cases, honest responses can be encouraged by ensuring that the payment mechanism is perceived as credible and applicable to each individual survey participant.

Inadequate information

The way that people respond to surveys will also depend on whether they have a good grasp on what they are being asked to value. The evidence suggests that people will answer survey questions even if they do not understand the questions or material provided.

In the absence of clear and unambiguous information, people might make their own assumptions to fill in the gaps (Hanemann 1994; Johnston et al. 2012). This may be especially likely where the policy outcomes being described are not expressed in terms that are directly valued by participants, but are instead proxies for the ultimate environmental outcomes that they care about — in which case they may draw on prior knowledge or make erroneous assumptions to make the relevant connections (Collins 2011; Johnston et al. 2012). Estimates of willingness to pay can also be biased when some important elements of the policy outcome are not mentioned in the survey (such as social impacts or how the policy will be implemented) and participants respond based on their own understanding of what these elements would likely be (Johnston and Duke 2007).

Alternatively, they might submit 'protest' answers if they question the credibility of the information presented or disapprove of the survey (for example, by indicating a very low or very high willingness to pay, or refusing to answer the valuation question) (Kling, Phaneuf and Zhao 2012). This might be the case, for example, if they believe that the option would be ineffective or infeasible, or that government intervention is not warranted. For example, there is evidence that responses are sensitive to whether participants consider the type of payment mechanism or distribution of costs to be 'fair' (Cai, Cameron and Gerdes 2010; Jorgensen and Syme 2000). Another possibility is that participants who are not convinced that the policy would be implemented in the way described modify their responses (for example, based on their perception of how likely it is that the outcome would be achieved).

When faced with an unfamiliar context or decision, people may fall back on behavioural rules-of-thumb. This could account for observations that people sometimes anchor survey responses to numbers seen previously in the survey or answer 'yes' when they are uncertain (Bateman 2011; Kling, Phaneuf and Zhao 2012). Such behaviour is consistent with evidence from experiments that have found deviations from models of rationality and evidence of 'rules of thumb' being

deployed when people are faced with unfamiliar choices or too many choices ('choice overload') (Iyengar and Kamenica 2010; Plott 1996). This behaviour may also be consistent with the 'focusing bias', where people have an exaggerated view of the relative importance of an issue simply because the survey makes them focus on it (Kahneman and Sugden 2005).

Behavioural economics also offers explanations for other findings in the literature. For example, evidence that responses differ depending on whether a change in environmental conditions is a 'loss' or 'presence' is consistent with an endowment effect that has been observed in some markets (see below). Very high discount rates, that have been observed when comparing estimates of willingness to pay based on one-off payments to regular payments, may be broadly consistent with evidence from markets and experiments (Frederick, Loewenstein and O'Donoghue 2002).

These behavioural responses suggest that stated preference methods may be more likely to generate biased estimates when survey participants have low familiarity with the non-market good being valued (which may be more likely for non-use values) , or when the good is not described in a way that they find credible or can easily relate to. In such cases, they may not have a prior sense of their willingness to pay and could construct their valuation of the good during the course of the survey, with little time for reflection (Bateman 2011).

However, there is evidence that these kinds of biases can be minimised through survey design. For example:

- more specific and detailed information about the non-market outcome and its context can reduce the need for participants to make their own assumptions (Blomquist and Whitehead 1998; Munro and Hanley 1999)

- the use of visual material (such as maps, photographs and diagrams) can help participants to contextualise the environmental good or risk levels (Bateman 2011; Corso, Hammitt and Graham 2001)

- providing an appropriate amount of information and questions, but not too much, can increase the likelihood that participants stay focused on the survey (Louviere et al. 2008)

- asking detailed questions on discretionary expenditure can make participants more aware of which household expenditures they may need to forgo for the good to be provided (their 'budget constraint') (Li et al. 2005)

- clearly marked practice questions can help to familiarise participants with the exercise before the formal valuation question is asked (Bateman 2011)

- follow-up questions can identify when participants are highly uncertain or did not understand the information provided, allowing their responses to be treated differently in the analysis (for example, coding these responses as 'no' or dropping the participants from the sample) (Loomis, Traynor and Brown 1999).

Substitutes

The context in which environmental outcomes are provided matters for the value that people place on them. This may explain why willingness to pay appears to fall depending on where a good is placed in a sequence, or whether people have considered the available substitutes. Economic theory predicts an 'embedding' or 'part–whole' effect, where the value of a good is higher when it is provided on its own rather than as a package of goods (Carson, Flores and Hanemann 1998; Hoehn and Randall 1989). This is the case where the goods are substitutes (so that the benefit of providing an additional good would be lower than when that good is provided on its own) and where providing a larger number of goods reduces the income people have available for other uses (such as market goods). These effects have also been found in market data (Randall and Hoehn 1996).

One implication is that the value that people place on a package of goods is expected to be less than the sum of the values they would place on each good if it was being provided independently. Another is that the value of a non-market outcome can be sensitive to the extent of available substitutes. For example, the value that a person places on conserving one particular habitat may fall if they become aware of other habitats that perform a similar ecological function. Likewise, the value people place on a project within one catchment management area may depend on whether there are other potential projects elsewhere that could provide comparable environmental benefits.

Put simply, people may value an environmental outcome more highly the scarcer it is likely to be. Arrow et al. (1993, p. 4605) noted that:

> … even if the willingness to pay responses to individual environmental insults are correct if only one program is to be considered, they may give overestimates when there are expected to be a large number of environmental problems. Similarly, if individuals fail to consider seriously the public or private goods that might be substitutes for the resources in question, their responses to questions in a [contingent valuation] survey may be unrealistically large.

While the evidence indicates that responses to stated preference studies are sensitive to how substitutes are described, it is less clear about whether stated preference surveys can be designed in a way that allows participants to properly consider all relevant substitutes (which they may not be fully aware of prior to the survey).

Nevertheless, the evidence does suggest that valuing one non-market outcome in isolation should not be relied upon when a specific policy change is associated with a set of non-market outcomes.

Assessing the impact of survey design

The evidence suggests that stated preference estimates can be highly sensitive to the way that surveys are structured and the presentation of information on the environmental good and its context. This evidence is largely consistent with survey participants responding in a rational or predictable way given the specific circumstances of the survey, and supports the validity of stated preference methods provided that surveys are appropriately designed. Where anomalies do arise, many can be explained by behavioural traits that have been observed in experiments and markets. Moreover, estimates should be expected to vary with differences in survey instruments, to the extent that these differences alter the environmental 'good' that participants are being asked to value.

A remaining question is how to assess whether a survey is likely to provide unbiased estimates of the value that people place on the underlying non-market outcomes of interest. This can be done in three broad ways.

- Focus groups and pre-testing can be used to assess whether the policy outcomes are credible and described in terms that are relevant to participants, to understand how participants are likely to interpret the information provided, and to identify the factors that they might consider when responding to the survey. Consultation with stakeholders can help to ensure that the information is objective and less likely to be disputed.

- Follow-up questions can be used to gauge whether participants understood the information provided and questions asked.

- Survey designs can be assessed subjectively, in terms of whether the survey contains all information one would expect to be relevant. This includes a clear specification of the environmental good and the broader context of its provision.

Criteria for assessing the quality of a stated preference study are set out in chapter 2.

Divergence between willingness to pay and accept

Non-market valuation studies have estimated the value that individuals place on non-market goods in terms of both their willingness to pay and their willingness to accept compensation. Willingness to pay is often seen as the more appropriate measure to assess the provision of a public good. By contrast, willingness to accept

is considered more relevant where property rights are affected and individuals 'lose' something, such as amenity or the right to use a parcel of land. Economic theory suggests that the two measures will be close together for a price change in competitive markets (with the difference reflecting the effect that each has on an individual's income).

However, many studies have found that stated preference methods provide estimates of willingness to accept that are substantially higher than estimates of willingness to pay (Horowitz and McConnell 2002). This gap is often much larger than found in market contexts. Critics have suggested that this indicates that neither type of stated preference estimate is valid (Hausman 2012).

There are several alternative explanations for the divergence. Behavioural economics predicts an 'endowment effect' (where people place a greater value on a good because they have a property right over it) (Kling, Phaneuf and Zhao 2012). This interpretation is supported by evidence from experiments (Knetsch and Sinden 1984) and from real markets where participants have not had much experience of interacting in the market (List 2011).

Alternatively, there may be problems with stated preference estimates of willingness to accept if participants submit protest answers because they do not consider it legitimate to use surveys to decide which publicly owned goods to give up (Carson, Flores and Meade 2001). A related concern is that the lack of a budget constraint when people are asked about accepting compensation (as opposed to paying a particular sum) could reduce the incentive to answer honestly (Arrow et al. 1993).

A further explanation comes from economic theory. Hanemann (1991) has shown theoretically that willingness to accept may be significantly greater than willingness to pay because everyone experiences the same level of a public good, which may not be a perfect substitute for market goods.

These explanations imply that stated preference methods can accurately reflect real behaviour and provide theoretically consistent results, at least when participants are familiar with the good they are being asked to value. However, such explanations can be difficult to test in stated preference data. As such, the evidence remains inconclusive on whether the divergence between willingness to pay and willingness to accept is consistent with the economic assumptions that underpin stated preference methods.

Income elasticity of demand

Economic theory makes predictions about the income elasticity of demand for different types of goods — that is, the amount that demand changes for a given change in people's income. Some economists have argued that environmental quality is a 'luxury' good since demand for environmental quality is likely to increase more than proportionally with income (McFadden 1994). This suggests that the income elasticity of demand should exceed one (in absolute terms). While this cannot be directly observed using stated preference data, the income elasticity of willingness to pay has been regularly observed to be less than one (Horowitz and McConnell 2003; Kriström and Riera 1996). This apparent anomaly may bring into question the validity of stated preference methods, or raise questions about whether environmental quality is a 'luxury' good.

Researchers have used economic theory to explain these findings. Theory suggests that the income elasticity of *demand* should exceed one, which is a measure based on changes in the quantity of a good available. But since the quantity of public goods is fixed (and these goods are available to all), stated preference data can only be used to measure the income elasticity of *willingness to pay* (Carson, Flores and Meade 2001). This is related to the elasticity of demand, but also depends on several other unobservable factors (Flores and Carson 1997). The implication is that economic theory does not make any clear predictions about what the income elasticity of willingness to pay should be, and that estimates less than one are not necessarily inconsistent with theory.

Do stated preference methods exhibit construct validity?

Taken as a whole, the available evidence suggests that anomalies found in stated preference data can mostly be explained by economic theories and/or findings from behavioural economics. There is no compelling evidence that stated preference methods *in general* are inconsistent with economic theory. However, the evidence does point to several conditions that are necessary for a study to provide results that align with economic theory, even though none of these alone can guarantee validity. In particular, careful design and implementation are essential for reducing bias in survey estimates.

C.4 Reliability

Reliability offers another perspective on the quality of stated preference methods. This refers to whether the results can be replicated.

A handful of studies have found that stated preference results can be successfully replicated when a different sample of people is surveyed, with no significant differences across samples and/or high correlation over time (McConnell, Strand and Valdés 1998). For example, Carson et al. (1997) found that responses to a contingent valuation survey did not significantly differ across two samples that were surveyed two years apart. Whitehead and Hoban (1999) found willingness to pay estimates that were lower when a new sample was surveyed five years after the original survey, but that the gap could be explained by differences in attitudes towards the environment and government. Brouwer and Bateman (2005) found a significant difference in estimates over a five year period, but noted that this may be due to differences in the two survey instruments and unobserved factors that influenced individuals' willingness to pay.

Other researchers have examined whether the same individuals or households express similar values at different points in time. For example, McConnell, Strand and Valdés (1998) surveyed the same recreational fishers two months apart, finding that estimates of willingness to accept did not differ significantly. Likewise, Loomis (1989) surveyed the same households twice, at a nine-month interval, and found no statistically significant difference in willingness to pay across a number of statistical tests.

This evidence, while limited, does suggest that stated preference methods can reliably estimate non-market values. While values could conceivably differ across populations and over longer periods of time (due to changes in people's preferences), differences across samples or shorter time periods can generally be explained by measurable factors.

C.5 Benefit transfer

There is a large body of evidence on the validity of benefit transfer, where non-market estimates are transferred from one policy context to another. This is tested by conducting a new study (at the 'policy' site) and comparing the resulting estimates to those generated from benefit transfer (derived from the 'study site(s)'), usually in terms of comparing average willingness to pay or the coefficients in statistical models. 'Transfer error' arises when the transferred estimates do not fit the policy context, and thus the two sets of estimates diverge. This has been

assessed mainly for the transfer of stated preference estimates, although research has also covered revealed preference estimates.

Statistically significant transfer errors have been found in a large number of stated preference studies. This is reflected in several literature reviews (Bergstrom and De Civita 1999; Brouwer 2000; Johnston and Rosenberger 2010; Kaul et al. 2013; Rosenberger and Stanley 2006). The magnitude of the error can vary considerably, ranging from close to zero (no error) in some cases, to well over 100 per cent or more in others (Rosenberger and Stanley 2006). Similar degrees of error have also been found when transferring revealed preference estimates (Chattopadhyay 2003; Rosenberger and Loomis 2000). In a meta-analysis of benefit-transfer studies, Kaul et al. (2013) found that errors are lower when values for quantitative environmental indicators are transferred, compared to values for environmental quality.

A general finding is that the errors tend to be lower when there is greater similarity between the study and policy sites (Johnston 2007; Kirchhoff, Colby and LaFrance 1997; Piper and Martin 2001). However, there is little agreement about which factors matter most. For example, Johnston (2007) found that the value placed on environmental protection (in terms of implicit prices) was closer across regions with more similar land-use types (in terms of the density of housing and open space). Another study on land management found lower transfer errors when land-cover types (along with geographical proximity and average incomes) were more similar across regions (Colombo and Hanley 2008).

Others have found that differences across populations have a more pronounced effect. For example, Van Bueren and Bennett (2004) found significant differences in how populations in different parts of Australia value environmental attributes within their own region. Morrison and Bennett (2004) found that non-use values for river attributes (such as fish and vegetation) varied considerably more across the populations of catchments than recreation use values. Morrison et al. (2002) found lower errors when transferring estimates of non-use values for wetland improvement across wetlands (for the same population) than across populations (such as urban or rural). Others have attributed such findings to differences across populations in attitudes towards environmental protection (and contributing financially to it) (Brouwer and Spaninks 1999; Jiang, Swallow and McGonagle 2005). Further possibilities are that errors might arise due to differences in the scale of environmental change due to a policy, the way the policy is implemented, or the available substitutes for the environmental good (Johnston and Rosenberger 2010).

Researchers have also investigated whether transfer errors tend to be lower for particular types of estimates. There are mixed findings on whether implicit price

estimates from choice modelling can be accurately transferred, with some studies finding evidence in favour (Morrison et al. 2002) and others against (Hanley, Wright and Alvarez-Farizo 2006). However, at least one study found that choice modelling can perform reasonably well for ranking environmental projects across regions, even when there are large errors associated with the transfer of willingness to pay values (Jiang, Swallow and McGonagle 2005).

Other researchers have focused on revealed preference methods and come to broadly similar conclusions to the stated preference literature. For example, Chattopadhyay (2003) found large errors when transferring hedonic pricing estimates for air quality, which may be because implicit prices are specific to local housing markets and the marginal value of air quality could vary significantly across these markets. Rosenberger and Loomis (2000) found large errors when transferring travel-cost estimates of outdoor recreation values to populations in different regions. These errors are within the ranges that have been found for the transfer of stated preference estimates.

Further research has focused on whether some benefit-transfer approaches produce lower errors than others. A common finding is that 'function transfer' (where willingness to pay is modelled as a function of variables, drawn either from one or multiple studies) performs better than a simple 'unit' transfer of values (Brouwer and Spaninks 1999; Kaul et al. 2013; Kirchhoff, Colby and LaFrance 1997; Rosenberger and Stanley 2006). However, some studies have found unit transfer to be more accurate (Bergland, Magnussen and Navrud 2002; Colombo and Hanley 2008; Johnston and Rosenberger 2010). This divergence in findings could partly reflect the specific study and policy contexts in each case, and any assumptions made.

There is also mixed evidence on meta-analysis, where multiple primary studies are analysed to develop a value function while controlling for methodological differences across studies. Using this method, researchers have found average transfer errors of around 74 per cent for wetland values (Brander, Florax and Vermaat 2006) and 186 per cent for the recreational value of coral reefs (Brander, Van Beukering and Cesar 2007). More generally, however, the meta-analysis of environmental valuation studies has been limited, in part because environmental goods are often highly specific (Navrud and Ready 2007).

The evidence suggests that transferring a non-market estimate from one context to another is likely to be very imprecise (and possibly misleading), unless there is a high degree of similarity between the 'study' and 'policy' sites (in terms of the environmental features, policy outcomes and population characteristics). Although the acceptable margin for error will generally depend on the policy context (rather

than whether differences are statistically significant), high errors — sometimes several orders of magnitude — have been found even when differences across sites are moderate. This may be due to the heterogeneous nature of environmental outcomes and the fact that there are many unobserved factors and potential measurement errors that can affect estimates of willingness to pay, as indicated by the low explanatory power of many statistical models of willingness to pay (Brouwer 2000; Kirchhoff, Colby and LaFrance 1997).

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