The Economic Significance of Environmental Resources: A Review of the Evidence

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The products and services of many environmental resources do not enter commercial markets and remain unpriced. The absence of market values presents a major difficulty for environmental projects in competing for ever-tightening budgets. In response to the need for assessing costs and benefits, a number of methods have been devised and applied to generate estimates of the value of unpriced resources.

This paper reviews briefly the approaches employed for generating value estimates for unpriced environmental resources, with particular attention being paid to the contingent valuation method. Estimates of value for environmental products and services for both the United States and Australia are presented. This evidence clearly demonstrates that a wide range of unpriced environmental resources have significant economic value to the community. The paper concludes with a discussion of the implications for project development, funding and policy in Australia.

1. Introduction

Population and industrial growth in recent decades have increased the use of the earth's natural resources and placed increasing pressures on them. In addition, rising incomes and increased leisure time have led to increasing demand for the services of environmental resources, largely in the forms of tourism, recreation and waste disposal.

The expansion of urbanisation and agriculture associated with rapid population growth has also had a supply side effect. The stock of natural environments has decreased.

The inevitable consequences of increasing demand and decreasing supply are a rise in the value of natural resources, and increasing pressure for access to and use of these resources and the services they provide, either as direct inputs into production processes or in the form of amenity services which generate utility for consumers.

In Australia, increasing public awareness and concern has been expressed particularly since the 1960s relating for example to exploitation of kangaroos, saving Lake Pedder, mining and oil exploration on the Great Barrier Reef and in national parks, woodchipping, logging of rainforest and the destruction of green belts in urban areas. In response to this concern, a considerable body of legislation has been enacted by the Commonwealth and States covering topics ranging from soil conservation, environmental protection and clean waters, to noise abatement and relics preservation (for a chronology of environmental events in Australia since 1965, see Department of Arts, Heritage and Environment 1986). More recently the establishment of and proposals for establishing Environmental Protection Authorities at federal and state levels has given further substance to the political response to community concerns.

The growing public awareness has also been reflected in rapidly increasing membership of the National Conservation Foundation in Australia (see Beeton and Collins 1985). Perhaps a more significant event, but not unrelated, has been the formulation of a National Conservation Strategy for Australia and its endorsement by each of the Commonwealth and State Governments with the exception of Queensland.

At a regional level, there has long been concern with environmental issues in the Murray Darling Basin particularly in relation to water quality and land degradation. The Basin accounts for a major part of Australia's rural production, and is the source of water for large numbers of irrigation farmers, urban populations and industrial users. In

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recognition of the critical importance of the Basin's natural resources, the fact that these extend beyond legislative and institutional boundaries, and the urgency for action in addressing key environmental issues in the Basin, the Murray-Darling Basin Ministerial Council was formed in 1985. The Council has the general objective of promoting and coordinating effective planning and management of the Basin's environmental resources and specific goals relating to water quality, land degradation and conservation and preservation of the natural environment.

The aims of this paper are to present evidence on the economic value of unpriced environmental resources, and to consider the implications this has for resource management. The estimates of value presented cover a diversity of environmental resources and reflect the ingenuity of economists in developing methods for assigning dollar values to the wide range of intangible goods and services generated by such resources.

To establish a link between the theory underpinning more traditional methods of economic evaluation and the approaches developed to generate these values, the first part of the paper considers sources of value and the special characteristics of environmental resources. On the basis of this preliminary foundation, approaches for valuing environmental resources are briefly reviewed, prior to the presentation and review of estimates of value for a wide range of such resources.

2. Sources of Value

The capacity to satisfy needs and desires is basic to the value that individuals place on different goods and services. The perceived benefits from a given consumption activity will vary between individuals and hence the concept of value is highly subjective (Brown 1984). Despite this, decisions are required and are made by producers and resource managers. These decisions are, by and large, perceived to meet the needs and desires of the individuals in a society.

The process of choice under the constraint of a limited budget provides a vehicle for determining the value to society of different goods and services. This process enables the translation of subjective individual values into more objective estimates of net social benefit or net monetary benefit.

Environmental resources generate utility directly for consumers when the services available from such resources are combined, for example, with the activities of tourism and recreation. Utility for consumers is also generated indirectly by production processes involving environmental resources, such as agriculture and scientific research, which eventually yield products and services for consumption.

Thus environmental resources are no exception to the view that their source of value lies in the benefits they confer on the users of these resources. Use in this context is broadly defined to include production and consumption, and to include direct use, *e.g.* as inputs to production or consumption, and indirect use in the sense of using the services supplied by such resources, *e.g.* amenity, scenic beauty. In addition, many environmental resources possess value unrelated to their use.

Characteristically, many environmental resources offer benefits which are intangible and qualitative in nature. Assessment of the value of these benefits is a formidable task. To quote Kellert (1984, p. 355): "... we are confronted by the dilemma of generating prices for the priceless, of quantifying the unquantifiable, of creating commensurable units for things apparently unequatable." And yet, if the task is not addressed, the benefits of environmental and wildlife resources will tend to be ignored by default. This is because "the lack of empirical criteria for measuring *all* environmental and wildlife values tends to result in (1) little more than superficial consideration of intangible and qualitative values, and (2) far greater emphasis on quantifiable values, particularly those measurable in money terms" (Kellert 1984, pp. 356-7).

In the interests of more informed decision making and efficient resource allocation, it is important to consider what can be done to resolve the measurement problem. In this context it will be useful to consider the different components of value.

Following Randall (1985) and Wilks (1990), components of value can be identified as follows:

- current use value;
- future use value - the uncertainty of future use leads to two kinds of values:
  - option value: an individual who expects to use the environmental resource but is uncertain about the future supply of the resource may rationally pay a premium to ensure future supply. If however, the uncertainty relates to future demand then the individual may expect a discount in the expectation that a purchased option may not be used, and hence the option value may be negative (see also Freeman 1984). Option value is unambiguously non-negative only in a case of certain future demand and uncertain future supply;
  - quasi-option value: if development is irreversible but new information about the value of preservation may emerge in the future, then there is a quasi- option value which is positive with respect to preservation now. Quasi-option value is thus the gain from being able to get information about future benefits by keeping open the option to preserve or develop, and can be viewed as the value of information conditional on not proceeding with development initially (see Clarke 1991);
- existence value: which is separate from and additional to use value, is derived from the knowledge that an environmental resource will continue to exist, and requires the availability of information about the resource;
- bequest value: Walsh *et al* (1984) identify a further non-use benefit as bequest value - the willingness to pay for the satisfaction derived from endowing future generations with given environmental resources;
- vicarious value: the utility gained from knowing about the consumption of others (Mitchell and Carson 1989; Pearce *et al* 1989)[1].

The various components may be categorised in a taxonomy of total economic value. There appears however to be disagreement in the literature about the precise nature of the interrelationships between the components. For example, Pearce *et al* (1989, p. 62) view vicarious value as a sub-component of option value. In contrast, Mitchell and Carson (1989, ch. 3) and Carson (1991) argue that option value and quasi-option value are not components of value in their own right, and should be regarded as technical corrections to *ex post* total measures of value to convert them to *ex ante* values. Mitchell and Carson (1989) also regard bequest value as a form of stewardship value which together with vicarious values are more appropriately categorised as sub-components of existence values.

On the basis of their discussion, a taxonomy of value can be derived as follows:

Total Economic Value:

• Use Value	- Direct Use or Consumptive Benefit *e.g.* recreation, harvesting, disposal
	- Indirect Use or Non-consumptive Benefit *e.g.* aesthetic viewing
• Existence Value	- Vicarious Benefit
	- Stewardship Benefit *e.g.* bequest

Non-use or preservation value has traditionally been defined as the sum of option, existence and bequest values (Greenley, Walsh and Young 1981; Loomis 1987a; Sutherland and Walsh 1985; Walsh *et al* 1987). Mitchell and Carson (1989, p. 60) suggest however that "unique estimates of the different benefit categories and subcategories generally do not and cannot exist" because they are interrelated, and hence cannot be considered to be additively separable without arbitrary restrictions.

Value of course is essentially an outcome of the interaction of demand and supply. In this context, Krutilla and Fisher (1975, p. 57) persuasively argue

[1] Wilks (1990, p.6) however defines vicarious value as "the inherent consumption of the environmental amenity through print or other media."

that the value of the environment's amenity resources can be expected to increase relative to that of commodities:

> "The services of amenity resources generally enter directly the utility functions of consumers. Since there is no production technology other than natural processes, we cannot look for advances in the "state of the art" to augment the supply and reduce the scarcity value of these resources. The value of an environment's commodity resources, on the other hand, may well be reduced through technical progress which makes substitutes (in production) increasingly abundant."

In considering the value of environmental resources, there are accordingly many factors which need to be taken into account. The issue of evaluation is taken up in the following section.

3. Approaches to Evaluation

The assessment of value for goods and services, over which property rights can be assigned, typically relies on the identification of market price or unit value. At a given price, however, there will tend to be some consumers who would be willing to pay a higher price than that determined in the market place, and so an evaluation based solely on prevailing market price would tend to understate the value of that good or service. This problem can be resolved by assessing how market demand would change with a price change and using the information to compute a market demand schedule which will allow value to be assigned (see Snaith (1972) for an example of an application to heritage resources, and Thomas and Syme (1988) for an application to water resources).

The value which a producer places on an environmental resource as an input in the production process, *e.g.* agriculture or mining, may be viewed analogously to that of consumer demand, with the demand for the input being a derived demand, *i.e.* derived from the demand for the product being produced.

In theory, the approach to evaluation appears reasonably straightforward, if we sidestep the difficulties associated with the concept of value and the assignation of value identified by Brown (1984). However, many environmental resources possess characteristics which make their evaluation less than straightforward.

Many environmental resources provide multiple benefits which stimulate multiple use. Because these uses may conflict, *e.g.* the logging of a forest may conflict with it's scientific research and amenity values, the evaluation of that resource cannot proceed on the basis of simply summing the values of the benefits associated with the different uses as if the uses are independent of each other. An assessment is required of the value in use contingent on given values for other levels of use. That is, the interrelated nature of the benefits needs to be taken into account. In this context, the studies by Boyle and Bishop (1987a), Cory and Martin (1985), and Keith and Lyon (1985) on valuing wildlife are relevant.

In a situation in which multiple uses are competing, the socially-optimal levels of each use are required, which implies analysis at the margin and the use of dynamic programming. In an application of this approach to an irreversible decision problem, Kennedy (1987) argues that analysis of the actual decision process is more satisfactory in addressing decision problems under uncertainty rather than attempting to assign arbitrary values to quasi-option value.

The value of a productive resource such as land is conventionally viewed as the discounted value of the stream of net benefits over time. In the case of living resources and renewable non-living resources, the time scale implied is infinite. To compute the net present value of the stream of benefits from a resource which is renewable is likely to involve questions about intergenerational equity as well as economic efficiency. The issue of choice of discount rate may also arise given that adjustment of the discount rate to allow for intergenerational equity considerations has been practised in the past - for a discussion of this issue see Pearce *et al* (1989), Collins and Young (1991) and Young (1992).

The holistic nature of environmental systems suggests that indirect as well as direct benefits should be taken account of when evaluating environmental resources to reflect the interdependencies of the system. Indeed, Mitchell and Carson (1989, p. 67), take the view that individuals make a "holistic judgement" when asked to value an amenity, and do not go through "a mental process in which they value each of the relevant benefit categories and subcategories and then combine these values in their minds to arrive at a total value...". Given also the inconsistencies identified in the traditional taxonomies of value components (see Smith 1987; Boyle and Bishop 1987b; Randall 1987; and Randall and Kriesel 1988), there may be merit in focusing on total value measures, since these are typically what are required for decision making.

A key concept in the valuation of environmental resources is willingness to pay (WTP). Total willingness to pay is the sum of consumer expenditures plus consumer surpluses. Market price is thus only an indicator of social value. The flatter or more elastic is the demand curve, reflecting a high degree of substitutability with other products or services, the lower will be the consumer surplus value and the closer will market price be to a measure of willingness to pay.

Following Sinden and Worrell (1979, p.33), two other points can be made. The first is:

Social Value = Utility - Disutility

where disutility represents what has to be given up to get utility, *i.e.* the opportunity cost (OC). If for example, the government decides to preserve a wildlife habitat rather than clearfell for timber, the OC is the value of the timber which is not produced.

Again following Sinden and Worrell (1979, p. 119), the conceptual definition of economic value given above can be made operational by using the two concepts willingness to pay (or utility) and opportunity cost (or disutility), *i.e.*:

Social Value = WTP - OC

The problem is to measure WTP and OC. This may be done as two separate benefit-cost exercises, where for example two activities compete for the use of the same resource. The WTP for one activity in this situation will be the OC of the other activity. A recent example is the analysis of the value of mining in the Kakadu Conservation Zone and the value of the alternative 'activity' of preservation (see Imber *et al* 1991).

It may be noted that for present purposes, it is assumed that WTP is interchangeable with willingness to accept compensation (WTA) for undesired changes or disbenefits, and that either WTP or WTA will be appropriate depending on the circumstances being analysed.

The issue of the relationship between WTP and WTA has attracted the attention of a number of analysts. Randall (1982) shows that WTA-WTP≥0 for normal goods, and indicates that substantial error may be introduced by use of inappropriate value measures for proposals involving significant changes in environmental resources; *i.e.* WTP should be used for increments in value and WTA for decrements (see also Bishop *et al* 1983; Randall *et al* 1983). Experimental evidence presented by Knetsch and Sinden (1984) demonstrates a large disparity between WTA and WTP measures of value. Coursey *et al* (1987) however provide qualifying experimental evidence which suggests that the disparity narrows in mature markets in which individuals can gain experience from repeated transactions. In response, Knetsch and Sinden (1987, p.694), point out that "the underweighting of opportunity costs relative to out-of-pocket costs, which give rise to the evaluation disparity, is a very strong intuition of most people" and that the lack of the equivalence of opportunity and other costs will continue to result in disparate valuations which reflect welfare changes. The implication of this is that in comparing value estimates, it is important to compare like with like based on consistent measurement.

A range of valuation methods for valuing environmental resources is discussed in detail by Hufschmidt *et al* (1983) and Sinden and Worrell (1979). In the present context, these can be viewed in terms of three basic categories (Randall 1985, p.19):

- in well functioning markets and for small quantity changes relative to total quantity in the market, price will be an adequate value indicator. For large quantity changes, WTP may be estimated from demand schedules derived from market data.

- for unpriced goods, there may exist complements or substitutes whose markets can be analysed to reveal information about the value of the unpriced good. The weak complementarity approaches, such as the travel cost method, and the implicit price approaches based on the hedonic price method fall into this category (Hoehn and Randall 1987).

- hypothetical or experimental markets can be devised for unpriced goods and used in survey or experimental contexts to generate value information.

Unpriced goods that provide non-use benefits are rarely exchanged in well functioning markets. In consequence, the methods covered by the third of these categories are usually the most useful for the estimation of non-use benefits.

Three methods which have been frequently used for the estimation of environmental benefits are the contingent valuation method (CVM), the travel cost method (TCM) and the hedonic price method (HPM).

The contingent valuation method involves asking individuals in survey or experimental settings to reveal their personal valuations of increments or decrements in unpriced goods (see Randall *et al* 1983; Bishop *et al* 1983; Mitchell and Carson 1989; and Bennett 1991 for a discussion of the merits and weaknesses of this approach). Recent examples using this approach are listed in Mitchell and Carson (1989) and Wilks (1990). Walsh *et al* (1984) for example describe the measurement of preservation value of wilderness by CVM by which a survey of a sample of the affected population was undertaken to obtain data on maximum WTP contingent on changes in the availability of wilderness.

The contingent valuation method relies on the generation of value estimates for hypothetical situations and responses may be subject to various types of bias (see Mitchell and Carson 1989; Wilks 1990; Hoevenagel 1991 for a discussion of these). Critical reviews by Schultze *et al* (1981b), Heberlein and Bishop (1986), and Cummings *et al* (1986) provide cautious support for the approach. Hoehn and Randall (1987) demonstrate the theoretical consistency between CVM and mainstream microeconomics models of choice, and establish that satisfactory benefit-cost estimates can be derived directly from CVM data. It is widely acknowledged however that considerable care is required in constructing a questionnaire and in eliciting responses from individuals sampled. The recent debate over the accuracy of the results of the CVM study for the Coronation Hill mining inquiry revolved around these issues (Resource Assessment Commission 1991). By way of qualification, Smith *et al* (1986, p. 289) point out that judgement is an inevitable component of any empirical model of an economic process. Specification selection and statistical inference typically require subjective judgement, and this does not render indirect methods of estimating benefits (as opposed to direct methods such as CVM surveys) infeasible for practical purposes.

Because CVM is the only available method for generating non-use values, the accuracy of such values cannot be tested empirically, and this has led to the specification of 'reference operating conditions' (Cummings *et al* 1986) and a proposal for a 'best practice standard' with associated criteria (Mitchell and Carson 1989, pp. 299-304) as means for achieving greater validity of CVM results. Despite the problems, Conrad 1986, (p. 1275), concludes "contingent valuation is a clumsy awkward elephant. But it seems to be the only elephant we've got".

The travel cost method has been widely used for valuing recreational facilities and is based on the assumption that the time and money spent travelling to a free or low-cost recreation site give a measure of the consumer's true valuation of that site, and can therefore be used to measure the economic value of an existing recreation site (Hufschmidt *et al* 1983, p.171). Examples of TCM applications are given in Everett (1979), Desvousges *et al* (1983), Sorg and Loomis (1984), Loomis (1987a) and Pearce *et al* (1989).

Various refinements to the approach have been developed to allow for the existence of "competing" or substitute sites (see Mendelsohn and Brown 1983) and to measure the area effectively usable for recreation rather than taking the area nominally designated (see Clawson 1984). An innovative approach particularly useful for addressing the substitute sites problem is the hedonic travel cost method (HTCM) described by Mendelsohn and Brown (1983). This approach synthesises the TCM and HPM by treating sites as bundles of characteristics and analysing visitor choices between sites in terms of expenditure incurred for the different sets of site characteristics. Examples of applications of HTCM to various forms of wilderness recreation are given in Sorg and Loomis (1984) (see also Table 2).

Estimates of benefits have however tended to be extremely sensitive to the treatment of time (as opposed to on-site costs) and consensus on the appropriate approach has still to be reached. A recent paper proposing valuation of travel time based on opportunity cost which allows differentiation between individuals with fixed work hours and those with discretionary time offers hope of some progress in resolving this issue (see Bockstael *et al* 1987). Nevertheless, the fact remains that some people have a negative opportunity cost for travel - that is, they gain utility from the travel activity and would be willing to pay for more travel time (see Mitchell and Carson 1989, p.79). How severe the upward bias this generates in TCM estimates remains to be resolved.

The hedonic price method, like TCM, estimates value by inference from consumers' purchases of market goods. Statistical analysis is used to identify the extent to which observed differences in market values are attributable to environmental differences. For example, differences in property values due to air pollution have been used to generate values for air quality (for a review of the HPM method, and examples of applications see Streeting 1990; for a recent application, see Coelli *et al* 1991).

Use of the WTP approach alone as a means of measuring value of benefits has been questioned by Cory (1985) and by Bockstael and Strand (1985) who make the point that benefit estimates are dependent on existing income distributions, and questions of equity are rarely considered. The further point is made that if the value of benefits (WTP) exceeds costs (OC), this does not guarantee that an actual increase in social welfare will occur, since the use of the WTP criterion without compensation paid will tend to redistribute income to more wealthy individuals. They conclude that distributional criteria are required as well as the efficiency criteria represented by use of WTP and OC.

The actual choice of valuation method should however be determined by the situation being investigated. Cory and Martin (1985) point out that TCM and CVM are useful for valuing mutually exclusive uses. Additional information is required to make such estimates relevant in a multiple use context. In their study of valuing wildlife for efficient multiple use, they conclude that "when outdoor recreation based on wildlife availability can coexist with extractive or production-oriented activities, subject to constraints imposed by the carrying capacity of land, efficient multiple use management requires marginal valuation estimates of wildlife numbers to compare to associated net benefit impacts occurring for competing uses".

In addition, Keith and Lyon (1985) offer the view that recreation valuation studies focus on variables which are not appropriate for use in wildlife management decisions and that information is required on the value of a change in the stock of wildlife compared to its cost.

In conclusion it may be said that there are significant conceptual and practical difficulties in valuing environmental resources, but that methods are available for overcoming these difficulties in some degree. If environmental resources are important, then it is important to make the attempt to evaluate them and to enable decision makers to incorporate estimates of their value in the decision making process. In this way more informed decisions and a more efficient allocation of resources can be achieved for the benefit of society at large.

4. Some Estimates

By far the bulk of the investigations into wildlife and environmental values have been undertaken in the United States (US), although the growing frequency of such studies in Europe and Australia suggests that the WTP and CVM approaches may be gaining greater acceptability elsewhere (see Carson 1991; Hoevenagel 1991; Kuick *et al* 1991; and Wilks 1990). The motivation has been partly to meet a growing demand in resource management planning to take account of economic efficiency considerations and partly the development of a theoretical and empirical basis for a general non-market valuation framework (Sorg and Loomis 1984, p. 1; Loomis and Walsh 1986).

The environmental resources for which empirical values have been assessed have ranged from water and air through wildlife and wilderness to wetlands and agricultural land. The majority of the US studies employ the CVM or TCM methods. Some estimates however are based simply on aggregate expenditure (*e.g.* Payne and DeGraaf 1975), while others adopt a more innovative approach. Lugo and Brinson (1978) for example quote estimated values of salt water wetlands based on the assessed value of the energy flow characterising these ecosystems.

Many of the earlier studies represent building blocks in the development and application of the CVM approach, designed to establish the credibility and feasibility of the approach. A feature of earlier studies was the generation of estimates for the different components of value and the summing of these to derive estimates for total non-use (preservation) value and total value. Examples of such estimates are given in Table 1.

Characteristic of these estimates is the variation in values between different resources even after allowing for inflation (see Sorg and Loomis (1984) for inflation adjusted comparisons of value estimates), both within and between classes of resource, *e.g.* water quality, wilderness and endangered species. This is of course not unexpected since many factors are likely to be influential including location, significance in terms of uniqueness, beauty *etc.* as well as income levels, and price or cost of access, but it does imply a need to adopt a case by case approach.

A further characteristic is the tendency for the sum of non-use values to exceed the direct use (recreation) values. In the South Platte River Basin study cited in Table 1, for example, non-use benefits total \$US76/household compared to \$US57/household for recreation value. Similar comparisons apply for the wilderness and endangered species studies also cited.

The practice of summing option, existence and bequest values to obtain a total non-use value has however been questioned on at least two counts. Carson (1991) points out that option value is a technical correction to convert an *ex post* estimate to an *ex ante* value, and does not form a component of non-use value. Because CVM estimates represent expected or *ex ante* WTP, the future use component of this WTP will already take account of the uncertainty covered by option value. The correct component to include, according to Mitchell and Carson (1989), is option price which is consistent with an *ex ante* value. The wildlife studies by Brookshire *et al* (1983) and Stoll and Johnson (1984) present option price estimates as combined recreation and option values to address this issue (see Table 1). It may also be argued that bequest and vicarious values are special forms of existence value (*e.g.* Mitchell and Carson 1989; Carson 1991). If option and bequest values are ignored in Table 1, and attention is focused on future use (*i.e.* recreation and option values) and existence values as Brookshire *et al* have done, then non-use values will not always exceed (future) use values.

Perhaps a more telling point is that the various components of value are unlikely to be independent of each other and hence it is invalid simply to sum component values to obtain a total value. Boyle and Bishop (1987a) address this problem by specifying a functional relationship between total value and components of value. Until the nature of the relationship between components of value is resolved, they argue that the needs of decision makers will be met by generating estimates of total value, since that is where the primary interest of decision makers will lie, rather than in component values (see also Randall and Kriesel 1988). Nevertheless, one might expect non-use values to be significantly larger than use values for certain resources which are major and unique, *e.g.* the Great Barrier Reef, since non-use values will tend to be held by a significantly larger number of people than those

Table 1: Selected CVM Estimates from US Studies of Recreation and Non-Use Values

Study	Unit of Value	Recreation Value	Option Value	Existence Value	Bequest Value	Preservation Value
Water Quality						
Merrimack River Basin Oster 1977	1973 $US/resident	12				
South Platte River Basin	1976 $US/household	57	34	25	17	76
Greenley *et al* 1981	Basin Aggregate $USm	26.5	10.5	14.4	9.8	35
Flathead River/Lake	1981 $US/household	7	11	20	26	57
Sutherland and Walsh 1985	Montana State Aggregate 1980 $USm		27.7	20.3	33.8	81-8
Monongahela River Desvouges *et al* 1983	1983 $US/household		10-38	42 (non-users) 66 (users)		
Wildlife						
Grizzly Bears Brookshire *et al* 1983	c1982 $US/respondent	21		24		
Bighorn Sheep Brookshire *et al* 1983	c1982 $US/respondent	23		7		
Endangered Species						
Whooping Cranes Stoll & Johnson 1984	$US/household	10-17		1 - 9		
	US Agregate $USm				573	
All fish and wildlife on US threatened and endangered species lists Walsh *et al* 1987	1983 $US/ Colorado household	17	12	13	16	41
Wilderness						
Colorado wilderness areas - 1.2m acres	1980 $US/ Colorado household	14	4	5	5	14
10m acres Walsh *et al* 1984		14	9	11	12	32
Prime Agricultural Land						
Amenity Value - South Carolina Bergstrom *et al* 1985	1982 $US/household					6-9

able to visit. Willingness-to-pay estimates for the Great Barrier Reef using the CVM approach (Carter 1987) testify to such a situation (see Table 4).

A further feature in Table 1 is the presentation of aggregate estimates for the populations from which samples have been drawn. Even though annual values per respondent or per household may be relatively small, the aggregation of these amounts to population levels produces a large absolute value. The Basin aggregate non-use value for water quality in the South Platte River Basin for example amounts to $US35m, whilst the corresponding Montana State aggregate for Flathead River and Lake was $US81m. In the case of whooping cranes, Stoll and Johnson (1984) estimate the national US aggregate preservation value to be $US573m.

Even allowing for the fact that the total non-use values shown in Table 1 may be overstated for reasons referred to above, the aggregate population values can be expected to be significant. Some examples are shown in Table 2. The Boyle and Bishop (1987) estimates for two endangered species in Wisconsin, and the Kriesel and Randall (1986) estimates for control of air and water pollution at the national level indicate that the community places significant values on environmental resources.

Given that hunting is a major form of environmental resource use in the US, estimates of value per activity day for various forms of hunting *e.g.* big game (deer, elk, antelope), small game, waterfowl and various types of fishing, have received some emphasis. Estimates of value from a number of studies for such uses are shown in Table 2. The approaches used to generate these estimates consist mainly of CVM and TCM and indicate that those engaged in sporting recreation place a consistently high value on the environmental resources which enable them to pursue their sporting activity. This conclusion appears to apply also to wilderness recreation activities for which visitor day average values ranged from over $US12 to almost $US75 (see Table 2).

As with the range of values specified for recreation activities, the range of values for other environmental resources shown in Table 2, wetlands, and visual air quality in selected national parks, is wide and reflects the influence of a number of factors. For example, it seems likely that in the case of national parks, the uniqueness, or non-substitutability, and significance of a resource such as the Grand Canyon accounts for an important part of the differences in value.

With regard to the study of the impact of reduced ozone levels on US agriculture by Adams *et al* (1986), the estimate of benefits from increased yields provides an interesting and significant perspective to the not uncommon view that the cost of fixing the environment will be large. Such a view ignores the substantial level of costs already being incurred by the continuing degradation of the environment, and hence the substantial potential benefits of fixing the environment, and reinforces the notion that rhetoric needs to be replaced by credible analysis. Costs need to be weighed against benefits for informed decision making to take place.

The evidence presented in Tables 1 and 2 suggests that the US community places a substantial value, in terms of both use and non-use benefits, on environmental resources, and that the range of resources over which significant values are held is both wide and varied.

The question then arises whether such a statement applies to Australia. Certainly, the evidence is more limited because fewer studies have been conducted in Australia. This perhaps reflects the relatively recent development of more widespread public awareness and appreciation of the environment as well as scepticism on the part of professional economists and policy makers about the validity of the valuation approaches used. Nevertheless, the issues of resource use and lack of data on the value of benefits of unpriced environmental resources are common to both the US and Australia, and the growing number of studies presenting value estimates attests to the growing demand for such information.

A selection of estimates from studies conducted in Australia is presented in Tables 3 and 4. Because of differences in orientation and measurement, not all of the estimates are directly comparable. For example, damage cost estimates are not a direct measure

Table 2: Value of Environmental Resources: Selected US Estimates

Study	Method	Estimated Value
Wildlife		
Elk - Brookshire *et al* 1980	CVM	Average value per hunter per annum of an increase in sightings of elk from 1 to 5: $54 (1977-78 US $)
- Cory and Martin 1985	Programming approach	Marginal annual net benefit of elk - $106 (1979 $US); corresponding value for competing cattle - $37 (1985 $US).
Utah mule deer - Keith and Lyon 1985	Household production function	Marginal benefit of one deer - $40 (1982 $US)
Nongame birds - Payne and DeGraaf 1975	Expenditure data	Total direct expenditure in US in 1974 - $500m.
Small game - Sorg and Loomis 1984	CVM/TCM/ HTCM/HPM	Average value per activity day hunting ranged from $15.74 to $42.58 (1982 $US)
Waterfowl - Sorg and Loomis 1984	CVM/TCM/ HPM	Average value per activity day hunting ranged from $16.26 to $84.73 (1982 $US)
Big game: - Sorg and Loomis 1984		
Deer	CVM/TCM	Average value per activity day hunting ranged from $18.40 to $131.80 (1982 $US)
Elk	CVM	Average value per activity day hunting - $36.37 (1982 $US)
Antelope	CVM/TCM	Average value per activity day hunting ranged from $18.81 to $19.68 (1982 $US)
All	CVM/TCM/ HTCM/HPM	Average value per activity day hunting ranged from $25.86 to $75.78 (1982 $US)
Endangered Species		
Bald Eagle	CVM	Aggregate preservation value for all Wisconsin taxpayers: c$28m (1984 $US)
Striped Shiner - Boyle and Bishop 1987	CVM	Aggregate preservation value for all Wisconsin taxpayers: c$12m (1984 $US)

Table 2: Value of Environmental Resources: Selected US Estimates (cont.)

Study	Method	Estimated Value
Fish		
Trout - Charbonneau and Hay 1978	CVM	Average value per angler day - \$21 (1975 \$US)
Bass - Charbonneau and Hay 1978	CVM	Average value per angler day - \$19 (1975 \$US)
Trout - Vaughan and Russell 1982	TCM	Average value per angler day ranged from \$11 to \$19 (1979 \$US)
Salmon/Steelhead: Idaho, Oregon and Washington - Sorg and Loomis 1984	CVM/TCM	Average values per activity day ranged from \$25.92 to \$99.02 (1982 \$US)
Adirondack fishery - Mullen and Menz 1985	TCM	Average value per angler day - \$20 (1976 \$US)
Lake Ecosystem		
Mono Lake, California - Loomis 1987b	CVM	State residents willingness to pay a higher water bill for protection of Mono Lake: Monthly household average - \$12.85 (1985 \$US) Annual State aggregate - \$1.5b (1985 \$US)
Wetlands		
Freshwater wetlands, Massachusetts - Gupta and Foster 1975	Opportunity cost	Estimated annual value of wetlands at 'medium' level of benefits - \$1610 per acre (1972 \$US)
Salt water wetlands - reported life support values quoted in Lugo and Brinson 1978	energy flow approach	Forested wetland: annual value - \$5900 per acre (c1971 \$US) Tidal marsh: annual value - \$4150 per acre (c1973 \$US) Mangrove forest: annual value - \$915 per acre (c1977 \$US)
Prairie Wetlands - Hammack and Brown 1974	CVM	Average value per hunter per season - \$256 (1968 \$US)
Visual Air Quality - National Parks		
Grand Canyon - Schulze *et al* 1981a	CVM	Willingness to pay to preserve visual air quality per household per annum - \$86 (1980 \$US)
Four Corners - Schulze *et al* 1981b	CVM	Willingness to pay to preserve visual air quality per household per annum - \$50 to \$85 (1980 \$US)
Lake Powell - Schulze *et al* 1981b	CVM	Willingness to pay to preserve visual air quality per household per annum - \$2.95 (1980 \$US)
Farmington - Schulze *et al* 1981b	CVM	Willingness to pay to preserve visual air quality per household per annum - \$57 to \$82 (1980 \$US)
National Park		
Jemez Mountains - Schulze *et al* 1981b	CVM	Willingness to pay to prevent development per visitor party day - \$2.45 (1980 \$US)

Table 2: Value of Environmental Resources: Selected US Estimates (cont.)

Study	Method	Estimated Value
Pollution Control		
Air and water pollution - Kriesel and Randall 1986	CVM	Willingness to pay for 25% decrease in national air and pollution loads: - per household per annum - $694.42 (1984 $US) - aggregate for the nation - $60.4b (1984 $US)
US agriculture - Adams *et al* 1986	Spatial equilibrium model	Benefits of 25% reduction in 1980 ambient ozone via increased yields - $1.7b (1980 $US)
Forest Management		
Recreation in the front range of the Colorado Rocky Mountains - Walsh *et al* 1989	CVM/TCM	Average willingness to pay per trip at existing tree density: $20.80 (1980 $US) For a 20% loss in tree density there was a 16.8% reduction in benefits. Corresponding TCM estimates of value for existing tree density were $22.59 (OLS) and $17.72 (2SLS).
Wilderness		
Recreation - Sorg and Loomis 1984	CVM/TCM	Average value per visitor day ranged from $12.78 for hiking in Colorado to $73.93 for backpacking in Oregon
Utah - Pope and Jones 1990	CVM	Average willingness to pay per household per annum ranged from $52.72 for preservation of 5% of Utah as wilderness to $92.21 for preservation of 30% of Utah as wilderness (1986 $US); Utah State aggregate annual WTP for preservation ranged from $26.7m for 5% level to $46.7m for 30% level (1986 $US)
Instream Flow		
Colorado - Daubert *et al* 1979	CVM	Marginal 1978 $US value/acre ft of flow/respondent: $15.75
Colorado - Walsh *et al* 1980	CVM	Marginal 1978 $US value/acre ft of flow/respondent: $19.04

of use value, whereas WTP estimates as represented for example by the payments made for habitat preservation and eucalypt preservation (Table 3) are likely to encompass both use and non-use values. In addition, the annual estimates are not directly comparable with present value estimates, which represent a capitalisation of the stream of annual values. Present value estimates are shown in Table 3 for the value of benefits from tourism on Kangaroo Island (Lothian 1985) and in the form of an increase in land value attributable to soil conservation works (King and Sinden 1988). However, as with the estimates for the US shown in Tables 1 and 2, the aim in Tables 3 and 4 is to present estimates of value for a range of resources which are indicative of the magnitude and variation in values characterising environmental resource valuation in Australia.

A common feature of many of the Australian studies is their policy orientation and their concern with the damage costs of resource degradation. In particular, there has been a strong focus on two of Australia's critically important productive resources, soil and water (Table 3). For example, the average annual damage cost of salinity in water supplies in the Murray Darling Basin was estimated to be $36m, whilst the annual damage cost of dryland salinisation in Australia was estimated to be $22m. The order of magnitude of annual damage cost due to soil degradation on arable land appears to be even higher judging by the Sinden *et al* (1986) estimate for soil erosion in a single New South Wales shire (Manilla Shire) of $33.3m and the estimated cost of lost production in Western Australia of $94m quoted by Upstill and Yapp (1987).

Most of the estimates in Table 3 take the form of annual damage costs representing the loss in commercial production due to environmental degradation. For rural producers, damage costs translate directly into an upward shift in the supply curve and hence can be interpreted as losses in consumer and producer surplus. The relative magnitudes of these losses will depend on the elasticities of supply and demand for the commodities concerned. A similar comment applies to the damage costs borne by local government as exemplified in the study by Sinden *et al* (1986) on the cost of repairing sedimentation damage. To the extent that damage costs translate into higher rates and taxes, surplus losses will result for both producers and consumers.

In the case of damage costs borne by consumers, for example as a result of more saline water causing increased maintenance costs of appliances, there will be a loss of surplus associated with the effective increase in the price of water consumption, apart from any direct reduction in utility associated with altered taste and visual attributes of the water.

In terms of environmental resource values, the estimates of damage cost represent only indirect indicators of use value. In the literal sense, they represent reductions in use value. They may also however be interpreted as upper bound values of benefits from conservation measures, and hence provide a positive indication of use value at the margin. In this sense, they are likely to form lower bound estimates of use value for the resource in an undamaged state. To the extent that users also attribute non-use values to the same resource, then their WTP for conservation may of course exceed the damage cost estimate. In the case of the study of tree retention in the Loddon catchment (Greig and Devonshire 1981), the salinity damage costs of clearing for example are likely to be an underestimate of the actual value of the trees to the community in that catchment because there are likely to be additional sources of use and non-use value associated with trees.

Thus estimated damage costs provide relevant information on the value of benefits of improving the management of environmental resources but represent only an indirect estimate of the value of such resources to the community. They do however have the advantage of being based on market information. A similar comment applies to most of the other estimates of benefits relating to national parks and vegetation in Table 3. The study by Ulph and Reynolds (1981) which produced an estimated recreation value for the Warrumbungle National Park of $100 per visitor day is noteworthy as one of the few TCM studies conducted in Australia (see also Carter 1987). Despite their *ad hoc* nature, the value estimates provided for wombat habitat preservation by Sinden and Mackay (1979), and for eucalypt preservation by Sinden (1986), provide examples of community concern about existence

Table 3: Value of Environmental Resources: Some Australian Evidence

Study	Estimated Value
Water Quality	
Murray Darling Basin - Dwyer 1984	Annual value per EC unit change in average salinity at Morgan - $70K (based on urban and agricultural damage costs)
Australia - Dept. of Resources and Energy 1983	Average annual damage of salinity in water supplies - $36m
Soil Quality	
Erosion	
- NSW - Junor 1984	Loss of 75mm of soil due to erosion resulted in annual reductions in wheat yield of 9.5% and protein content of 21.5%
- NSW - Sinden and Yapp 1987	Lost agricultural income from 4.7m ha of combined sheet and gully erosion: $73m
Salinity	
- NSW - saline scalding	decline in annual productivity - $2.0m
- saline seepage	decline in annual productivity - $0.4m
- Standing Committee on Soil Conservation 1982	
- Berriquin, Wakool Irrigation Districts, NSW	Annual loss in agricultural production - $4m
- Grieve *et al* 1986	
- Australia - dryland salinisation	Damage cost per annum - $22m
- irrigation soil salinity	Damage cost per annum - $6m
- Dept. of Resources and Energy 1983	
Waterlogging	
- Berriquin, Wakool Irrigation Districts, NSW	Annual loss in agricultural production $10m
- Grieve *et al* 1986	
Sedimentation	
- NSW	Estimated annual cost of repairing damage by LGA - $16.5m
- Sinden *et al* 1986	
Conservation	
- Manilla Shire, NSW	Increase in land value per $1 invested in soil conservation works: $2.28
- King and Sinden 1988	

Table 3: Value of Environmental Resources: Some Australian Evidence (cont.)

Study	Estimated Value
Degradation	
- Manilla Shire, NSW	Value of annual loss in wheat yield - $33.3m
- Sinden *et al* 1986	
- Western Australia	Annual cost of lost production: $94m
- NSW	Average cost for repair of damage to road surfaces, removal of water borne sediment and other erosion damage per annum over 1980-81 to 1982-83: $7.5m
- Jerramungup District, WA	Annual cost of keeping sand off roads in 1982: $40K
- Darling Downs	Annual costs of siltation and erosion of roads per ha of cultivation: $1-2/ha Annual costs of sediment in streams and dams per ha of cultivation: $1-2/ha
- quoted in Upstill and Yapp 1987	
Wildlife	
Habitat preservation: Hairy nosed wombat - Sinden and Mackay 1979	$18000 raised in SA in 5 weeks in 1968 to preserve one of three remaining habitats; an additional $12000 was raised across Australia over the period 1968-1977.
Vegetation	
Preservation: eucalypt - Sinden 1986	Residents of northern NSW donated $90K to control dieback and restore eucalypt woodland
Tree retention: Loddon catchment - Greig and Devonshire 1981	Salinity damage costs to household water users of clearing 4179ha - $1829 per annum
Production: Murray River red gum forests - Greig 1979	Gross value of annual average production of Barmah and Gunbower forests - $7.3m
Trees on farms - Tisdell 1985	Net increase in crop/pasture yield from shelter belts - 12.5%
National Parks	
Tourism/recreation	
Cooloola - McDonald *et al* 1980	Tourism and recreation in 1982 estimated to add $8.3m output, $1.8m income and 159 jobs to Queensland economy
Kangaroo Island - Lothian 1985	Tourism in 1984 added $1.75m benefits (value-added) versus $0.5m costs; present value of net benefits estimated at $17m (20 year time horizon, 10% discount rate)
Warrumbungle - Ulph and Reynolds 1981	Recreation value estimated at $100 per visitor day (TCM)

Table 4: Selected Examples of Australian CVM Studies

Study	Unit of Value	Estimated Value
National Parks		
Nadgee Nature Reserve	1979 $	Canberra residents WTP for existence benefits:
- Bennett 1984	once only	mean: $27.08
	payment	median: $5.21
Great Barrier Reef	1986 $/	Annual aggregate WTP for reef management:
- Carter 1987	respondent	Visitors - $5.6m
		Non-visitors - $45.4m
		Annual aggregate WTP for research and control of Crown of Thorns starfish:
		Visitors - $1.3m
		Non-visitors $15.6m
Ben Boyd	1983 $/	Average reduction in WTP per visit due to bush fire:
Morton	visitor group	Ben Boyd - $0.61
Kosciusko		Morton - $0.29
- Bennett 1991		Kosciusko - $0.79
Kakadu Conservation Zone	1990 $/	Annual median WTP per Australian for ten years to preserve the Kakadu Conservation Zone:
- Imber *et al* 1991	respondent	Minor impact - $52.80
		Major impact - $123.80
		National aggregate median WTP per annum:
		Minor impact - $647m
		Major impact - $1518m
Forest Management		
Eucalypt dieback - New England	1981 $/	Annual average WTP to preserve eucalypts on farms by New England households: $6.84
- Sinden and Jones 1983	household	
- Ekanayake and Sinden 1985		Average WTP a once off payment by Armidale households for:
		Knowledge to prevent dieback - $12.40
		Woodland area for local school: - $4.80
		Health benefits from scientific discoveries - $3.70
Forest preservation - Fraser Island	1989 $/	Annual median WTP for Australian population to preserve forests from logging: $205.00
- Hundloe *et al* 1990	respondent	Annual Australian aggregate WTP to preserve forests from logging: $664.8m

Table 4: Selected Examples of Australian CVM Studies (cont.)

Study	Unit of Value	Estimated Value
Soil Conservation		
Soil erosion in wheat producing areas - Sinden 1986	1986 $/ resident	WTP for higher bread prices to reduce soil erosion in wheat producing areas: Sydney residents: 7-11 cents per loaf NSW country town residents: 10-12 cents per loaf
Pollution Control		
Air quality - Australian Environment Council 1982	1982 $/ resident	Annual average WTP for control of air pollution Sydney residents - $17.66 Adelaide residents - $10.44
Water Pricing		
Price elasticity of demand - Thomas and Syme 1988	1982 $/ household	Estimated price elasticity of demand for public water supply per household in Perth: - 0.18
Research		
Biological control - dung beetle - Johnston 1982	1979 $/ household	Annual average WTP for CSIRO research program by Canberra households: $13.40
Forestry management - Young and Carter 1990	1979 $/ respondent	Annual average WTP for forestry research which identifies potential conservation areas: SE Forests - $2.84 Australian forests - $17.19 Annual aggregate WTP for NSW/ACT - SE Forests - $2.1m

values. The other studies cited in Table 3 relate to use benefits of environmental resources.

For decision making, however, it is total value which is likely to be of primary interest, and for unpriced environmental resources, the application of CVM is necessary. Again the number of CVM studies conducted in Australia has been relatively small but growing. The results of a selection of these are presented in Table 4.

Many of these studies also have a strong policy orientation, ranging from forest management and soil conservation to pollution control, water pricing and scientific research. However, the magnitudes involved at the aggregate level appear substantial and reflect the extent of the significance placed on environmental resources by the community. For example, the study on the Great Barrier Reef (Carter 1987) reports an aggregate WTP of over $50m for reef management, whilst the WTP aggregates for Australia for the Kakadu Conservation Zone (Imber *et al* 1991) and for forest preservation on Fraser Island (Hundloe *et al* 1990) are substantially higher.

The estimates in Tables 3 and 4 appear to be broadly consistent with the corresponding US estimates in Tables 1 and 2, in terms of magnitudes and degree of variation. The considerable variation between case study estimates appears to reflect the influence of similar determinants, particularly uniqueness and significance of the resource. A woodland in northern New South Wales, or even the South Eastern Forests in New South Wales, would not on that basis be expected to generate existence benefits of the magnitude expected for the Great Barrier Reef or Kakadu or Fraser Island, and the estimates in Tables 3 and 4 tend to support that view. It is of course conceivable that if a unique biological organism or a significant aboriginal sacred site were to be identified in the woodland, then perceived values may change to a level commensurate with those of the national heritage resources.

Apart from the effect of inflation, two other factors noted by Carson (1991) which account for variation in value estimates are reporting variously by household, by visitor group or by individual, and reporting either mean or median values. Carson (1991) suggests erring on the conservative side and proposes that WTP estimates be reported by household at the median level.

The responses to the CVM results for the Kakadu Conservation Zone (Imber *et al* 1991; Resource Assessment Commission 1991) suggest that acceptability of the CVM approach in Australia is not widespread. The underlying issue concerns the accuracy of responses in a hypothetical market framework in which payment is not actually made. This in fact has been a focus of much of the empirical work based on the CVM approach. An important aspect of this work has been the testing for the significance of different sources of bias (examples include Boyle *et al* 1985; Boyle and Bishop 1988; and Sinden 1988). A comprehensive discussion of the various sources of bias and their significance is contained in Mitchell and Carson (1989). At this point in time, it seems that most of the bias problems have been addressed by refinements to questionnaire design, including adoption of a discrete choice format (see for example Imber *et al* 1991), to survey design and sampling, including the use of the referendum approach (see Hoehn and Randall 1987; Hundloe *et al* 1990) and to data presentation. It would however be premature if not foolhardy to conclude that the CVM represents nirvana in approaches to the valuation of environmental resources.

Issues which remain to be resolved and which lie at the heart of the debate in Australia include those of framing bias, or mental account bias as it is referred to by Hoevenagel (1990), and the related problem of aggregation, and the relationship between attitudes, as reflected in CVM responses, and actual behaviour.

The framing issue refers to the potential bias due to failure to make the respondent aware that the resource for which a response is being sought is not the only one which the respondent should mentally take into account, and as a result the income constraint may not properly apply. If for example, an individual in year 1 offers a WTP amount of $x for forest preservation, and the following year offers the same amount in response to a CVM questionnaire on national park preservation, and the sum $2x exceeds the respondent's discretionary income, does it necessarily follow that the responses are untruthful?

Carson (1991) states that it is not appropriate to treat public goods as independent and then to sum the WTP amounts to get a total WTP. Because of income and substitution effects, total actual WTP will be less than the sum of 'independent' WTP values. Presumably the same point can be made about aggregation of WTP amounts across studies at an aggregate level as well as at an individual level. But such a point then begs the question: do the individual study WTP's represent anything more than an expression of an attitude which has no real monetary significance?

One solution is to treat the WTP responses as being equivalent to the net benefit estimates of traditional benefit cost analyses which are typically conducted in isolation. It is then up to the decision maker to prioritise across the choice set. In the case of public goods, Carson (1991, p. 27) proposes that "It is the job of government to look at the possibilities for providing new public goods that most of the public wants and to provide these goods at the lowest cost possible so as to maximise the public's consumer surplus".

Nevertheless the suspicion may remain that WTP responses which are constrained by income only at a single point in time are not quite the same as revealed preference behaviour for market goods.To help address this problem, Wilks (1990, p. 20) in acknowledging that actual payment is dependent on ability to pay, proposes that respondents may require explicit instructions to think about their ability to pay and alternative uses they might have for their discretionary income. In a complementary approach to the problem, Young and Carter (1990) include in their CVM questionnaire on WTP for research on the South East Forests, a WTP question for all Australian forests. The attempt is thus made to specify a framing strategy which makes the respondent aware of the need, identified by Carson (1991, p. 43), to avoid the enhancement of the environmental resource being considered in isolation from other public problems.

The issue of the influence of information provision on WTP estimates from survey respondents is a concern of a number of commentators. On the one hand, the ability of respondents to comprehend complex issues in a questionnaire is perceived to be limited, whilst on the other the need for the respondent to understand and become familiar with the good being valued is recognised. But if the value of an environmental resource is a function of information about it (see Sharples *et al* 1986; Carson 1991), does that make estimates of its value less valid than say the value of a marketable resource the value of which may also change as more information about it becomes available? Information bias, like payment vehicle bias, is no longer widely regarded as a matter for concern, because the information presented forms an integral part of the CVM scenario and the WTP response is contingent on all the information presented in the questionnaire (Hoevenagel 1990, p. 20). Admittedly, the main concern of Sharples *et al* (1986) seems to be that the analyst can manipulate values by the amount of information provided, particularly where "the non-use component of total value is proportionately large." (p. 307). The risk of manipulation by the analyst however is also present in virtually all other forms of economic and scientific research and represents an ethical issue to be addressed by the profession concerned.

Clearly, there is scope for further work in the area of behavioural responses within a CVM framework. Whilst the development of the CVM approach has already benefited from socio-psychological input (see Wilks 1990, pp. 18-20), psychological research indicates that individuals in processing information to arrive at a decision under conditions of uncertainty may not conform to the behaviour of the expected utility theory, which underlies the CVM. On this basis Hoevenagel (1990) proposes that further research is required on how individuals' preferences are formed and implicit prices determined for environmental public goods.

Despite these qualifications, the evidence to date suggests that the CVM currently has the potential to generate order of magnitude estimates of the value of environmental resources which can validly assist decision makers to arrive at more informed policy decisions on resource use involving unpriced environmental resources. The response to Carson (1991, p. 29) that "The key question then is not whether there are potential contingent valuation biases - clearly the answer is yes - but rather whether these biases can be avoided to the extent that useful information about the public's willingness to pay ..... can be obtained using contingent valuation." is accordingly in the affirmative. Given however the scope for further research, particularly of a multidisciplinary nature, the assessment by Hoehn and Randall (1987) that CVM represents a "program in progress" seems eminently appropriate.

Overall the evidence reviewed in this paper supports the conclusion that the community places a significant positive value on many environmental resources. A second conclusion is that the methods developed by economists to measure these values can validly provide policy makers and resource managers with information which will assist them in their decision making.

5. Concluding Comments

It has been clearly demonstrated by the evidence presented that a wide range of unpriced environmental resources have significant economic value to the community. In considering options for resource management therefore, it is highly desirable that such values be taken account of in the decision making calculus. Arguments for preservation and conservation should be supported by value estimates rather than mere rhetoric.

Such an approach would be consistent with the proposal by Charlesworth (1987) to develop "hard-nosed" arguments rather than rely on what he termed "sloppy" arguments exemplified by "primeval is good" and "technology is bad".

Many of the issues being faced by resource managers in developing resource management strategies relate to natural resources which have no market price tag. In competing for funds in a tight budget situation, there would be considerable advantage in generating credible dollar values for unpriced environmental resources in the evaluation of options which address these issues. The studies initiated by the Resource Assessment Commission in this context represent an important step in this direction. The lessons learned can be viewed as the pay off to this initial investment and the time seems ripe to capitalise on that investment.

In terms of policy making, there is considerable scope for greater economic input. In addressing environmental resource management issues to date, much of the running has been made by scientists and engineers. The plea however is not for economic input to replace or supplant the professional input of other disciplines, but for evaluation which is the outcome of multidisciplinary effort. Pontification from professional cave dwellers or ivory tower recluses must give way to cooperative analysis and assessment. There are indeed some encouraging signs that this is already happening in a number of areas including the economic evaluation of scientific research, and the extension of economic modelling to include technology and the environment.

Some top down pressure from policy makers would also help. For example, if policy advisers are required to provide professional multidisciplinary assessments of value, then an effective stimulus to the provision of more rigorous and balanced advice would result.

Recent evidence suggests that such pressure may be forthcoming. The Primary Industries and Resources document which appeared as part of the 1988 May economic statement (Department of Primary Industries and Energy 1988, p. 18) contained the following: "With a view to maximising the net benefits to the community from the nation's resources - both economic and other benefits - the Government will seek to develop processes that provide for the equitable consideration of the costs and benefits in the process of reconciling development and conservation goals".

The work of the Industry Commission and the Resource Assessment Commission represent important steps in the implementation of that statement of intent. There are however many resource use issues which are not considered by these agencies. The wider application of benefit cost processes to these issues and the publication of the results may do much to help resolve the resource use conflicts which continue to dog decision making in Australia.

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