LINKING THE ECOLOGICAL AND ECONOMIC VALUES OF WETLANDS: 
A CASE STUDY OF THE WETLANDS OF MORETON BAY

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Thesis submitted in fulfilment of the requirements of the degree of
Doctor of Philosophy

June, 2002
ABSTRACT

This thesis examines the relationship between the underlying ecological values of wetlands and the economic values they produce. The importance of the roles that wetlands play is now well recognised yet losses continue at a global level. It is argued that one cause of wetland loss is a lack of awareness of the values of these systems due to inadequate information of the nexus between ecological functions and economic values. For example, the off-site, indirect benefits provided by wetlands have largely been ignored. This has led to an undervaluation of these ecosystems.

The integration of ecological and economic values requires incorporating differing types of information and systems of value derived from differing disciplines with differing paradigms. To understand the differing disciplinary perspectives the thesis explores the ecological functions of wetlands and the economic goods and services that they provide. The functions and benefits of wetlands are linked at the ecological-economic interface.

A consensus on the definition of ecological value could not be discerned within the discipline of ecology. Thus, a definition and index of ecological value is developed to demonstrate the attributes of coastal and wetland systems that provide for instrumental human benefits. These attributes include productivity, the ability to provide habitats for dependent species and the diversity of species and organisation they support. However, ecological information is not presently available to operationalise the index. The ability of economic techniques to capture this ecological value is then investigated.

Three approaches for assessing non-market values (direct linkage models, revealed preference and stated preference models) are reviewed with respect to their ability to capture ecological value. An alternative biophysical approach, namely energy analysis, is also considered. The review suggests that it may be possible to measure ecological value using the contingent valuation method. The role of information in preference formation
and willingness to pay bids is then investigated along with a number of other issues that need to be resolved before using the contingent valuation method.

The wetlands of the case study area, Moreton Bay, Australia exhibit both ecological and economic values. The wetlands contribute approximately one-third of primary productivity in the Bay, provide habitat for a wide range of dependent species (including internationally recognised migratory wader birds) and have a diverse fauna with a relatively large number of endemic species. Economic values of the wetlands include both direct and indirect use values (for example, fishing, recreation, water quality improvements and storm buffering) and non-use values. Non-use values include the value in preserving the environment for future generations (bequest value) and the existence of vulnerable animals such as turtles and dugongs, which one may never expect to see. If consumers are willing to pay to preserve these animals, this is also a valid economic value.

The economic technique of contingent valuation is tested to determine if it is possible to capture ecological value by providing respondents, selected by random sample, to a survey with the relevant information. A case study is undertaken in Moreton Bay to determine respondents’ willingness to pay to improve water quality and hence protect the wetlands. To test the effects of differing types information, four different versions of the survey were sent to four groups of 500 respondents. Case A provided no extra information so it could be used as a control. Case B included information about the ecological values of the wetlands of Moreton Bay. Case C provided information about the economic use values of the wetlands in the Bay including direct and indirect use. Case D provided information about the non-use values of endangered species resident in the Bay that are dependent on the wetlands. The results indicate that the provision of different types of information influences willingness to pay. However, willingness to pay when provided with ecological information is not significantly different from willingness to pay when provided with other information.
As it was not possible from the research undertaken to state that the contingent valuation method can capture ecological value, an alternative approach is proposed to link ecological and economic values. It is argued that ecologists and economists need to develop common aims and scales of assessment. Further, communication between the two disciplines can be enhanced through the use of agreed indicator terms. Through an iterative approach it should then be possible to understand the linkages between changes in indicators of ecosystem values and indicators of economic value.
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ACKNOWLEDGEMENTS

I would like to thank Dr John Tisdell, my principal supervisor, for his guidance, support and patience during the preparation of this thesis. Special thanks also go to Dr Rod Connolly who provided advice on the ecological content and direction of the thesis. The Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management supported some of this research with a scholarship and research funds to conduct the planned survey component. I am grateful for this support and the advice of a supervisor from the Coastal CRC, Dr Paul Lawrence, who provided comments on a draft of this thesis.

Many of the maps used in the thesis were provided by the Queensland Parks and Wildlife Service. My special thanks go to Nicola Udy who produced the maps and provided encouragement. Thanks also go to “The Printing Office” who provided assistance in the production of the survey.

I owe a considerable debt to my close friends and family who have supported me over the time I was involved in this study. In particular, I thank my children, Max, Ella and Nick who may have doubted the wisdom of my actions but gave up their childhood holidays so I could continue my research. This thesis would not have been possible without the education provided to me by my parents, Brian and Sheliah Clouston. Thus, I would like to dedicate this thesis to the memory of my mother.
STATEMENT OF ORIGINALITY
OF WORK

This work has not been previously submitted for a degree or diploma in any university. To the best of my knowledge and belief, the thesis contains no material previously published or written by another person except where due reference is made in the thesis itself.

Signed

Date

Elizabeth M Clouston
In the twenty-first century, many natural resource managers attempt to balance considerations of the biophysical system with those of the social economic system. In the last decade, the need to integrate the values of systems and stakeholders has led to the development of integrated management techniques. For example, management in the coastal zone is conducted through a process of integrated coastal zone management and in the adjacent catchment, integrated catchment management is commonplace. However, methods for achieving this goal are still being developed. In order to integrate biophysical and socioeconomic systems the interactions between the systems must be understood. This in turn requires incorporating differing types of information and systems of value derived from differing disciplines with differing paradigms to produce the desired integrated outcomes.

Costanza (1993) argues that the ecological and economic systems are considered to be subsystems of one very large and complex system, the biosphere. Each system exhibits the characteristics of complex systems, that is, strong (usually non-linear) interactions between parts, complex feedback loops, differing time and space lags, discontinuities, thresholds and limits which lead to the inability to arrive at a large scale result by simply adding the smaller parts. Ecological and economic systems are not independent but dynamically interdependent. Ecological processes are impacted upon by economic activity and economic activities are provided for and constrained by the natural environment. It is this complexity that makes the study of the linked ecological-economic system very difficult. Ecological economics attempts to develop new ways of linking the two systems while employing the underlying paradigms of the differing disciplines of study. Thus, Norgaard (1989) points out the methodology used to investigate the interdependencies and co-evolution of the economic and ecological systems is one of conceptual pluralism. Therefore, this study draws upon the disciplines of both ecology and economics in order to link the two sets of value.
Although the dynamic interaction of the two systems is now recognised, the paradigms and values that have developed within the disciplines of ecology and economics have been derived in isolation for much of the 20th century. For example, traditionally ecology has been the study of nature that does not include humans, and economics has been the study of human interactions isolated from nature (Costanza, 1996). However, as the scale of impact of humans on ecosystems has resulted in global consequences (such as global warming) the ecological and economic systems can no longer be seen in isolation. The interactions between the two systems are, however, so complex and poorly understood that it is not possible to disentangle them all within one research project.

The link considered in this study is between the values that can be assigned to the respective systems using wetlands as an example of an ecosystem type. The concept is considered important because every choice made about the use of an ecosystem, whether it is destruction or preservation, implies that a value has been placed upon it. However, it is argued the economic value assigned to ecosystems may not fully incorporate the underlying ecological values. This thesis contributes to overcoming this perceived deficiency by exploring wetland values from both an ecological and economic perspective. As the terms value and wetland are crucial to understanding this link, the concept of value and a definition of a wetland as used throughout this study are provided before proceeding with an outline of the study’s aims and structure.

1.1 PERSPECTIVES OF VALUES

Value has many different meanings. In ethical philosophy judgements of value are based on the goodness or badness, desirability or undesirability of certain objects, ends, experiences or states of affairs (Frankena, 1973). Frankena (1973) suggests that when the term is used as a verb (‘to value’) it denotes the act or attitude of valuing or valuation. The act of valuing in philosophy thus encompasses utilitarian, aesthetic and moral assessments (Barry and Oelschlaeger, 1996). Moral actions are chosen not for their own sake but as an instrument aimed at the achievement of a higher good.
However, the term value for the average person is an expression of the importance or desirability of something (Bingham et al., 1995). Brown (1984) asserts that there are two types of values ‘assigned’ values and ‘held’ values. Held values are those values of enduring beliefs about what is preferable and generally desirable (Brown, 1984). Assigned values are the relative worth of objects or actions and are often expressed in some quantitative unit of value, for example monetary worth. These values are derived from an individual’s ‘held’ values (Brown, 1984). The assessment of ecological and economic values of an ecosystem is an act of assigning values based on the concept of the relative worth of the ecosystem.

Perspectives of value differ in ecology and economics depending on the underlying paradigm of the discipline (Norton, 1998). Bingham et al., (1995) note that an ecologist’s perspective of value may be related to important attributes or functions, while an economist approaches the concept from what people are prepared to pay to maintain that system or some of its attributes. Defining value in ecology is a relatively new problem (Page, 1992). Page (1992, 118) states this idea as “the biologists have a connected view of ecological systems, but not a value theory to go with it”. Norton (1998) argues that ecology often studies systems or species at the local level and it is not always possible to generalise from these findings to produce an abstract ecological theory. That is, ecology has a broad methodological base of competing theories without a hierarchy of theoretical axioms, laws and ‘truths’ (Shogren and Nowell, 1992). Given the complexity of ecosystems, Shogren and Nowell (1992) argue that ecologists are still far from establishing “universal” laws or even if they exist. According to Perrings (1995) there is no systematic discussion within ecology of the relative importance of ecosystems and the components of ecosystem.

Shogren and Nowell (1992) point out that economics, on the other hand, has a well-developed general abstract theory, which is then applied to the specific. For example, a dominant paradigm in economics is the neo-classical theory of optimization. This theory has a well-defined theoretical structure that is inherently correct which in turn has limited the need for observation through experimentation. Within this theory, Goulder and Kennedy (1997) note that the value of an ecosystem to society is the value it confers to persons. Thus, economists value ecological
systems in terms of the value of the services that are provided to humans by ecosystems (Daily, 1997). The economic value of the system can be defined as the ‘willingness to pay’ or ‘compensation demanded’ for changes to the system in question. The relative worth of the system is then expressed in monetary units. From this anthropocentric viewpoint, value is determined by the instrumental preferences that are held by people.

Some philosophers argue that natural resources have an intrinsic worth beyond the instrumental values provided to humans (Barry and Oelschlaeger, 1996). However, the concept of inherent value describes a very well defined preference based value. Perrings (1995) for example, states that the value placed on non-human species is derived from ‘rights’ assigned by humans to those species. Thus, all values are assigned from an anthropocentric point of view. Sagoff (1992) argues that the difference in valuing the health of nature on intrinsic or instrumental grounds is distinguishing between what may be healthy for nature and what may be healthy for humanity.

As there is no general ecological theory on which to base values for nature, the approach taken throughout this thesis is to consider value in terms of what may be healthy for humanity. That is, values are assigned to ecosystems based on the economic perspective of instrumental human values. From this perspective, the assessment of ecological value is determined by considering the ecosystem attributes that provide instrumental human benefits. It is argued that as ecosystems ultimately provide life support and essential processes for humanity an instrumental view is sufficient for determining ecological value. This is not to say that the entire range of values including social, intrinsic, moral, aesthetic and spiritual values is not important but they are outside of the scope of this study.

The differences in perspectives of value provide an example of the difficulties of defining and linking the values of ecological and economic systems. It might be thought that the concept of a wetland is easier to define. However, there are over 50 different definitions of wetlands (Dugan, 1993). Therefore, it is necessary to clearly define how this term is used throughout the thesis.
1.2 DEFINITION OF WETLANDS

The group of ecosystems described collectively as ‘wetlands’ are not uniform but heterogeneous (Dugan, 1990). These systems are found throughout the world under many different names such as swamps, bogs, marshes, mires and fens (Mitsch and Gosselink, 1993). It is estimated that wetland ecosystems account for about 6–8.6% of the land surface of the world or 7 to 8.5 million km$^2$ (Mitsch, 1998, 20). Wetlands are ubiquitous as they are found from the tropics to the tundra on every continent except Antarctica (Maltby, 1986). It is estimated that up to 60% of the estimated total wetland area in the world (4.8 million km$^2$) is found in tropical and subtropical regions (Mitsch, 1998, 20).

Definitions of wetlands usually include three main components: the presence of water, either at the surface or within the root zone, unique soil conditions that differ from adjacent uplands, vegetation that is adapted to wet conditions and conversely is characterized by an absence of flooding-intolerant vegetation (Mitsch and Gosselink, 1993). The definition used throughout this thesis is taken from the Ramsar Convention on Wetlands of International Importance, Especially as Waterfowl Habitat$^1$, which provides the broadest, internationally recognised definition. This convention, adopted in 1971 was the first international treaty related to an ecosystem type, which may signify the importance attached to these systems and the concerns related to their degradation. The definition of wetlands under the convention is:

“areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres” (Environment Australia, 1997, 29).

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$^1$ The Ramsar Convention on Wetlands of International Importance, Especially as Waterfowl Habitat will hereafter be referred to as the Ramsar Convention.
In addition, Article 2.1 of the convention provides that wetlands “may incorporate riparian and coastal zones adjacent to the wetlands, and islands or bodies of marine water deeper than six metres at low tide lying within the wetlands” (Environment Australia, 1997, 29).

This definition encompasses seven landscape units in which wetlands are an important component and includes estuaries, open coasts, floodplains, freshwater marshes, lakes, peatlands and swamp forests. Each of these types of wetlands provides a range of ecological and economic services. No single wetland will provide all the services ascribed to wetlands but all wetlands yield multiple benefits depending on the interaction of the biological, chemical and physical characteristics of the site (Dugan, 1990).

Primary emphasis is given to coastal wetlands in this thesis. Currently 86% of the population of Australia resides in the coastal zone which occupies 17% of the land mass (Resource Assessment Commission, 1993, 8). This zone which is at the interface between terrestrial and marine systems is under increasing pressure due to population growth and contaminant accumulation.

### 1.2.1 Characteristics of coastal wetlands

Coastal wetlands include estuaries, mangroves, subtidal aquatic beds (including kelp beds and seagrasses) and intertidal flats and marshes (Australian Nature Conservation Agency, (ANCA), 1996). Under the Ramsar Convention definition they may also include shallow coral reefs and rocky shores. However, as Keddy (2000) notes, due to the difference in the ecological characteristics between reefs and rocky shores and other components of the wetland landscape these systems require a separate treatment and thus they are not discussed further. McComb and Lake (1990) state that estuaries support a range of wetland habitats including seagrasses, intertidal sand and mud flats, salt marshes and mangroves. Estuaries are considered to be areas of high productivity and are among the most naturally fertile in the world (McComb and Lake, 1990). This high productivity of estuaries supports extensive
food webs which permit the rapid growth of juvenile fish which use estuaries as nursery areas (Dugan, 1990). Other coastal wetlands include those on open coasts that support a similar diversity of wetlands as estuaries.

The major floral components of coastal wetlands in the study area are mangroves and seagrasses. According to Lugo and Snedaker (1974) the term mangrove refers to two different meanings. The first describes a group of about 80 species of halophytic (salt-tolerant) plants belonging to about twelve genera in eight different families (Lugo and Snedaker, 1974, 43). The second concept refers to the complex of plant communities fringing sheltered tropical and subtropical sheltered coastlines shores referred to collectively as mangrove forests. Maltby (1986) estimates that there are 14 million hectares of mangrove wetlands worldwide. Mangroves are thus a diverse community of plants whose composition varies according to conditions of salinity, tidal system and substrate (Dugan, 1993). However, mangroves share a variety of morphological, physiological and reproductive adaptations that allows them to live in an unstable and difficult environment (Lugo and Snedaker, 1974). They are regarded as being amongst the most important of wetland habitat types as they constitute an important economic resource for coastal people (Hamilton and Snedaker, 1984; Maltby, 1986).

Another dominant component of coastal wetlands is seagrasses. Seagrasses are benthic dwelling marine angiosperms (flowering plants) which occur worldwide in sheltered shallow coastal waters in both tropical and temperate zones (Hyland et al., 1989). Seagrass meadows are of considerable importance in coastal marine systems. Pollard (1984) points out that seagrass habitats act as nursery areas for both the juvenile and sub-adult stages of fishes and crustaceans. The role of seagrasses as fish nurseries is related to their primary productivity, the shelter they provide for small fish and the provision of abundant food sources particularly in the form of small crustaceans which in turn depend of the seagrass detritus for food (Pollard, 1984). Poiner and Peterken (1995) argue that seagrasses reduce the wave energy reaching the shore by increasing bottom friction and trap and bind loose sediments and thus help to stabilise the substrate. In Australia, seagrasses are the principal food of dugongs (*Dugong dugon*) and green turtles (*Chelonia midas*) (Poiner and Peterken, 1995).
1.2.2 Estimates of Wetland Losses

Despite the recognition of the important roles that wetlands play, losses continue at a global level (Turner et al., 2000). This has led to wetlands being described as among the most threatened of ecosystems (Williams, 1990a; Turner, 1991; Mitsch and Gosselink, 2000). Barbier et al., (1997) argue that one main reason for the losses is the lack of awareness of the values of these systems and Dixon and Lal (1997) see that this may result in an unrecognised social cost. The estimates of global loss of wetlands are reviewed to emphasise the scale of the problem. Causes and reasons for wetland losses are given to illustrate the need for improved information to bridge the nexus between the functions and values of these systems.

Dugan (1993, 45) states that it is estimated that 50% of wetlands have been lost worldwide. In the United States approximately 54% (870 000 sq km) of wetlands which existed prior to European contact have been lost, with some states having an even higher proportion (Dugan, 1993, 45). Conversion to agriculture accounts for 87% of recent loss (1950’s-1970’s) in the US (Maltby, 1986, 10). Further, Dugan (1993, 45) states that 40% of the coastal wetlands of Brittany have been lost since 1960 and two-thirds of the remainder are seriously affected by drainage. In southwest France 80% of the marshes of the Landes have been drained. In Portugal, 70% of the western Algarve wetlands have been converted to agricultural and industrial development (Dugan, 1993, 45). It is estimated that in New Zealand over 90% of wetlands have been destroyed since European settlement, and drainage continues. In the North Island of New Zealand 14% of the remaining wetlands were drained between 1979 and 1983 (Dugan, 1993, 46). In the Philippines, some 3000 square kilometres (67% of the mangroves) were lost in 60 years (1920-1980) with 17 000 ha converted to culture ponds for shrimp and milkfish (Dugan, 1993, 46). In Brazil, most estuarine wetlands have been degraded as a result of pollution.

According to Turner (1991) and Gosselink and Maltby (1990) the precise loss of wetlands for other nations is not generally available. However, it is known that mangrove swamps in Asia and Africa are rapidly disappearing due to land reclamation, fishpond construction, mining and waste disposal (Turner, 1991; Gosselink and Maltby, 1990). A similar situation exists in Australia where it is
estimated that 50% of wetlands have been converted to other uses since European settlement just over 200 years ago (Environment Australia, 1997, 7). This loss has been even greater in some regions. For example, Environment Australia (1997, 7) estimate that 70% of the wetlands in the Swan Coastal Plain of Western Australia have been filled or drained and 89% of the wetlands in south-east South Australia have been destroyed.

1.2.3 Causes and Reasons for Wetland Loss

Wetlands are lost and gained as a result of natural processes of formation, conversions and degradations (Gosselink and Maltby, 1990). Wetland losses and alterations are also caused by agricultural intensification, urbanisation, industrialisation, port expansion, pollution, the disruption of natural processes, dam construction, regional water transfers and other forms of intervention in the ecological and hydrological system (Pinder and Witherick, 1990; Dixon and Lal, 1997; Turner et al., 2000). According to Turner (1991) and Barbier et al., (1997) these losses are a result of poor understanding of the full economic values of wetlands leading to an undervaluation of the resource. Thus, the need to identify and understand the links between the ecology and economic systems is highlighted in the case of wetlands. The public good nature of wetlands and externalities arising outside the wetland from human processes has also contributed to conversion and degradation of these systems (Dixon and Lal, 1997).

For centuries the drainage of wetlands was encouraged in the interests of societal health and progress (Maltby, 1986). In some cases, wetland reclamation has been needed and desirable as these conversions have been in society’s best interests (Pinder and Witherick, 1990, Turner et al., 2000). However, until the 1960s it was assumed, in general, that a drained wetland was always better than an undrained wetland (Williams, 1990b). The period between the mid 1960s and the mid 1980s saw an increased recognition of wetland values concerning water quality improvements and storm mitigation (Williams, 1990b). However, Turner (1991) points out the policies to convert wetlands were still actively encouraged in the industrialised and developing world in this period. For example, in the European Economic Community prices for agricultural products were maintained by subsides
and guarantees that encouraged wetland conversion (Williams, 1990b, Turner et al., 2000). In the developing world it was estimated that 21% of EEC development assistance between 1976 and 1986 was spent on hydroagricultural projects which affected wetlands in one way or another (Dugan, 1990). It has been since the 1980s that attempts have been made to assess and order the relative social merits and values of the differing and competing functions of wetlands (Williams, 1990b).

Wetlands are linked to both land and water which places them at risk from activities occurring within the catchment as these may lead to wetland degradation downstream (Turner, 1991). Pollution damage to wetlands from agricultural chemicals, increased sedimentation from soil erosion and point source pollution from sewerage treatment plants have all inflicted serious damage on estuarine wetlands (Turner, 1991). Examples include Chesapeake Bay, in the USA, Norfolk Broadlands in UK, the Wadden Sea in the Netherlands, West Germany and Denmark and the Brazilian Pantanal (Dugan, 1990; Turner, 1991). Similarly, Moreton Bay has also suffered from similar externalities due to land clearing, and point and non-point sources of pollution (Dennison and Abal, 1999).

The undervaluing of wetland in economics is in part a result of the public good nature of wetlands. Where there is open access to wetlands it may lead to depletion of the resource (Turner et al., 2000). Also, as the services derived from a wetland such as water purification and storm surge protection are considered ‘free goods’ they have often been ignored in past economic calculations. This has increased the tendency for the conversion of wetlands for development (Dugan, 1990). Another difficulty related to the public benefits provided by wetlands is that they may be privately owned. As wetland owners are not able to appropriate the off-site benefits of the wetland, it limits the incentive to maintain it. That is, the private benefits to the owner do not reflect the full benefits to society (Turner et al., 2000).

As many of services of wetlands, in particular the ecosystem and global level services, are not traded in markets, defining an economic value for these services has been difficult. As a result they have been ignored (Dixon and Lal, 1997). The absence of markets also extends to population level services where they provide for subsistence in many tropical wetlands. Failing to take the values of subsistence
fishing, hunting and fuelwood extraction into account may be a major factor behind policy decisions that have led to the over-exploitation or excessive degradation of tropical wetland systems (Barbier *et al*., 1994).

This summary of wetland losses and causes demonstrate that an overriding reason for the loss and degradation of wetlands is ignorance by decision-makers as to the full ecological and economic values of these systems. Dugan (1990) argues that increasing the awareness of the values of wetlands and designing economic incentives to conserve them should reduce the rate of wetland loss. Barbier *et al*., (1994) also acknowledge that the essential ecological services and resources of wetlands are significant. However efforts to assess these values and incorporate them into decision making have been too few. “The undervaluing of wetland resources and functions is a major reason why wetland systems are misallocated – often to conversion or exploitation activities yielding immediate commercial gains and revenues” (Barbier *et al*., 1997, 20). Further, Barbier *et al*., (1997), argue that it is an area where ecologists and economists should work together to improve analysis. That is, a framework that incorporates both an economic and ecological perspective is essential if the true values of wetlands are to be understood and used as a basis for management decisions. This thesis attempts to address this deficiency.

1.3 RESEARCH AIMS AND ORGANISATION OF STUDY

To account fully for all the values of wetlands and hence reduce losses resulting from undervaluation requires an estimation of both the ecological and economic values of these systems. Thus, the main aim of this thesis is to establish the linkages between the ecological values and economic values of environmental goods, using wetlands as an example. One means of linking the two sets of values is to incorporate the underlying ecological values of the system within the economic valuation. This would entail finding economic methods that could capture both the underlying ecological values and the interdependent and simultaneously produced goods and services of the wetlands. Thus, the overarching question is: Can economic non-market valuation techniques capture ecological values?
In order to test the hypothesis it is necessary to first investigate the relationship and linkages between the ecological values and economic values. This leads to the first of the research questions which is:

1. How have the ecological values of environmental goods been linked to the values derived from expressed human preferences?

Wetlands are chosen as a subsystem of the ecological system to investigate the relationship between ecological and economic value. As noted, wetlands are multifunctional, providing a range of goods and services for humans but they continue to be lost and degraded throughout the world. One reason for this loss is lack of information about all the benefits that wetlands provide. Valuing only those on-site resources provided by wetlands has led to an underestimation of their full value to society. Therefore, information is required that fully describes the interaction of the structures, processes and functions of wetlands and a framework for connecting these functions to economic outputs.

Chapter 2 begins to address the thesis question and presents a framework to investigate the relationship between ecological and economic systems and the interface between the two. The essential workings of the ecological components of wetlands are outlined. This is followed by a review of studies that have attempted to assess the linkages between the ecological and economic sectors of wetlands and outlines further possible approaches for these linkages to be established. A direct link between each ecological function and each good or service is unlikely to be found as outputs are produced interdependently and simultaneously by the system. To examine the economic sector of the framework, the chapter then introduces the concept of total economic value, the economic goods and services provided by wetlands and studies that have previously been undertaken to value wetlands.

From the review of the literature in Chapter 2, it is argued that any ecosystem valuation will require information from both ecology and economics. Economists seek information on the important underlying values of an ecosystem. Similarly, ecologists need to be aware of the ecosystem characteristics which need to be conserved to provide for human use. Thus, in order to investigate further the
relationship between ecological and economic value it is necessary first to define ecological value and investigate the available information which can be used to measure ecological value. Thus, the next two questions addressed in this thesis are:

2. What is the definition and meaning of ecological value?
3. How can ecological value be measured from an ecological perspective?

Chapter 3 addresses the second and third thesis questions by assessing the attributes of ecosystems that provide instrumental human benefits. The approach taken is to define the attributes that provide ecological value so that ecosystems can be ranked in terms of these attributes. However, as noted there is no consensus on what the attributes of ecological value are, largely because the concept of value has been avoided in ecology. The concept of value has, however, been made explicit in the fields of biological and ecological conservation. The attributes that are used to identify marine and wetlands areas for protection have been used as a basis for selecting the attributes of ecological value. This has entailed a somewhat deductive approach where those attributes that are related solely to conservation aims (for example, uniqueness) have been excluded from the list to arrive at those attributes related solely to ecological value. The chapter proceeds to derive an index of ecological value based on the selected attributes of productivity, dependency and biological diversity and organisation. The definition and index of ecological value provide a valuable advance in the linking of ecological and economic systems. However, again the importance of information became apparent as it was realised that the relationship between the attributes has not been fully examined within ecology.

The alternative to valuing wetlands from an ecological perspective is to value wetlands from an economic perspective using ecological information. The question of interest which is directly related to the hypothesis to be tested then becomes:

4. Are non-market economic valuation techniques capable of capturing ecological value?
Chapter 4 addresses this fourth thesis question from a theoretical perspective. It is considered that the values of the ecosystem must precede the secondary economic values produced. That is, ecological value is separate from and occurs prior to the components of total economic value. Nevertheless, ecological value, and the services produced by ecosystems, include both use and non-use values. Thus non-market techniques will be required to capture it. In Chapter 4, three approaches to non-market valuation are assessed with reference to four criteria, which have been specifically derived to estimate the ability of the techniques to capture ecological value. The criteria include the ability to measure the change in ecological value, the number of attributes of ecological value that can be assessed, links between the good being examined and ecological value and the components of total economic value that can be assessed.

From the analysis of techniques undertaken in Chapter 4 it was determined that the contingent valuation technique may be able to capture ecological value if respondents to a survey were provided with information about these values. This then led to the issue of how does the provision of information in contingent valuation survey alter willingness to pay? Chapter 5 reviews the literature related to information impacts in contingent valuation surveys with particular reference to studies that have attempted to measure ecological value. The chapter also outlines the major issues that need to be considered in the design and implementation of contingent valuation surveys.

Once again the issue of adequate information about ecological values and the ability of the public to include ecological value in their preference formation became a central concern. It was considered that the public would need to have prior knowledge of the environmental resource being valued before being able to derive a monetary value for the ecological values which provide instrumental human benefits. Further, a review of the literature raised the issue of how much and of which type of information should be provided to respondents of a survey used to assess the ecological and economic values of an environmental resource. This then raised the further questions of:

5. Can the contingent valuation technique capture ecological value?
6. What is the effect of providing differing types of information about an environmental resource on willingness to pay using the contingent valuation technique?

The wetlands of Moreton Bay, Australia were selected as a case study area to test empirically if the contingent valuation method could capture ecological value. Chapter 6 is devoted to presenting the currently available information about the ecological and economic values of the wetlands of Moreton Bay. The region was chosen as the wetlands present exhibit a range of ecological values, which in turn provide a number of economic values. Thus, an understanding of the values of the area may provide information on the manner in which the two sets of value are linked. In recognition of the importance of Moreton Bay for migratory birds it has been declared as a site of international importance under the Ramsar convention (ANCA, 1996). Moreton Bay is also listed as a wetland of national importance and has been declared a State Marine Park (ANCA, 1996). However, there have been no studies undertaken to determine the full nature of these values. In fact, there have not been any comprehensive studies undertaken of the economic values of subtropical coastal wetlands in Australia. Thus, this thesis makes an important contribution in understanding these values as they are under particular threat due to increased development and pollution arising from the rapidly expanding population in the region.

From the information provided in Chapter 6 a contingent valuation survey was designed. Chapter 7 outlines the methodology used to design and implement the survey. The study was designed to answer the fifth and sixth thesis questions. To assess if the method can capture ecological value information was provided to a sub-sample about these values. Three other sub-samples were given either no information, information about use values or information about the non-use value of endangered species dependent on the wetlands. This allowed a comparison to be made across the samples in terms of the effect of providing differing types of information about an environmental resource on willingness to pay. Chapter 8 presents the results of the pilot study. Chapter 9 analyses the results of the main study with reference to these questions.
The results presented in Chapter 9 indicate that it is not possible to accept definitely the hypothesis that economic techniques can capture ecological value. However, an approach for linking ecological and economic values using an integrated research program is presented in Chapter 10. This approach requires the cooperation of both ecologists and economists so that there is an agreed purpose of study and a mutually determined spatial and temporal scale of assessment. To overcome the barriers of communication across disciplines it is suggested indicators should be developed. The ecological health monitoring program currently being undertaken in Moreton Bay is used as an example of how the approach may be applied (Environmental Health Monitoring Program, 2001a). Conclusions are then drawn in Chapter 11 with respect to all the thesis questions posed, the approach adopted in this study and further research needs.

1.4 CONCLUSION

This chapter has outlined the approach taken in this study to link the ecological and economic values of wetlands. The concept of value used throughout the study is an assigned one. As ecological value has not been well defined within ecology the perspective of value is taken from economics so that ecological and economic values are considered in terms of the instrumental human benefits they provide. The definition of wetlands used throughout the thesis is taken from the Ramsar Convention with particular emphasis placed on coastal wetlands. It is argued that an underestimation of the value of wetlands has been one of the reasons for the global loss of these multifunctional ecosystems. One method for overcoming this deficiency may be to include the ecological value of the wetlands within an economic assessment. Therefore, the question to be tested is that ecological value can be captured using economic techniques. This question in turns leads to a number of other questions that are addressed sequentially in the thesis.
The purpose of this chapter is to explore the interface between the constructs of ecology and economic in relation to the value of ecosystems. A conceptual framework is proposed to outline the relationship between the ecological system, the economic system and the interface between the two. To illustrate the interface between ecological and economic values, wetlands are used as an example of a multifunctional ecosystem which posses both ecological and ecological values. Each system is then reviewed in turn to demonstrate the concepts and links between wetland functioning and wetland values. The chapter concludes by considering two possible approaches for linking ecological and economic values.

2.1 A FRAMEWORK TO INVESTIGATE THE ECOLOGICAL-ECONOMIC INTERFACE

Given that humans are dependent on ecosystems then Boyden and Dovers (1997) argue that consideration of the value of those ecosystems must encompass both ecological and economic systems. A framework which encapsulates both these systems must include the ecosystem values and the value of the economic goods and services they provide. The ecological value of the system will result from the interaction of the structures, process and functions, which provide instrumental goods and services for human benefit. The structure of the ecosystem is considered to include the physical abiotic and biotic components of the system. The processes refer to the dynamics of transformation of matter or energy (Turner et al., 2000). Ecosystem functions then result from the interaction of the processes and structures. The functioning of the system in turn provides services that provide benefits for humans.

If humans have a preference for these benefits and are willing to forego some other resource in order to obtain an extra amount of them, then they will have an economic value (Turner et al., 2000). The value of the goods and services produced by the ecosystem is determined within the economic system. A total economic value
framework gives some guidance of the benefits that may be produced in a natural ecosystem. Within total economic value, there are a number of different components that can be identified. These include use values (direct and indirect) and non-use values (Pearce, 1993).

The step between ecological functions and economic valuation is the essential link between the two systems. The functions themselves do not provide economic value rather it is the use of the services provided by the functions. Therefore, to understand the link between the two systems requires an understanding of the underlying structures, processes and functions of the system, the resultant goods and services and the human perception of the value of those resources relative to the value other resources. However, the interface between these two systems has not been fully explored and as a result, information on the links between the systems is not always available when making resource allocation decisions (Turner et al., 1995). It is this sector which links the ecological value of the wetland to the economic value which is the focus of this thesis.

The difficulty in determining the optimal uses of resources is particularly vexing when the ecosystem possesses multiple simultaneously produced and interdependent ecological and economic values. Wetlands are an example of ecosystems that are multifunctional in this respect (Barbier, 2000). Figure 2.1 provides a framework for an ecological–economic analysis of wetlands. The purpose of this framework is to highlight and clarify the link between wetland functioning, wetland uses and wetland values. This conceptual framework is retained throughout the study and expanded upon where appropriate.
In Figure 2.1 the ecology sector of the wetland includes the interaction of the interdependent structures, processes and wetland functions. The functions of the wetlands can be considered in terms of hydrological, biogeochemical and ecological components, which are interdependent. The combination of the structures, processes and functions will produce a specific set of attributes in relation to that system. If the
attributes of the wetland ecosystems can be defined in terms of instrumental human benefits, then it may be possible to assess the ecological value of the system. Alternatively, if a demand exists for the goods and services provided by the system the value of the system may be determined through economic analysis (Turner et al., 2000).

The ecological-economic interface illustrates this transition from wetland function to wetland uses. Examples of goods and services produced by wetlands, which are considered to provide benefits, are given in this sector. It is through this sector that information about the structures, processes and functions of the system can be linked to instrumental human benefits they provide.

The lower section of Figure 2.1 illustrates the ways in which the goods and services of the wetland may be considered in terms their contribution to total economic value. The direct and indirect uses of the system include food and fuel resources, water quality improvement and biogeochemical cycling (Dugan, 1990; Mitsch and Gosselink, 1993). A persistent problem for the economic valuation of these goods and services is that they may neither be traded in traditional markets nor perceived to be of value to the individual user. Non-use values by their nature are non-marketed and non-market economic methods have been developed to measure these values.

The emphasis in Figure 2.1 is on the ecological-economic interface. However, it must be acknowledged that these systems interact with the social system so that the total environment (or biosphere) and its value will encompass the three differing systems. Turner et al., (1995) argue that even if all the goods and benefits that an ecosystem can produce could be measured using economic techniques, the valuation exercise would still fail to capture the underlying ecosystem values. These ecosystem values are those attributes that are crucial to the viability of the ecosystem but are not directly useful to people (Bingham et al., 1995). Bingham et al., (1995) argue these attributes are important as they support the processes and structures which provide functions which do support instrumental human benefits. In the view of Gren et al., (1994) and Turner et al., (1995) the total value of the system will include the secondary value of the economic output and the primary or ‘glue’ values of the ecosystem. That is, secondary economic values are dependent on the...
existence, operation and maintenance of the ecosystem life support system (Folke, 1991; Odum, 1989). Turner (1999) argues that this primary value is not amenable to quantification. However, if these values are to be taken into account in resource allocation they should be defined and measured where possible. The term ecological value is used to encapsulate the concept of ‘primary value’ in this study.

Turner et al., (1995) also argue that the value of the aggregate life support system, including the structure and function, has seldom been recognised or taken into consideration in economic valuation. Nevertheless, it is considered that the entire value of the system will include the values derived from the ecological sector and the economic values that are generated subsequent to these values. For this reason the outer border of Figure 2.1 is labelled total environmental value to illustrate this concept. From this viewpoint, the total environmental value of a wetland consists of primary or ecological values and secondary or economic values (Gren et al., 1994).

An impediment to measuring primary or ecological value is the lack of information on important ecological and hydrological processes that underpin the various values generated by wetlands (Barbier et al., 1997). If the attributes of ecological value that provide instrumental human benefit could be identified, then it may be possible to link ecological and economic value (Bingham et al., 1995). This concept is illustrated in Figure 2.1. Lack of information of ecosystem values on which wetland functions and services depend thus increases the difficulty in assessing the full economic value of wetlands. It is also noted that there is continuing scientific uncertainty about the precise extent and significance of wetlands functions and services and therefore it is not possible to calculate the marginal product of a wetland accurately\(^1\) (Turner, 1991). This uncertainty also extends to the global function of wetlands in their biogeochemical cycling role (Turner, 1991).

It is because of this uncertainty that fears have been raised that the conversion of wetlands to other uses is resulting in a loss of total benefits (for example, see Dugan, 1990; Maltby, 1986; Mitsch et al., 1994; Williams, 1990a). That is, it is concluded

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\(^1\) Mitsch and Gosselink (2000) note that the relationship between wetland area and marginal value is complex. The marginal value of wetlands increases with human development only to the point where wetland functions begin to be lost. As wetlands become smaller and isolated their ability to provide the full range of functions and services will be lost.
that a lack of knowledge and an inability to incorporate the value of ecological attributes in economic valuation of wetlands has led to a loss of these valuable resources. A summary of the wetland structures, processes and functions is given so that they may be linked to economic values through the ecological-economic interface.

2.2 THE ECOLOGICAL SYSTEM

Before examining the ecological system with reference to wetland functioning in more detail, it is necessary to define the terminology used. The following section will define the terminology. The structures, processes and resultant functions of wetlands, which provide services, are then outlined. The purpose in detailing the functions of wetlands is to gather information on the possible economic values they may provide. This information is necessary for determining the important wetland functions provided in the case study area, Moreton Bay.

2.2.1 Terminology

An understanding of the interface between the ecological and economic systems requires knowledge of the structures, processes and functions within individual wetlands. An initial stumbling block to this understanding is a lack of common terminology in describing the features of ecosystems (Turner et al., 2000). For example, Dugan (1990) argues the processes among the components of wetlands allow them to perform functions – such as flood control and storm protection and to generate products such as forest and fishery resources. For Dugan (1990) the attributes of wetlands are considered to be biodiversity and cultural uniqueness/heritage. Usher (1986) uses the terms attribute, criteria and values while economists discuss the value of services provided by wetlands (Costanza et al., 1997).

To promote consistency the terms used in this thesis to describe the features of an ecosystem are ecosystem structures, processes and functions. The outcomes of the interactions of ecosystems structures, processes and functions can then be considered as the attributes of the systems that then provide services for human benefit. The
context and use of the terms is schematically shown in Figure 2.2 which expands on the ecology sector in Figure 2.1 to distinguish between these terms and includes an example of the process of energy flows. The term *ecosystem structure* includes the geomorphologic and hydrological characteristics, the soils and flora and fauna at the site in question. The term also includes the species richness, composition and trophic organisation (McInnes *et al.*, 1998). *Processes* are changes or reactions which occur naturally within the ecosystem. These processes may be physical, chemical or biological (McInnes *et al.*, 1998). The term *function* is used to describe activities or actions, which occur naturally in the ecosystem as a product of the interaction between ecosystem structure and processes (McInnes *et al.*, 1998). Examples of the functions of wetlands include actions such as flood and water control, nutrient retention and food web support. It is these functions of ecosystems that provide services for humans\(^2\) (Costanza *et al.*, 1997).

**Figure 2.2** An example of the difference between structure, process, function, attribute and service

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<tr>
<td>*e.g. fishery and timber</td>
<td></td>
</tr>
<tr>
<td>products</td>
<td></td>
</tr>
</tbody>
</table>

Source: Adapted from Maltby (1998)

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\(^2\) The term ‘services’ is taken to include both the goods and services provided by ecosystems.
Figure 2.2 provides an example of the difference between structure, process, function, attribute and service. From Figure 2.2 it can be seen that information is required for an understanding of the complexities of ecosystem functioning and its relation to service provision. Also, the attributes of the system that combine the outcome of the structures and processes and functions need to be articulated and defined. It will be argued in Chapter 3 that ecosystems that exhibit particular attributes are ecologically valuable from an instrumental viewpoint. That is, they provide benefits for humans.

2.2.2 Wetland Structures, Processes and Functions

The National Research Council (1995) state that an assessment of wetland values requires a comprehensive scientific knowledge of wetland function. Keddy (2000) argues that this emphasis on function provides a bridge between wetland ecology and wetland services. To illustrate the connection between ecosystem function, services and values the functions of wetlands are reviewed using the concepts from Figure 2.2 of hydrological functions, biogeochemical functions and ecological functions. As noted, these functions result from an interaction of the structures and processes within the wetland. Thus, although the emphasis is given to wetland functions, as these are linked to the services provided in wetlands, the underlying structures and processes are acknowledged. The description of these functions (and the underlying structures and processes) is given in general terms noting Odum’s (1978) observation that wetlands occur under highly variable soil, geologic and vegetative conditions. Thus, each wetland will be to some extent ecologically unique depending on the extent and location of wetlands in the landscape and the size of the wetland (Mitsch and Gosselink, 1993). Turner (1991) notes that the interaction between wetland hydrology and topology, soil and emergent vegetation will determine the characteristics and significance of the processes that occur in a particular wetland.

**Hydrological Functions**

The hydrological functions of wetlands are related to water quantity functions and include floodwater control, groundwater recharge, groundwater discharge and surface water generation (McInnes et al., 1998). Odum (1978) states that the hydrology of the system is the chief forcing function that controls the wetland
structure and function. Mitsch and Gosselink (1993) point out that the hydrological conditions will affect many abiotic factors including soil anaerobiosis, nutrient availability and in coastal wetlands, salinity. These factors then influence the flora and fauna that in turn are active in altering the hydrology (Mitsch and Gosselink, 1993). Hydrologic pathways such as precipitation, surface runoff, groundwater, tides and flooding rivers transport energy and nutrients to and from wetlands. Thus, hydrology affects the species composition and diversity, primary productivity, organic accumulation and nutrient cycling in wetlands.

Wetlands provide for flood mitigation, storm abatement and aquifer recharge (Mitsch and Gosselink, 1993). As these functions occur at the ecosystem level they provide indirect human benefits and have often been ignored in wetland valuation. These functions will vary according to the season and from year to year. As noted the character of wetlands is strongly influenced by the hydrology of the area but at the same time wetlands influence regional water-flow regimes (Mitsch and Gosselink, 1993). Where wetlands intercept storm runoff and store water they attenuate the flood peaks to slower discharges over a longer period of time thus reducing flood damage downstream (National Research Council, US, 1995). The ability of wetlands to reduce downstream flooding will be related to the size of the wetland, the distance the wetland is downstream, the size of the flood, the closeness to an upstream wetland and the lack of other upstream storage areas such as reservoirs (Mitsch and Gosselink, 1993). Wetlands also provide for long term surface water storage, which maintains habitat for species during dry periods.

In coastal areas, saltmarsh and mangrove wetlands also act as giant storm buffers (Mitsch and Gosselink, 1993). Similarly, seagrass beds reduce wave action on adjacent coastlines. Dennison and Berry (1993) argue that the effects of strong storm surges are dampened through the friction provided by wetlands, which also reduces current velocities. Erosion is also reduced through the increased durability of the shoreline provided by the binding of roots (Dennison and Berry, 1993). Thus, natural marshes are able to protect development further inland. Williams (1990a, 17) estimates that coastal marshes dissipate more than 50% of wave energy within the first 2.5 m of coastal marshes and it is virtually eliminated at 30 m. Removal of these wetlands has led to high damage costs of inland developments that are often
paid for by the public (Mitsch and Gosselink, 1993). Also, as wave damage to the shore line is decreased there will be an increase in sediment build up. This in turn further reduces the potential for erosion (William, 1990a).

Some wetlands also recharge groundwater but it must be noted that this ability is limited where the soils under the wetlands are impermeable (Maltby, 1986, Williams, 1990a). However, it is thought coastal wetlands maintain water pressure on groundwater supplies. This inhibits the seepage of saline water into the aquifer (Maltby, 1986).

**Biogeochemical Functions**

Biogeochemical functions are related to the ability of wetlands to improve water quality (McInnes et al., 1998). According to the National Research Council (1995) these functions include the transformation and cycling of elements, retention and removal of dissolved substances, the accumulation of inorganic sediments and the accumulation of peat. Some of these functions occur at the individual wetland level and again may not be included in wetland valuation as these functions provide indirect benefits. Other biogeochemical functions influence global cycling of chemicals so that these benefits accrue at the landscape, regional and global level and cannot be attributed to a single wetland.

Wetlands are able to improve water quality by accumulating nutrients, trapping sediments and through the removal of organic and inorganic nutrients and some toxic material from the water that flows across them (Mitsch and Gosselink, 1993; National Research Council, 1995). When water enters the wetlands the velocity is slowed causing sediments and chemicals attached to them to drop out of the water column. The anaerobic and aerobic processes within the wetlands then allow denitrification, chemical precipitation and other chemical reactions that remove certain chemicals from the water (Mitsch and Gosselink, 1993).

The role of wetlands in removing nitrogen and phosphorus is becoming increasingly important as they receive increasing amounts of fertilizer enriched runoff and sewage (Williams, 1990a). Nitrogen removal occurs in wetlands due to the low-oxygen soils favouring denitrification by certain bacteria. Under aerobic conditions nitrifying
bacteria convert $\text{NH}_4^+$ to $\text{NO}_2^-$ and $\text{NO}_2^-$ to $\text{NO}_3^-$ (Dennison and Berry, 1993). This nitrification process only occurs when nitrogen is initially present in the form of ammonia. $\text{NO}_3^-$ can be used by plants and facilitates denitrification. Greenway (1993) notes that denitrification occurs under anaerobic conditions when $\text{NO}_3^-$ is reduced to oxides of nitrogen and finally nitrogen gas which is released to the atmosphere. Denitrification is the only long term process for nitrogen removal from wetland systems (Maltby, 1986; Greenway, 1993). Maltby (1986) states that 40-90% of nitrogen can be removed from wetlands through this process. Microorganisms cannot transform phosphorous into gas but it is taken up temporarily by plants and released when they decompose (Dennison and Berry, 1993). However, large amounts of phosphorus are inactivated by bonding to inorganic ions, mainly aluminum and iron (Maltby, 1986).

The toxic residues of heavy metals, pesticides and herbicides can also be removed from the water in wetlands through ion exchange and absorption in the organic and clay sediments (Williams, 1990a). Given the high productivity of wetlands there is then a high uptake of these minerals by the vegetation and subsequent burial in the sediments when the plants die (Mitsch and Gosselink, 1993). Williams (1990a, 21) states that depending on the effectiveness and efficiency of these process between 20 –100% of pollutants can be removed depending of the nature of the pollutant and the type of wetland.

Given the ability of wetlands to improve water quality, Odum (1978) suggested that large wetlands in urban and agricultural regions are able to provide solar-powered, self-maintaining tertiary treatment systems for wastewater. This ability has been exploited to provide primary, secondary and tertiary treatment for wastewater flowing through wetlands (Ewel, 1997). Maltby (1986) notes that wetlands can transform, fix and render harmless viruses, coliform bacteria and the suspended solids that remain after secondary treatment.

Williams (1990a) suggests that the ability of wetlands to process animal and human wastes is related to their high productivity (with plant growth removing pollutants from the water and substrate), the absorption of pollutants by the high rate of sediment disposition and the bacterial action in the sediments. Evidence from
cypress swamps in Florida, US demonstrates that these systems when used for tertiary treatment can remove 98% of nitrogen and 97% of all phosphorus before entering groundwater (Maltby, 1986). Further, a study of wastewater treatment in these Florida cypress ponds by Ewel (1997) showed that some of the sediments remained as suspended solids, some were mineralized and others were buried. However, Ewel (1997) notes that the use of natural wetlands for water quality improvements may lead to the diminution or loss of other services.

Mitsch and Gosselink (2000) note that while the value of wetlands in storm abatement and water quality improvement is increasingly being recognised at a local scale (although perhaps not fully valued) the role wetlands play in the global cycles of nitrogen, sulfur, methane and carbon dioxide have received less attention. These benefits are the most difficult to quantify as they are by their nature global. They suggest that one of the most important functions of wetlands may be their role in the sequestering and releasing the major proportion of fixed carbon in the biosphere. This property of wetlands is of increasing importance with accumulation of carbon dioxide in the atmosphere producing global warming (Mitsch and Gosselink, 1993).

The global carbon cycle may be affected in several ways by the cycling of carbon within wetlands (Mitsch and Gosselink, 1993). Northern peatlands continue to accumulate carbon. However, this accumulation is offset from the draining of wetlands for agriculture and other uses, the combustion of peat and emission of methane (Mitsch and Gosselink, 1993). Thus, Mitsch and Gosselink, (1993, 527) state that “wetlands like tropical rain forests may be shifting from a net sink to a net source of carbon to the atmosphere”. The authors therefore consider that wetland protection should be encouraged. Wetlands may also be important in the global cycling of sulfur. It is known that the global atmospheric levels of sulfur have been increasing due to fossil fuel burning and natural biogenic sources. This has led to acidification of lakes and streams when the sulfur is washed out of the atmosphere by rain. However, when sulphates are washed into wetlands they are reduced to sulfides, some of which can be returned to the atmosphere as hydrogen, methyl and dimethyl sulfides (Dennison and Berry, 1993; Mitsch and Gosselink, 1993). Thus, volatile carbon, nitrogen and sulfur are all important outputs from the anaerobic sediments of wetlands (Odum, 1978).
Ecological Functions

The ecological functions of wetlands are related to habitat provision and include ecosystem maintenance and food web support (McInnes et al., 1998). It should be stressed that these functions are dependent upon and interconnected with the hydrological and biogeochemical functions of the wetlands. Wetlands provide both temporary and permanent habitats for a variety of flora and fauna (Greenway, 1993). These habitats provide roosting, breeding and nursery areas for wildlife and provide vital refuges during periods of drought. They are also host to vast genetic resources, which are intermediate goods that enhance the production of other goods. Maltby (1986) states that the genetic resources of wetlands are not well understood.

The populations that depend on wetlands for their survival provide value for both human consumption and through their contribution to the organisation of species diversity within the wetland. The biological diversity of wetlands includes macrophytes (aquatic plants), microorganisms, invertebrate and vertebrate species (Greenway, 1993). Only a fraction of the rich and diverse population of animals and plants is tapped for commercial use. For example, rice, sago, oil palm, mangroves, crayfish, shrimp, oysters, caimans, waterfowl, fish and fur bearing animals provide products for humans and are dependent on wetlands and each other though complex food and nutrient cycles (Maltby, 1986). Wetland vegetation also supplies timber resources and reeds and palm fronds are also harvested (Maltby, 1986; Mitsch and Gosselink, 1993). For example, the cutting of mangroves provides timber for firewood, charcoal and poles and nipa leaves are used for thatching (Hamilton and Snedaker, 1984). Wetlands worldwide also provide great reservoirs of buried peat. In some cases, such as the former Soviet Union, peat resources have been used as a fuel source for hundreds of years (Mitsch and Gosselink, 1993).

It is the high primary productivity of wetlands compared to other ecosystems that allows them to sustain large populations of animals dependent on each other (Mitsch et al., 1994). While many factors influence productivity such as nutrients, climate, soil or sediment conditions, a common factor is the positive effect of water flow (Odum, 1978). The flow of water helps to circulate the nutrients, food and waste products among the system allowing organisms to use more of their productive energy for growth (Odum, 1978). Williams (1990a, 22) estimates that 24% of total
world net primary productivity is generated by wetlands and yet they cover only 6% of the earth surface. Maltby (1996, 25) notes that the average yearly animal protein production in swamps and marshes is 9g m\(^{-2}\), which is 3.5 times the average for natural terrestrial ecosystems. In turn, estuaries are considered to be twice as productive as swamps and marshes. Primary productivity of wetlands rivals that of rain forests and cultivated land but, unlike agricultural production, is generated without artificial inputs of fossil fuels and fertilisers (Keddy, 2000).

It is estimated that nearly two-thirds of the commercial, salt-water fish caught in the US are dependent on coastal estuaries and their wetlands (Williams, 1990a). Further, Berry (1993, 55) estimates that between 66% and 90% of the commercially important fish and shellfish species on the Atlantic and Gulf coasts depend on wetlands for at least part of their life cycle. Some of these species are permanent residents of the wetlands while others are more transient, with coastal wetlands providing important nursery and feeding areas (Mitsch and Gosselink, 1993). Nearly all freshwater species are dependent on wetlands to some degree. Saltwater species typically spawn offshore and move into the coastal wetlands during their larval and juvenile phases (Mitsch and Gosselink, 1993). Quinn (1993, 39) estimates that in Queensland, Australia, 75% of commercially harvested fishery resources (by weight) are dependent on coastal wetlands during some part of their life cycle.

The most researched and well known functions of wetlands are their importance as year-round habitats, breeding grounds and areas of wintering for migratory birds (Williams, 1990a). This function produced the impetus to conserve these ecosystems through the Ramsar Convention. In the United States, it is estimated that more than 50% of the 800 species of protected migratory birds rely on wetlands (Mitsch and Gosselink, 1993, 510). Another example of this dependency is found in the intertidal flats of Banc d’Arguin National Park in Mauritania where the wetlands provide a wintering site for some 3 million shorebirds each winter (Dugan, 1993, 27). In the case study area of Moreton Bay, Queensland, ANCA (1996, 370) estimate that more than 50000 migratory waders from 34 species depend on coastal wetlands for survival during the non-breeding season.
Wetlands support a disproportionately high number of endangered and threatened species. For example, Mitsch and Gosselink (1993, 517) estimate that wetlands occupy about 3.5% of the land area of the United States and 50% of the 209 species listed as endangered in 1986 depend on the wetlands. Examples of species considered endangered that use wetlands include species of crocodiles, monkeys and turtles and the West African manatee (Maltby, 1986). In the case study area, ANCA (1996) asserts that the wetlands of Moreton Bay support the largest feeding concentration of endangered Green turtles (*Chelonia mydas*) in Australia and the significant population of dugongs (*Dugong dugon*) is considered threatened.

The above summary of wetland functions demonstrates the multifunctional nature of these systems. It also highlights the fact that many of the benefits accrue at the landscape or even global level, which increases the difficulty of assigning value to these functions. An emphasis has also been placed on the interconnections between these functions to demonstrate that the use of wetlands for one function may result in the loss of other functions. Thus, from a management perspective it is necessary to understand the complexity of the interactions of functions within the wetland while taking a whole of catchment approach\(^3\). The information on wetland function is necessary to understand how these functions relate to the provision of instrumental benefits.

### 2.3 THE ECOLOGICAL ECONOMIC INTERFACE

The ‘sector’ termed “ecological economic interface” in Figure 2.1 attempts to represent the transformation of ecological functions to economic goods and services. From the economic perspective, the important issue is how does a change in ecosystem function (ecological value) lead to a change in economic value? It is assumed that if secondary economic values are dependent upon the outcomes of the interactions of the structures, process and functions of a system, then changes in these functions would also result in an alteration of their economic values. To understand what changes would occur requires knowledge of the links between the systems (Costanza *et al.*, 1989).

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\(^3\) A catchment is the area of land which collects and transfers rainfall to the waterways and wetlands. There may be several areas of wetlands connected to and within the catchment area.
From a management perspective the question is: under what conditions should wetlands be left intact and when should they be converted to other uses? (Dixon and Lal, 1997). To answer this question requires an understanding of the ecological impacts of conversion to both the remaining wetland and to the overall production of goods and services dependent on the wetlands (Dixon and Lal, 1997). It is also necessary to understand the marginal impacts of wetland conversion. That is, what area of wetland needs to be maintained to produce a certain level of goods and services? Without a knowledge of the links between the two systems, past analyses of wetland systems have included the direct cost of conservation (in terms of development benefits forgone) but have tended not to include the indirect benefits from conservation (Dixon and Lal, 1997). Benefits that may be ignored include traditional subsistence uses, benefits from products that are spatially removed from but dependent in some way on the wetland and the ecological benefits for which there are no measured market values (Dixon and Lal, 1997). Thus, Dixon and Lal (1997) state that a combined economic and ecological analysis is needed to analyse the management and development options for coastal wetlands.

The two systems could be linked if models could be developed that allowed information on ecological functioning to be directly linked to economic goods and services. However, each function may produce more than one service or alternatively more than one function may be necessary to produce one output. Thus, it is unlikely that any one model will be able to capture the links between ecosystems functioning and all the goods and services produced in wetlands. Nevertheless, a number of studies have been undertaken that consider the linkages between wetland area and fisheries production.

The linkages between coastal and offshore fisheries production and mangroves is well documented (see, Ruitenbeek, 1994; Dixon and Lal, 1997; Rönnbäck, 1999). Dixon and Lal (1997) point out that while the impact of destroying an entire wetland on offshore fishery production could be determined, what is not known is the marginal impact of the destruction or conversion of part of that wetland system. The models which use ecological data (fish species and abundance) and economic information (catch, effort and market price) provide one approach to overcome the
problem identified by Dixon and Lal (1997), albeit in a very limited manner. Studies which have attempted to link fisheries production and wetlands are reviewed in order to support the conclusion that information of the ecological-economic interface is currently limited. A review of these studies also demonstrates the difficulties that exist in linking the simultaneously and independently produced services of wetlands to wetland area and function.

2.3.1 Linkages between wetlands and fishery production

Ruitenbeek (1994) attempted to estimate the optimal area for wetland conversion to forestry by estimating the biogeophysical linkages between mangroves and the production of other goods such as offshore fisheries. However, it was noted that although the ecological linkages between the components were significant they were uncertain. For example, there were no specific data on the ecological linkages in the area, thus the linkage scenario had to be assumed. Ruitenbeek (1994) notes that it is unknown if the assumptions used were correct. However, the study did demonstrate that where strong ecological linkages were assumed to exist (including a factor that included a delay in impacts) the preferred forestry option changed from the total clear cut option to one where only 25% of total area was cut (Ruitenbeek, 1994, 245). Ruitenbeek (1994) argues that if the nature of interactions is not known (or not included) then economic valuation may underestimate the true impacts of conversion. In conclusion, Ruitenbeek (1994) states that there is a need for information relating to the linkages between ecosystem components and that policies which mitigate the impacts of the linkages will have economic as well as ecological merit.

Another approach taken to assess the linkages between wetlands and fisheries has been to view the ecological functions of mangrove wetlands as an input into the production of off-shore fisheries (Lynne et al., 1981; Kahn and Kemp, 1982; Barbier, 1994). This contribution can be valued using a production function approach. However, Lynne et al., (1981) note that the linkages between wetlands and populations of fish species are not known with any degree of precision. Similarly, a study undertaken by Kahn and Kemp (1985) to estimate the loss of fish due to losses of submerged aquatic vegetation, only included one species (striped bass) in their
study, as data were not available for other species. Barbier (1994) also considers that applying the production function approach to tropical wetlands may be a useful way of estimating indirect values such as fisheries. However, Barbier (1994) states that the importance of understanding the relationship between the environmental regulatory function and the economic activity was essential.

Bell (1997) undertook a study to link recreational fishing to wetlands in the southeastern United States. A production function approach was used to link recreational catch to the inputs of angler fishing effort and wetlands. Cross-sectional data was used to link changes in wetland area to changes in the recreational demand curve. Consumer surplus was then estimated by linking shifts in the empirical production function to angler demand curve through success elasticities. This approach allowed an estimate of consumer surplus related to changes in wetland area. However, it was noted only one product arising from wetlands was estimated. This approach, while highlighting the value of recreational fisheries, cannot account for all linkages between wetland area, function and goods and services produced. Bell (1997) also notes that except for Lynne et al., (1981) there was little empirical evidence linking the fish harvest to changes in wetland area.

Nickerson (1999) used a population dynamics model to illustrate the effects of changes in mangrove area to the population of a single mangrove dependent fish species in the Philippines. The purpose for developing the model was to examine differing management regimes for mangrove areas, given that mangroves in the Philippines are being converted from a common property resource providing multiple benefits to a privately owned resource for shrimp culture production. The model considered only the influences of catch and habitat on the population and assumed productivity was constant. The relationship between the dependency of the biomass on the carrying capacity and the relation between carrying capacity and habitat was not addressed. Thus, although the study provides useful insights into the trade-offs between leaving the area underdeveloped and other development choices, it still lacks the ability to model accurately the relationship between wetland area and function and the outputs of the system.
The loss and degradation of wetlands resulting from an underestimation of the value of mangroves was also noted by Gilbert and Janssen (1998). The authors identified a key obstacle to overcoming this problem. It was the inability of ecologists to deal with the interconnectedness within and between ecosystems. To overcome the difficulty of ecosystem functions producing multiple goods, each good and service provided by the mangroves in the study area (Philippines) was linked back to an environmental function using system diagrams. Gilbert and Janssen (1998, 328) linked 11 functions to 14 goods and services. However, information was not available to link the off-shore fisheries to mangrove area through a production function approach. Monetary estimates were only made for the value of fisheries and wood production but all other benefits were only considered in a qualitative way with respect to the management regimes being investigated. The study assumed that the waste assimilation services provided by the mangroves were only utilised by the aquaculture sector in that area.

This study was critiqued by Rönnbäck and Primavera (2000) for a number of reasons. The review highlights a number of the obstacles in linking wetland function and service. For example, Rönnbäck and Primavera (2000) noted that productivity estimates did not include stock assessments and were based on incomplete surveys. Also, Rönnbäck and Primavera (2000) disputed the cost of harvesting. The estimation of catch and effort is an important element of the production function approach and has implications for the values found. The Gilbert and Janssen (1998, 337) study assumed harvesting costs of 87.75% of total costs while Rönnbäck and Primavera (2000, 137) argue that harvesting cost for the mangrove on-site fisheries was only 0-10% which would lead to differing conclusions as to the best management regime for the wetland. Further, Rönnbäck and Primavera (2000) note that the study failed to consider the value of offshore fisheries and made no effort to quantify the ecological services of disturbance regulation, waste treatment, nutrient cycling and biodiversity. The conclusions of Rönnbäck and Primavera (2000) highlight the manner in which mangroves will be undervalued if the above goods and services of wetlands are not included in calculations.

Rönnbäck (1999) argues undervaluation resulting from ‘ecological illiteracy’ is a major force in conversion of mangroves. That is, the lack of understanding of the
ecological functions of mangroves has led to an underestimation of their value. Further, it is noted that the economic value of mangroves is often underestimated since only one or a few species of commercial importance are included in the calculations. In addition, Rönnbäck (1999) argues that the link between mangroves and marketed fish is not always perceived as many species are harvested outside of mangroves although they have been dependent on them during other parts of their life cycle. Also, a proportion of the catch is non-marketed subsistence and recreational harvest which is not included in fishery statistics. Nevertheless, Rönnbäck (1999) reviewed a number of studies to illustrate seafood production supported by mangrove ecosystems. Rönnbäck (1999) notes that further research is required to evaluate all the natural products and ecosystems services provided by mangroves. It was concluded that the value of penaeid fishery production was 162 kg/ha which at a market price of US$7.00 equals a value of US$1134. The market value of commercial fish species that utilise mangroves was estimated at US$475-5330 /ha/year (Rönnbäck, 1999, 247).

Evidence for the commercial value of fish species used by Rönnbäck (1999) comes from a study undertaken in Moreton Bay, Australia by Morton (1990). Morton (1990) estimated the total biomass in a 3340m² area of mangroves twice a month for a whole year. From the total fish catch of Moreton Bay, the average value of marketed fish was estimated. However, the value did not include the numerous important juveniles caught in the study (Rönnbäck, 1999).

The studies reviewed demonstrate that it is possible to link wetlands to capture fisheries but care needs to be taken to include all species that are dependent on the wetlands. Difficulties with applying the production function approach due a lack of ecological knowledge also need to be overcome. Further, care must be exercised with respect to how the value of catch and effort are included in the calculations. Also, Barbier (1995) argued that the application of the production function approach may be more problematic when considering multiple use systems such as wetlands as the ecological relationships between the multiple uses must be carefully constructed. Nevertheless, the studies do demonstrate that this is an area where is may be possible to straddle the ecological-economic interface.
2.3.2 Scale and the ecological-economic interface

A difficulty in straddling the ecological-economic interface is the difference in the scale of studies undertaken by ecologists and economists. This relates to the fact that the scale of the problem is studied at different levels, for different populations with differences in the spatial and temporal extent. For example, ecologists may relate a change in wetland area to the population of one species of fish. On the other hand, economists estimating the value of the change in wetland area need to account for all products of the wetland at the scale of the local and regional economy. That is, the value of the wetland for differing stake holders will differ depending on the spatial scale of analysis (Mitsch and Gosselink, 2000). This problem is exacerbated as common definitions of appropriate scales do not exist within disciplines or more or less between disciplines (Gibson et al., 2000). Thus, the issue of scale is crucial for understanding the ecological-economic interface.

Gibson et al., (2000) have provided a review of definitions of scale, levels, extent and resolution. These are reproduced in Figure 2.3 to provide a framework for investigating the differences in studies and identify possible solutions to aid in the linking of the ecological and economic systems.

<table>
<thead>
<tr>
<th>Term</th>
<th>Definition</th>
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<tbody>
<tr>
<td>Scale</td>
<td>The spatial, temporal, quantitative, or analytical dimensions used to measure and study any phenomenon</td>
</tr>
<tr>
<td>Extent</td>
<td>The size of the spatial, temporal, quantitative, or analytical dimensions of a scale</td>
</tr>
<tr>
<td>Resolution</td>
<td>The precision used in measurement</td>
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<tr>
<td>Hierarchy</td>
<td>A conceptually or causally linked system of grouping objects of processes along an analytical scale.</td>
</tr>
<tr>
<td>Levels</td>
<td>The units of analysis that are located at the same position on a scale.</td>
</tr>
<tr>
<td></td>
<td>Many conceptual scales contain levels that are ordered hierarchically, but not all levels are linked to one another in a hierarchical system</td>
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Source: Gibson et al., (2000, 218).
The level at which studies are undertaken determine, in part, the extent and resolution of scale. Hierarchies can be considered as naturally occurring levels that share similar spatial and time boundaries and interact with higher and lower levels in a systematic way (Costanza, 1996). For example, Gibson et al., (2000) argue that nested hierarchies are characteristics of complex systems, including all living organisms and most complex non-living systems. Lower levels combine into new units that have new organisations, functions and emergent properties. Thus, natural scientists study from the level of the molecule to cells, tissues, organisms, populations, communities, ecosystems, landscapes, biomes and finally the biosphere (Gibson et al., 2000). Social scientists on the other hand may start at the individual level, and then families/households, neighbourhoods, cities, landscapes, regions, nations and international organisations (Gibson et al., 2000). The differing levels have given rise to separate sub-disciplines in both the natural and social science. The question here, is what is the appropriate level, extent and resolution for the study of wetlands that can fully incorporate both ecological and economic values.

Mitsch and Gosselink (2000) consider that the hierarchy levels at which ecologists study wetlands are population, ecosystem and biosphere. For economists the difficulty of calculating the economic value of wetlands increases as the level study moves from the population to the biosphere. This results from the fact that at higher ecological levels benefits extend to higher economic level. For example, the studies presented above have considered the links between ecological function at the population level and the benefits to the level of the individual or at best the local economy.

The extent of measurement fixes the outer boundaries of the phenomenon being measured (Gibson et al., 2000). Thus, the extent of scale of an ecologist studying a wetland will depend on the nature of the investigation. It may consist of a few square metres and the rate of denitrification over hours or days or may consider changes in populations of species within the entire wetland (many square kilometers) over years. If the purpose of the ecological investigation was to provide information on the linkages between ecological functions and the production of services then it is suggested that the extent could be increased from a single patch of wetland to the interaction of the mosaic of saltmarsh, mangrove, seagrass and coral components
over a period of years. Further, Mitsch and Gosselink (2000) suggest that individual wetland should also be considered within a catchment perspective.

If this information was available then economists could also extend the spatial and temporal extent of their studies. For example, if the time over which benefits accrue was extended then value of wetland (undiscounted) over tens or hundreds of years could be calculated. This stream of benefits may in turn provide a greater value for a greater number of individuals than the short term destruction for the benefit of a few. The spatial extent over which the benefits are received may also change the economic calculations, as shown in the Gilbert and Janssen (1998) study which considered only the local scale the value of offshore fisheries and waste assimilation services of wetlands for the region were ignored. Yet, it was these ecosystem services which extend beyond the local level of scale that are considered important (Rönnbäck and Primavera, 2000).

The resolution or precision of measurement required to estimate changes in economic value is another limiting factor. The value of a wetland is measured by economists as the value of a marginal change of wetland area. Thus, the change of goods and services resulting from a loss (or gain) in area must be sufficient to be perceived in market prices or willingness to pay. If the resolution of the data on which these estimates are based is coarse (for example, aggregated annual value of fish catches) the change in value may not be detected. Again, the level at which the data is collected (individual fisher or national statistics) will influence the aggregation and hence resolution of the data and the changes which can be detected.

The review of studies demonstrates there is presently insufficient information on the linkages between the structures and processes that transform matter and energy inputs into ecosystem services to value wetlands as a direct input into the production of goods and services (Bingham et al., 1995). The extent of scale and level of study also needs to be extended for both ecological and economic studies before it possible to relate changes in the ecological value of wetlands to changes in economic value. Given that this scale of information does not presently exist it is necessary to use other methods to link ecosystem functions and the economic value of their outputs.
Previous attempts at placing an economic value on wetlands are reviewed in an attempt to determine if suitable methods exist for this task.

2.4 THE ECONOMIC SYSTEM

The third sector of Figure 2.1 relates to the economic value of wetlands using a total economic value framework. This framework considers that the economic valuation of wetlands should include both the use and non-use values of the systems. Figure 2.3 expands on the economic sector of Figure 2.1 to illustrate the components of total economic value. The taxonomy of total economic value includes explicit use benefits, indirect use benefits and implicit non-use benefits (Turner and Pearce, 1993). Sub-components of these categories include the notions of option value, quasi option value, bequest value, existence value and vicarious values. The classification of these sub-components of total economic value has been debated in the economic literature (Young, 1991). For example, Mitchell and Carson (1989) consider bequest and vicarious values to be sub-components of existence value, while Pearce et al., (1989) view vicarious value as a sub-component of option value. Wilks (1990) considers that the category of non-use value includes option and quasi-option values in addition to bequest and existence and vicarious values. Given the problem in agreeing on the correct taxonomy it is suggested that the focus should remain on total economic values measures as these are typically what are required for decision making (Young, 1991, Boardman et al., 1996). The classification of the components of value, the taxonomy presented in Figure 2.2 is derived from Barbier (1993; 1994; 1995) which in turn has also been adopted by Turner, (1991; 1999) and Turner and Pearce (1993) and Turner et al., (1995). Each of these components of total economic value is reviewed to illustrate the range of goods and services included in each category.
2.4.1 Use Values

Direct use values are those values which provide direct benefits for humans. Using wetlands as an example, this category would include all the products derived from the wetland either on or off site such as fishery and forest products and water use. This category also includes non-consumptive services such as recreation, tourism, in situ research and education and the aesthetic benefits provided by the presence of the wetland (Wilks, 1990; Barbier, 1993). Examples of economic estimation of the direct use value of wetlands include the value of wetlands for the blue crab fishery by Lynne et al., (1981). This study used a direct linkage model which examined the marginal value of the productivity of wetlands. Barbier (1993) estimated the value of Hadejia-Jama’re floodplain, Nigeria, in terms of the opportunity cost of its loss for crop production, fuelwood and fishing. Driml and McBride (1982) used the zonal travel cost method to estimate the value of boating and associated recreation in Southern Moreton Bay, Australia. The annual value of the site for recreational purposes was estimated to be A$1.7 million (1982 $) (Driml and McBride, 1982, 196). However, it was emphasised in the paper that this was an absolute minimum value as it did not account for the existence or option values and only considered one type of direct use. Similarly, Farber (1988) and Costanza et al., (1989) estimated the recreational value of wetlands in Terrebonne Parish, Louisiana using the travel cost and contingent valuation techniques. Bergstrom et al., (1990) also investigated the recreational value of the Louisiana wetlands by considering recreational expenditures and consumer surplus using the contingent valuation technique.
Ecosystem level functions typically provide for indirect use values. Their value derives from supporting or protecting economic values (Barbier, 1993). Examples from wetlands include storm buffering and flood reduction, ground water recharge, water quality improvements, biogeochemical cycling and the organisation of biodiversity (Barbier, 1993). As this contribution to human welfare is unmarketed and is usually financially unrewarded these values are more difficult to evaluate. However, as noted by Costanza et al., (1997, 256) wetland functions such as flood control, storm protection, nutrient cycling and waste recycling accounted for 80% of their economic value. Another example of the valuation of indirect wetland services is provided by Thibodeau and Ostro (1981) who considered the cost of property damage from flooding if the wetlands in Charles River watershed were lost. This study also included the values the wetlands provided in terms of increased property values, pollution reduction, water supply and recreation using a variety of techniques. Costanza et al., (1989) also included an estimate of the storm protection services provided by wetlands by assessing the increase in property damages from wind, which could be expected from hurricanes if coastal wetlands were lost. A more recent study by Farber (1996) considers the lost value resulting from wetland disintegration in terms of water treatment and aquifer recharge as well as other direct uses such as commercial and recreational values.

Option and quasi-option values are also considered a form of use value as these values relate to the benefit received from preserving the option to use the resource in future (Turner et al., 1995). That is, option value is the value obtained from retaining the opportunity to possibly use a resource at some time in the future. It is thus an expression of preference for the preservation of the environment against some probability the individual may use it some time in the future (Pearce et al., 1989). The option price is the sum of money or premium that the individual would pay now for the right to consume in the future. The option value is then the difference between expected consumer surplus and the option price (Krutilla and Fisher, 1975). Option value reflects individual risk aversion. In the absence of risk aversion, willingness to pay will equal the mean use value and option value would be zero (Goulder and Kennedy, 1997). Plummer and Hartman (1993, 81) have shown in the case of income and quality uncertainty option value will be positive but in the case of
uncertainty relating to consumers’ tastes it may be positive, negative\textsuperscript{4} or zero. Brookshire \textit{et al.}, (1992, 113) have also shown that option value is positive for risk averse individuals when future availability of the resource is uncertain.

Quasi-option value results from retaining the option of obtaining better information in the future in terms of alternative uses of the environment. This information can then be taken into account in determining development decisions that may result in irreversible environmental loss. If the development is undertaken it cannot be affected by this new information (Krutilla and Fisher, 1975). This value may be obtained from the benefit associated with new technology or knowledge which enhances the value of the natural resource (Wilks, 1990). The value of this information is conditional on not proceeding with the development initially (Young, 1991).

\textbf{2.4.2 Non-Use Values}

Resource values that are independent of the current and future consumption of an environmental amenity can be considered non-use values (Wilson and Carpenter, 1999). As Turner (1999) states it, individuals do not need to use or intend to use the resource either directly or indirectly to derive value from it. These values include the notions of bequest value, existence value and vicarious value. Bequest value represents the value for the current generation of preserving the environment for the use of future generations. Pearce \textit{et al.}, (1989) note that there is some contention as to whether bequest value is a form of option value as the value may be related to the use of the resource in the future. However, it is considered here that the value is derived from knowing the resource will exist in the future regardless of whether it is used or not. Pearce (1993) notes that when making a bequest for direct descendants one may be fairly confident about their preferences, however, when extended into the future there is uncertainty related to what the preferences of future generations will be. Thus, the preservation of the resource is not dependent on an expectation that it will be used.

Existence value is obtained from the knowledge that an environmental amenity exists

\textsuperscript{4} Option value may be negative for risk takers.
regardless of any desire to gain direct use or indirect uses of the goods and services it provides (Goulder and Kennedy, 1997). Wilks (1990) notes that existence value may also extend to the protection of culturally important resources. Existence value could include a pure biodiversity component, that is, an appreciation of the variation we observe in the ecosystem (Goulder and Kennedy, 1997). Public support for endangered species (such as whales) which most would never expect to see, is an indication that existence values exist (Pearce et al., 1989). Results from empirical studies using a questionnaire approach suggest that existence values are a substantial component of total economic value (Pearce, 1993). For example, Kirkland (1988, 112, 116) found that 78.2% of mean willingness to pay (NZ$12.68/household) for preservation of the Whangamarino wetland in New Zealand was for bequest and existence value. Stone (1992, 63) conducted a pilot study on the preservation value of the Baramah wetland, Australia and found that 71% of the mean willingness to pay (A$30.01) was related to bequest and existence values. A more recent study of the value of the wetlands at Lake Kerkini in Greece estimated that 96% of total economic value was attributable to non-use factors (Oglethorpe and Milliadou, 2000). Vicarious value is the value obtained from indirect consumption of an environmental resource, such as the existence of whales, through print and other media (Wilks, 1990). The plethora of ‘nature’ documentaries is testament to the vicarious value derived from species and ecosystems that might not otherwise provide use values. It is considered a subset of existence value and is implicitly included within this category throughout this thesis.

2.4.3 Limitations of the total economic framework

Difficulties will arise with the use of total economic framework if care is not taken to avoid double counting and conflicts between uses. This results from the fact that the system may provide for two or more components of value simultaneously, or the use of one benefit may reduce other benefits (Turner et al., 2000; Barbier, 2000). For example, coastal wetlands may provide water quality improvements through the retention of nutrients from up-stream. These nutrients may in turn support juvenile fish that supply an offshore fishery. If both the support to fisheries and the nutrient retention function are counted this will lead to an overestimation of value (Barbier, 2000). Similarly, there may be trade-offs among wetland functions. For example, if
A mangrove swamp is being valued as a forestry resource, thus destroying the wetland, the other benefits ascribed to the system may be lost.

The difficulties in assessing the components of total economic value are particularly apparent in the linked land-ocean system of coastal wetlands where complicated linkages affect the production of a wide range of goods and services of social benefit. The problem as Dixon and Lal (1997) state is that the narrow focus of many analyses fails to consider the dispersed nature of the products found in wetlands and those found outside the wetlands but are dependent on them. Thus, it is essential that when the values of wetlands are assessed it should be undertaken within the regional landscape as it is their contribution at this level that may be more important that at an isolated level (Odum, 1978, Mitsch and Gosselink, 2000). Hamilton and Snedaker (1984) provide a diagrammatic representation of the relationship between the location and type of mangrove wetland goods and services and their economic valuation. This illustrates the range of benefits that have been captured in economic assessments of wetlands and the offsite services that have typically been ignored. A modified version of this diagram is presented below in Figure 2.4.

Figure 2.4 The location and type of mangrove goods and services and their economic valuation

<table>
<thead>
<tr>
<th>Valuation of Goods and Services</th>
<th>Location of Goods and Services</th>
<th>Valuation of Goods and Services</th>
<th>Location of Goods and Services</th>
</tr>
</thead>
<tbody>
<tr>
<td>Marketed</td>
<td>On-site</td>
<td>1</td>
<td>Marketed</td>
</tr>
<tr>
<td></td>
<td>Usually included (e.g. poles, charcoal, mangrove crabs)</td>
<td>2</td>
<td>May be included (e.g. fish or shellfish caught in adjacent waters)</td>
</tr>
<tr>
<td>Non-Marketed</td>
<td>3</td>
<td>Seldom included (e.g. domestic fuelwood, nursery area for juvenile fish, feeding grounds for estuarine fish, viewing and studying wildlife)</td>
<td>4</td>
</tr>
</tbody>
</table>

Source: Adapted from Hamilton and Snedaker (1984, 110)
Figure 2.4 highlights the need to consider wetland services at all scales, local, regional and global. For example, most often only the on-site marketed products of the mangroves have been estimated in past analysis and where links are demonstrated between the mangrove and off-site marketed resources these values may also be included. However, it is the valuation of non-marketed goods and services which are often ignored in valuation due to either to the limitations of available methods or a lack of information on the linkages between the mangroves and services, that are required.

Thus, although the total economic framework outline above is well accepted as a framework for identifying all the values of wetlands, many studies related to measuring the economic benefits of wetlands only assess a subset or a number of subsets of these values. Some of these studies as reported in the literature are illustrated in Table 2.2 to show the range of economic studies that have been previously been undertaken for wetlands. From the table, it will be noted that the direct use values of fisheries and recreation have received greater attention than indirect use values and non-use values. Also, some studies estimate a range of values (using a variety of economic techniques) in an effort to determine the total economic value. However, the studies which are considered to provide for total economic valuation are those which include an energy analysis. This type of analysis has been included in Gren *et al.*, (1994), Gosselink *et al.*, (1974), Farber and Costanza (1987), Costanza *et al.*, (1989), Folke (1991) and Dale (1993).

Nevertheless, these studies have not included an estimate of the non-use values of wetlands. As such it could be argued that even these estimates fail to capture the full value of the wetlands. For example, Folke (1991) evaluated the life-support functions of the Martebo mire on the island of Gotland in the Baltic Sea using energy analysis. The monetary estimate of the annual undiscounted replacement cost was $0.4-$1.2 million (Folke, 1991), although this amount was an underestimate of the total value of the system as it only accounted for some of the secondary values and part of the system’s support value.

---

5 It may never be possible to determine the total economic value of an environmental resource.
6 An analysis of valuation techniques is given in Chapter 4.
Table 2.2 Previous reported studies of wetland values

<table>
<thead>
<tr>
<th>Service Valued</th>
<th>Studies Reported</th>
</tr>
</thead>
<tbody>
<tr>
<td>Direct Use Values</td>
<td>Gosselink et al., (1974)</td>
</tr>
<tr>
<td></td>
<td>Lynne et al., (1981)</td>
</tr>
<tr>
<td></td>
<td>Farber and Costanza (1987)</td>
</tr>
<tr>
<td></td>
<td>Kirkland (1988)</td>
</tr>
<tr>
<td></td>
<td>Costanza et al., (1989)</td>
</tr>
<tr>
<td></td>
<td>Morton (1990)</td>
</tr>
<tr>
<td></td>
<td>Barbier (1993)</td>
</tr>
<tr>
<td></td>
<td>Dale (1993)</td>
</tr>
<tr>
<td></td>
<td>Ruitenbeek (1994)</td>
</tr>
<tr>
<td></td>
<td>Dixon and Lal (1997)</td>
</tr>
<tr>
<td></td>
<td>Gilbert and Janssen (1998)</td>
</tr>
<tr>
<td></td>
<td>Nickerson (1999)</td>
</tr>
<tr>
<td></td>
<td>Rönnbäck (1999)</td>
</tr>
<tr>
<td>Value of fisheries (or value of potential loss)</td>
<td>Thibodeau and Ostro (1981)</td>
</tr>
<tr>
<td></td>
<td>Driml and McBride (1982)</td>
</tr>
<tr>
<td></td>
<td>Farber and Costanza (1987)</td>
</tr>
<tr>
<td></td>
<td>Farber (1988)</td>
</tr>
<tr>
<td></td>
<td>Costanza et al., (1989)</td>
</tr>
<tr>
<td></td>
<td>Bergstrom et al., (1990)</td>
</tr>
<tr>
<td></td>
<td>Gren et al., (1994)</td>
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<tr>
<td></td>
<td>Farber (1996)</td>
</tr>
<tr>
<td></td>
<td>Bell (1997)</td>
</tr>
<tr>
<td>Recreational and amenity values</td>
<td></td>
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<tr>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
</tr>
</tbody>
</table>

| Indirect Values                 | Gosselink et al., (1974)          |
|                                 | Thibodeau and Ostro (1981)        |
|                                 | Farber and Costanza (1987)        |
|                                 | Kirkland (1988).                  |
|                                 | Costanza et al., (1989)           |
|                                 | Dale (1993)                       |
|                                 | Gren et al., (1994)               |
|                                 | Farber (1996)                     |
| e.g. Flood control and nitrogen abatement | Kirkland (1988) |
|                                 | Whitehead and Bloomquist (1991)   |
|                                 | Stone (1992)                      |
|                                 | Fankhauser (1995)                 |
|                                 | Bennett et al., (1997)            |
| Preservation Value              |                                    |
|                                 |                                    |
|                                 |                                    |
|                                 | Farber and Costanza (1987)        |
|                                 | Costanza et al., (1989)           |
|                                 | Dale (1993)                       |
|                                 | Gren et al., (1994)               |

Given the large range of studies that have been used to value wetland services the range of values estimates for wetlands is also large (Woodward and Wui, 2001). A recent meta-analysis of wetland studies notes that the values contributed to wetlands
range from US$0.06 – US$22 050 per acre (Woodward and Wui, 2001, 257). As the values obtained are dependent on wetland characteristics and the type of study undertaken it is recommended by Woodward and Wui (2001) that site specific studies are needed for individual wetlands. However, an aggregate estimate of the value of wetlands has been formulated by Costanza et al., (1997) as part of an attempt to value the world’s ecosystem services and natural capital. They also noted the importance of some minimum level of ecosystem infrastructure is necessary to provide goods and services but this component of value was not included in the analysis. The results indicate that 15% of the value of the world’s ecosystem services and natural capital are generated by wetlands (Costanza et al., 1997). The world was divided into 16 biomes and ecosystem services were divided into 17 categories. Under the definition of wetlands provided by the Ramsar Convention (as given in Chapter1) wetlands are present in the biomes, estuaries, seagrass/algae beds and tidal marsh/mangroves (ANCA, 1996). However, the latter category is considered to define coastal wetlands with the tidal marshes of temperate areas providing similar functions to the mangrove wetlands of tropical and subtropical areas (Costanza et al., 1997). As noted above, these ecosystem level indirect benefits are often ignored in the assessment of the benefits of wetland conversions. The results of the study for the six ecosystem services identified for the biome and an estimate of the total ecosystem services as determined by energy analysis are given in Table 2.2. The table also includes some estimates from other studies cited above, using a variety of economic techniques, as a comparison.
Table 2.3  Review of estimates of the value of coastal wetlands

<table>
<thead>
<tr>
<th>AUTHORS</th>
<th>STUDY</th>
<th>METHOD</th>
<th>RESULTS</th>
<th>TOTAL</th>
</tr>
</thead>
<tbody>
<tr>
<td>Costanza et al. (1997)</td>
<td>Value of worlds ecosystems</td>
<td>Disturbance Regulation (damage estimation, WTP and replacement costs)</td>
<td>$/ha/yr (US1994) $1 839 $6 696 $ 169 $ 466 $ 162 $ 658 $ 466</td>
<td>$9 990/ha/yr (US1994) (Total global flow value $648/yr x10^9). This does not include total ecosystem services.</td>
</tr>
<tr>
<td></td>
<td>Global</td>
<td>Waste Treatment (replacement cost)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Tidal Marsh/mangroves Biome</td>
<td>Habitat/refugia (market price)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Food Production (market price)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Raw Materials (market price)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Recreation (travel cost CVM)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>TOTAL ECOSYSTEM</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$11 029</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ruitenbeek (1994)</td>
<td>Commercial and subsistence fish and crustacean spices (mangrove dependent)</td>
<td>Fisheries production from mangroves - assumed 50% loss would occur with conversion of mangroves</td>
<td>$150/ha/yr ($1986)</td>
<td>$2 700/ha (5% discount rate)</td>
</tr>
<tr>
<td></td>
<td>Indonesia</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Costanza, Faber,</td>
<td>Commercial Fishing</td>
<td>marginal productivity of acre of wetlands commercial fishery (MP, market price)</td>
<td>$317/acre (8%) $845/ acre (3%) $151/acre (8%) $401/acre (3%) $46/acre (8%) $181/acre (3%) $1915/acre (8%) $7549/acre (3%)</td>
<td>TPV -Best Estimate $2429-6400/acre (8% discount rate) $8977 -17 000/acre (3% discount rate)</td>
</tr>
<tr>
<td></td>
<td>Louisiana, USA</td>
<td>trapping (market price)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>recreation (from travel cost and CVM)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>avoided property damages</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>energy analysis based GPP conversion</td>
<td></td>
<td></td>
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<td></td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$6400-10600/acre av marsh (8%) $17 000-28200 (3%)</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Table 2.3 (continued).

<table>
<thead>
<tr>
<th>AUTHORS</th>
<th>STUDY</th>
<th>METHOD</th>
<th>RESULTS</th>
<th>TOTAL</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lynne et al., (1981) Florida, USA</td>
<td>Blue Crab Fisheries</td>
<td>Marginal productivity of wetlands for blue crabs</td>
<td>$0.30/marginal acre</td>
<td>TPV $3.00/marginal acre (10% discount rate)</td>
</tr>
<tr>
<td></td>
<td>Recreation and Amenity value of Broadland wetlands</td>
<td>Contingent Valuation Method</td>
<td>859 Swedish kroner/km for the restoration of the wetland</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Wetlands restoration for nitrogen purification - Gotland, Sweden</td>
<td>Marginal value of nitrogen abatement</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rönnbäck (1999) Global</td>
<td>Penaeid shrimp production</td>
<td>Average annual value based on 43 data sets</td>
<td>162/kg/ha@US$7/kg</td>
<td>SUS1134/kg/ha mangrove</td>
</tr>
<tr>
<td></td>
<td>Market value commercial fish species utilising mangroves</td>
<td>Annual Market value/ha based on 4 data sets</td>
<td>237 – 5840kg/ha/yr</td>
<td>SUS475-5330/ha mangrove</td>
</tr>
<tr>
<td>Morton (1990) Moreton Bay, Australia</td>
<td>Catch of marketable fish from mangroves</td>
<td>Average value of fish netted in a mangrove area</td>
<td>$8380/ha ($A 1988)</td>
<td></td>
</tr>
</tbody>
</table>

The studies reported in Table 2.1 and Table 2.2 illustrate the range of economic methods that have been used to value wetlands. The tables also demonstrate that different methods in differing wetlands produce a large range of values. One consensus from the studies however is the need to consider both the ecological and economic values of these systems. Examples include Lynne et al., (1991); Barbier (1994); Kahn and Kemp (1985), Baite and Mabbs-Zeno (1985), Costanza et al., (1989), McDonald and Brown (1992), Bergstrom et al., (1990), Gren et al., (1994), Turner et al., (1995) and Dixon and Lal (1997). As noted the studies undertaken by Lynne et al., (1991), Kahn and Kemp (1985) and Barbier (1994) attempted to estimate the productivity of wetlands for fisheries production. However, one of the limitations of these studies was a lack of reliable data of the linkages between the ecological and economic systems. Other studies that also identified the need to incorporate ecological and economic linkages are given to demonstrate the limitation of available ecological information.

At the single direct use level Baite and Mabbs-Zeno (1985) considered the benefits of wetlands for recreational housing using the hedonic pricing technique. Even at this level it was noted that the ecological information required to develop monetary measures of wetlands in their natural state is lacking. Further, the linkages between the existence of individual wetland area and levels of service are not well established. Another direct use benefit of wetlands that has received attention from economists is that of the recreational value of wetlands. For example, Bergstrom et al., (1990) examined the recreational values of wetlands and assumed that a loss of wetlands would result in a loss of dependent species which would in turn reduce the recreation opportunities to capture these species. Again, however, the information required on the links between physical and biological changes in the system and the change in services provided and thus economic value was lacking (Bergstrom et al., 1990).

A more comprehensive study undertaken by Costanza et al., (1989) estimated a range of values for a variety of wetland functions. The authors stressed that the economic value of ecosystems is connected to their physical, chemical and biological role in the overall system whether the public fully recognises that role or not. Further, they argue that valuing wetlands through the preferences of individuals who
may not be fully informed, will not capture the value of the overall system. Rather, the valuing of these systems from a scientific perspective is required and this information can then be used in decision making to correct market signals.

Similarly, Gren et al., (1994) and Turner et al., (1995) argue that due to lack of information and ecological knowledge, human preference models may not capture the indirect uses of these ecosystems. Gren et al., (1994) argue that the environmental values of complex ecosystems cannot be observed by individual behaviour in markets and thus economic methods cannot capture the value of ‘healthy’ ecosystems in sustaining socio-economic activity. This proposition is tested in this thesis.

McDonald and Brown (1992) attempted to use an input-output analysis to examine the ecological and economic linkages in Moreton Bay, Australia (the case study area). The authors stressed that it was essential to recognise that the economic activity in the Bay is dependent upon and intimately connected with the supporting ecological processes. It was noted that an understanding and quantification of the economic-ecological relationship was a difficult task and had not been addressed in the region before (McDonald and Brown, 1992). The two approaches taken in this thesis of determining the value of these wetlands through ecological and economic methods will thus add to the understanding and quantification of the ecological-economic relationships within Moreton Bay.

2.5 CONCLUSION

Using wetlands as an example, this chapter outlined the difficulties in assessing the full value of multifunctional ecosystems. Ecosystem services continue to be lost where these services are undervalued or ignored particularly at the ecosystem and global level. The obstacles in determining these value are the lack of information on the ecological functioning of the system, the ways in which these processes and functions provide instrumental benefits and a lack of economic techniques capable of assessing ecosystem and global level services.

A review of the literature presented in this chapter suggests that an ecological economics framework may provide a means of assessing both the ecological and
economic values of ecosystems such as wetlands. A conceptual framework for considering the values of the two systems has been developed and presented in Figure 2.1. In an ideal world, with complete information and understanding, the ecological functioning of these systems would be understood to a level which allowed an assessment of the trade-offs between ecological values and economic values. In this way, the marginal effects of a change in habitat size could be determined for the range of functions provided by wetlands and their economic value assessed accordingly. That is, the dynamics within the ecological-economic interface would be known. However, it has been demonstrated that this knowledge does not currently exist, even in well studied areas such as commercial fisheries. Thus, other approaches must be devised that can link the ecological system with the economic system. This thesis considers two possible approaches, one from an ecological perspective and the other from the economic perspective.

From an ecological viewpoint, the structure, processes and functions of an ecosystem could be described in terms of the attributes of the ecosystem that provide instrumental benefits. These attributes will encapsulate the ecological value of the system. If these attributes of ecological value can be defined and measured then it may be possible to rank ecosystems. In this way choices about conversion and conservation could be made based on an assessment of these ecological values. This approach, suggested by Costanza et al., (1989) and Gren et al., (1994) will be further developed in Chapter 3.

From an economic perspective, the systems could be linked by explicitly including the ecological values of a system when determining economic values. That is, if information on the ecological values of a system is provided to individuals then their economic preferences may reflect these values. Thus, the proposition that economic methods cannot capture the ecological value of wetlands as suggested Costanza et al., (1989) and Gren et al., (1994) will be tested. This approach requires the use of an appropriate non-marketed economic technique. Alternative non-market techniques are reviewed in Chapter 4.
CHAPTER 3  
DEFINING AND MEASURING ECOLOGICAL VALUE

The purpose of this chapter is to define ecological value. If the attributes of ecological value can be defined then their contribution to the production of instrumental benefits can be assessed. Changes in ecological value would then reflect changes in instrumental benefits. The chapter begins by examining approaches to, and perspectives on, ecological valuation. From this review, it is suggested that there is not a consensus on the definition of ecological value. This chapter fills this gap by developing a concept of ecological value from the attributes used to identify protected areas. The attributes considered important in protected area management are categorised as either relating to representativeness, viability or ecological attributes. It is argued that it is this latter category which includes the attributes of productivity, dependency and organisation, which reflects the essential ecological process and life support services provided by ecosystems. Indices of integrity and health are reviewed to aid in the development of an index of ecological value. An index of ecological value is then presented. Finally, consideration is given to operationalising an index that measures and ranks the relative value of ecosystems based on these attributes.

3.1 INTRODUCTION

Resource managers are faced with making resource allocation decisions that take into account the economic value of an ecosystem and the underlying ecological value of those systems. The choices made by decision makers are based on the human expressions of values given to the uses of the area or the area itself. That is, when choices are made about the uses of resources, values are implicitly or explicitly placed on the resources (Bingham et al., 1995). Determining the appropriate uses of ecosystems thus requires an understanding of and choices between the impacts of human use and resulting human benefits (Page, 1992; Norton, 1998). Approaches to valuation are discussed in the following section. A review of the literature does not produce a consensus on the definition of ecological value. However, the concept of conservation value provides criteria, which may include the attributes of ecological
value. The evolution of a definition of ecological value from conservation value is developed in Section 3.2. Sections 3.3 and 3.4 provide a definition of ecological value and possible approaches to its measurement. The final section outlines an index for ecological value and considers the problem related to measurement particularly with respect to scale.

3.1.1 Approaches to Ecosystem Valuation

Economics provides a theoretical framework for measuring both the benefits and costs of a variety of uses of ecosystems. However, these valuations have been criticised for a number of reasons. First, Costanza and Folke (1997) argue that economic valuations are based on the subjective preference of the current generation of humans for the perceived benefits of a resource. From this anthropocentric viewpoint, Goulder and Kennedy (1997) argue that value is determined by the instrumental preferences that are held by people. These preferences are often based on incomplete knowledge of the consequences of ecosystem use and may fail to consider the longer term implications for future generations and ecosystem functioning. Second, Goulder and Kennedy (1997) argue that economic valuations may only incorporate a subset of the values of a system with emphasis placed on the value of commercial products produced by the ecosystem. Nevertheless, Pearce et al., (1989) argue that it is recognised that ecosystems produce goods and services for humans that include both direct and indirect use and non-use values. The difficulty remains in developing economic methods that are capable of valuing the full range of benefits derived from ecosystems on which humans ultimately depend. Costanza et al., (1997) note that the previous approaches to environmental valuation have examined the value of goods and services derived from ecological systems. However, Costanza et al., (1997) note that the provision of these services depends on some minimum level of ecosystem infrastructure, which was not included in the analysis.

If it is considered that ecosystem valuation should not rely on the subjective preferences of consumers who may not be fully informed, then the task must fall on ecologists. If it is possible to define and measure the attributes of an ecological system that provide value for humans then these values could be used in conjunction
with subjective economic values. However, as noted in Chapter 1 there is no consensus on a definition of ecological value. Further, De Leo and Levin (1997) suggest that scientists should avoid assigning values and limit their advice to science and informing decision-makers but leave the complex trade-offs required for resource management to the stakeholders. The reason for this view is the present poor scientific knowledge of ecosystem structure and function and the need to avoid subjective judgements (De Leo and Levin, 1997). However, to leave the decisions to stakeholders who may also be poorly informed and likely to act on their own subjective interests may not improve the decision making process.

Given the lack of an ecological value system, the approach taken in this chapter is to value ecological systems using the economic concept of value, where value is based on an ecosystem’s ability to provide instrumental human benefits. Thus, the focus in this chapter is on developing a definition and index of ecological value, which reflects the underlying instrumental human benefits produced by ecosystems. In essence it is an ecological economics perspective of value. Following Reid (2000) it is assumed within this perspective that humans will continue to have a preference for ecosystem goods such as food, timber, genetic resources, medicines and services such as water purification, flood control, carbon sequestration, biodiversity conservation and disease regulation.

The development of a definition and index of ecological value provides a link between ecological and economic values. As the index includes both ecological and economic information, it may aid decision making when there is a choice to be made between two similar ecosystem types in terms of continuing degrading practices or conservation. Alternatively, where there is a proposed conversion of a system, such as the building of a canal estate through a wetland, an index of ecological value may be able to measure the value of the system before and after conversion. The difference in ecological value, and hence human benefits, could be compared to the perceived value of the conversion. That is, an index of ecological value would provide an added layer of ecological-economic information for decision-makers.

A difficulty with attempting to measure ecological value is that the term is generally used but not well defined within the literature (Bingham et al., 1995; Perrings, 1995,
Norton, 1998). To date, the concept has received greater attention from ethicists than ecologists or economists (for example, Rolston, 1986; Sagoff, 1988, 1992). As Norton (1998) points out, each discipline holds different perspectives of value depending on the underlying paradigm of that discipline.

While the halls of theoretical ecology may be silent in terms of what constitutes ecological value, there is one group of scientists, conservation biologists and ecologists who explicitly value ecosystems in terms of their conservation values. Conservation values are not synonymous with ecological value as they are used to define areas based on their particular aim such as conserving biodiversity (Primack, 1995). However, for the non-ecologist the criteria that are used to identify areas for conservation provide a basis for determining the attributes of ecological value. The leap from conservation value to ecological value is made by identifying the aims of conservation, such as protecting unique areas, and removing those aims that are solely related to the conservation objectives. The remaining criteria are considered to constitute the underlying ecological value of the ecosystem. The development of a formal definition of ecological value, and an associated index, is thus likely to evolve from the notion of conservation value. The purpose of defining ecological value is not to illustrate which ecosystems should be protected. The aim is to understand which ecosystems are likely to provide ecosystem services for human benefits so that these benefits will be fully accounted for in resource management decisions. The best methods for ensuring that ecosystems continue to sustain that range of benefits are outside the scope of this chapter.

3.2 THE EVOLUTION OF A DEFINITION OF ECOLOGICAL VALUE

The aims of conservation can be found within a set of ethical and ideological statements asserted by the discipline of conservation biology (Barry and Oelschlaeger, 1996). For example, it is contended by Primack (1995, 8-9) that “the diversity of organisms is good, the untimely extinction of populations and species is bad, ecological complexity is good, evolution is good and biological diversity has intrinsic value”. Thus, from a conservation biology perspective, conservation value is related to ecosystems that continue to sustain biodiversity. One method for conserving biodiversity is to establish protected areas.
Areas selected for protection for the purpose of conserving biodiversity should exhibit conservation values. Usher (1986) states that the aim of protected area management is to establish a system of representative ecosystems. The purpose of establishing these areas is, according to Bruckhorst (2000), to ensure their long-term ecological viability, to maintain their ecological processes and to protect biodiversity at all levels\(^1\). To meet these aims, protected areas are identified in terms of their representativeness, viability and their underlying ecological attributes. It is necessary to identify these components in order to determine which attributes relate solely to ecological value. The attributes used to select marine protected areas (MPAs) and wetlands will be considered sequentially for this purpose\(^2\). A possible classification of these attributes is provided in Table 3.1 under the headings of representativeness, ecosystem viability and ecological attributes.

Table 3.1 Criteria used for the Identification of Marine and Wetland Protected Areas

<table>
<thead>
<tr>
<th>Representative Attributes</th>
<th>Ecosystem Viability</th>
<th>Ecological Attributes</th>
</tr>
</thead>
<tbody>
<tr>
<td>• Comprehensiveness</td>
<td>• Integrity</td>
<td>• Productivity</td>
</tr>
<tr>
<td>• Adequacy</td>
<td>• Health</td>
<td>• Dependency - contains nursery or juvenile areas, or feeding, breeding or resting areas</td>
</tr>
<tr>
<td>• Representativeness</td>
<td>• Resilience</td>
<td>• Biological diversity and organisation</td>
</tr>
<tr>
<td>• Biogeographic importance</td>
<td></td>
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<tr>
<td>• Uniqueness</td>
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<tr>
<td>• Habitat Variety</td>
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<tr>
<td>• Naturalness</td>
<td></td>
<td></td>
</tr>
<tr>
<td>• International or National importance</td>
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</tbody>
</table>

Sources: Adapted from Salm and Clark (1984); Kelleher and Kenchington (1992); Australian Nature Conservation Agency (ANCA), (1996); Thackway (1996) and Environment Australia (1998b)

\(^1\) These aims are also in accord with the *National Strategy for the Conservation of Australia’s Biological Diversity* (Australian and New Zealand Environment and Conservation Council, 1996).

\(^2\) The sources of the criterion are presented in Appendix A.
3.2.1 Representative Attributes

The term representative attributes refers to the ecosystem attributes which are considered important for inclusion in a protected areas strategy (Usher, 1986). According to Brunckhorst (1994, 2000) the system of protected area management in Australia considers the representative attributes of comprehensiveness, adequacy and representativeness (CAR) foremost when establishing a sample of sites to be protected. Representativeness, as a subset of a broader group of representative attributes, can be defined as the degree to which an area represents a “habitat type, ecological process, biological community, physiographic feature or other natural characteristic” (Salm and Clark, 1984, 224). To ensure biodiversity is systematically represented, a number of biogeographic regions have been identified. For example, the marine and coastal regionalisation for Australia describes the environmental characteristics for 60 meso-scale regions (Cresswell and Thomas, 1997, 18). Areas that represent a biogeographic type will then be given priority for protection in the reserve system (Cresswell and Thomas, 1997). It is anticipated that the comprehensive, adequate and representative system for identifying protected areas within a bioregional framework will preserve biodiversity by protecting a representative sample of sites as an insurance against the loss of types of ecosystems (Kelleher et al., 1994).

Other representative attributes considered important when identifying protected areas are uniqueness, variety and naturalness of habitats. Areas that contain unique habitat or support unique species (Salm and Clark, 1984; Kelleher and Kenchington, 1992), and areas where a variety of habitats occurs within one ecosystem are considered valuable (Kelleher and Kenchington, 1992). For instance, the largest MPA in Australia, the Great Barrier Reef, is the largest single reef system in the world which makes the area unique (Kelleher et al., 1994). If it can be shown that an area is critical to a valuable species or more than one process or species then Salm and Clark (1984) suggest this area should be valued and hence protected. The naturalness of an area that has been free from human influence or not subject to human induced change is also considered valuable from a conservation perspective (Salm and Clark, 1984; Usher, 1986; Kelleher and Kenchington, 1992; Environment Australia, 1998a).
The value in protecting areas that are subject to national or international conservation agreements is also acknowledged (Kelleher and Kenchington, 1992; Environment Australia, 1998a). Further, areas that are or have the potential to be listed as World Heritage or Biosphere reserves are also considered for protection (Kelleher and Kenchington, 1992; Thackway, 1996; Environment Australia, 1998a).

Ecosystems chosen for protection under the auspices of representative attributes and specifically the CAR system are considered valuable in both the terrestrial and marine context. Representative attributes, however are human constructs based on notions of preserving variety, uniqueness and naturalness. These values are not based on the inherent ecological functions of the area under consideration. This is not to say that representative systems do not have ecological value, rather that it is the underlying ecological features of a system that provide ecological value. It is the identification of these features and not representative attributes that may lead to a definition of ecological value.

### 3.2.2 Ecosystem Viability

The second category in Table 3.1 includes attributes related to the continuing viability of the ecosystem, and includes the concepts of ecological health and integrity and resilience. The health and integrity of an ecosystem is considered important from a conservation perspective, as the more ecologically self-contained an area is the higher will be the probability that its values can be effectively protected (Salm and Clark, 1984). The resilience of the system according to Holling (1986) is related to its ability to maintain its structures and patterns of behaviour following disturbance. That is, the area is resilient to degradation from natural or human-induced events (Salm and Clark, 1984). Ecosystems that are resilient to such degradation will be the preferred candidates for protection. These concepts also include an evaluative component as the ability of ecosystems to maintain ecological integrity and health is seen as being socially desirable. Norton (1998, 362) contends that as health and integrity are indicators of both empirical information and agreed-upon social values and goals these terms are “normatively thick”.
Haskell et al., (1992) argue that an ecological system can be viewed as healthy if it is stable and sustainable, that is, it maintains its organisation and autonomy over time. The emphasis is on the system exhibiting organisation and productivity that is similar to a natural habitat of the region. The definition of ecosystem health will depend on which functions or components of the system are under consideration (De Leo and Levin 1997). The concept of health, according to De Leo and Levin (1997) and Hall (1999) is based on the notion that biological communities are structurally and functionally like organisms and that there is a definable ‘normal’ state that defines health. This view assumes that ecosystems are operating at or moving towards an equilibrium. However, Holling (1973; 1986) argues that ecosystems are seldom close to equilibrium. Thus, the concept of health provides a useful metaphor for management but reliance on the use of the ‘natural’ state as a baseline conditions ignores the complexity and dynamic nature of ecosystems (Hall, 1999). The concept does not provide information on the attributes of the system, which are considered valuable as its purpose is to define the ecosystem in terms of its variation from the ‘natural’ state.

De Leo and Levin (1997) argue that ecological integrity is a measure of ecosystem functioning and reflects the system’s capacity to support services of value to humans. This notion acknowledges that ecosystems are dynamic systems that exhibit characteristic patterns on a range of scales of time, space and organisational complexity. De Leo and Levin (1997) define integrity in terms of structural and functional characteristics. Structural definitions emphasise the individual species and population dynamics of species within isolated systems. A functional definition includes the concepts that the system is capable of supporting and maintaining a balanced, integrated, adaptive community of organisms and exhibits attributes that are comparable to that of the natural habitat of the region (Salm and Clark, 1984; Karr, 1992; Reid, 1994; De Leo and Levin, 1997). The two concepts are linked. For example, a system may be able to maintain a function such as primary productivity although there has been a structural change in species composition (De Leo and Levin, 1997). However, in the longer term structural changes may lessen the ability of system to adapt to disturbance (De Leo and Levin, 1997, Peterson et al., 1998).
The ability of an ecosystem to continue to act as a biologically functional self-organising ecological unit is of concern when assessing a site for protection (Kelleher and Kenchington, 1992; Thackway, 1996). This notion is related to the stability or resilience of the system. The concept of resilience had been defined in two different ways in the ecological literature (Holling and Meffe, 1996). Pimm (1991, 13) considers that one condition of stability, resilience, measures how fast a variable that has been displaced from equilibrium returns to it. This traditional view is contested by Holling (1973, 1986) who defines resilience as the magnitude of disturbance that can be absorbed before the system changes its state (Holling and Meffe, 1996). This view asserts that there may be more than one stability region or domain and therefore multiple equilibriums are possible (Holling, 1986). Also, ecosystem behaviour may be discontinuous when ecosystem variables move from one domain to another after reaching some threshold (Holling, 1986). This implies that ecosystems are not stable structures that return to the same stable state when perturbed but complex, adaptive systems that may move from one stable state to another. According to Holling and Meffe (1996) attempts to manage ecosystems for stability (for example, constant yields of fish) may in fact reduce resilience.

Karr (1992) argues that efforts to protect ecological integrity and health should be designed to protect attributes that support essential ecological services and long-standing earth processes. Further, the attributes used to assess ecological integrity and health should encompass both the structures and processes of ecological systems. The attributes used to assess ecological integrity for ocean systems include diversity, organisation and productivity as well as levels of variability (Environment Australia, 1998b). Holling and Meffe (1996) argue that levels of variability will be maintained by ensuring that management actions maintain the resilience of the system by allowing natural change rather than attempting to control them. These same attributes of diversity, organisation and productivity are listed under the heading of ecological attributes in Table 3.1. It is the measurement of these attributes that will determine if an ecosystem exhibits ecological integrity and health and resilience. It is therefore necessary to describe these ecological attributes to find the source of value.
3.2.3 Ecological Attributes

The third category of criteria used for the identification of protected areas relates specifically to the ecological attributes considered significant when identifying areas for protection rather than representative or viability attributes. These attributes are not, however, an assessment of the total conservation value of the area. According to Odum (1989) ecological attributes are important as they provide essential life support services. These ecological attributes have been derived from guidelines produced by Salm and Clark (1984), Kelleher and Kenchington (1992), Thackway (1996) and Environment Australia (1998b) to identify marine protected areas and wetlands. However, final selection will also be dependent on the area meeting social, economic, cultural criteria (Salm and Clark, 1984; Ballantine, 1991; Kelleher and Kenchington, 1992; Ray and McCormick-Ray, 1994; Johnson, 1995). When non-biological criteria are included it is difficult to determine whether a site was chosen because of ecological values or because of the contribution of other variables (Johnson, 1995).

The ecological attributes in the third category of Table 3.1 are examined in turn with examples from marine and wetland systems. The aim is to determine which, if any of the attributes should be included in the definition of ecological value. The possible means of measurement are also outlined.

Productivity

Productivity can be defined as “the rate at which biomass is produced per unit area, by any class of organism” (Begon et al., 1996, 996). It is the capture of solar energy and its conversion to chemical energy, termed primary productivity, which provides the basis for all further biological activity (Begon et al., 1996). Productivity is therefore considered important from an ecological viewpoint as the loss of this attribute could result in the loss of many if not all of the other services of the ecosystem (Bingham et al., 1995). For example, Salm and Clark (1984) and Kelleher et al., (1994) argue that productive areas may be important for the maintenance of marine biodiversity and ecosystem sustainability. Kelleher et al., (1994) state some coastal areas of high biomass production such as mangroves, salt marshes, seagrass beds, estuaries, coral reefs and kelp forests provide rich feeding,
breeding and nursery grounds for many and varied species. Without this attribute of productivity, the resources to support other life forms would not be available. It could therefore be argued that as primary productivity is the underlying process that supports the life of all components of the biosphere that this process is of "paramount importance for humanity" (Lieth and Whittaker, 1975, 203) and it should be included as an attribute of ecological value.

**Measurement of Productivity**

Net primary productivity can be measured as dry weight of biomass (gm⁻²yr⁻¹), grams of carbon (gCm⁻²yr⁻¹) or an energy equivalent per unit area per unit time (Kormondy, 1996). A difficulty in determining the productivity for a particular system is that high levels of productivity per se do not necessarily indicate a higher value as these may include eutrophic areas or areas that receive a high nutrient input from anthropogenic sources. Salm and Clark (1984) argue that it is the contribution of productivity to the sustainment of the ecosystem that needs to be measured.

**Dependency**

Dependency, in the context of ecological attributes is taken in this chapter to refer to the potential of an ecosystem to provide for individual organism’s needs for suitable areas for reproduction, feeding and resting. In marine environments, reproduction may in turn require separate nursery, juvenile and breeding areas (Kelleher et al., 1994). At an ecosystem level, the provision of habitats that act as areas for reproduction or for feeding and resting are essential for the continued existence of local populations of the species (Salm and Clark, 1984; Kelleher and Kenchington, 1992; Thackway, 1996; Environment Australia, 1998a). Furthermore, marine and coastal species such as marine mammals, birds, reptiles, adult fish and invertebrates may be widely scattered or have a large total range, but they often return to specific areas to spawn or breed (Kelleher et al., 1994). Examples of this dependency in marine areas include the mating and calving of whales and dugong, breeding colonies of seabirds and nesting beaches of sea turtles. Similarly, wetlands and shoals provide spawning and recruitment grounds for fish and invertebrates (Rapport, 1992; Kelleher et al., 1994). The importance of wetlands for waterfowl as feeding,
resting, nesting and brood rearing areas have been recognised by management agencies (Reimold, 1994).

Fairweather and McNeill (1993) argue that it is essential that areas critical to the life history of local populations of species are maintained if the species are to continue existing. These critical areas on which local species are dependent could therefore be considered valuable as the total loss of these habitats has the potential to severely curtail the reproduction or survival of these local populations. As such the provision of dependency by an ecosystem should be considered an ecologically valuable attribute of that system.

Measurement of Dependency
Dependency for a particular species can be measured by the ratio of numbers of the local population of the species being measured that are dependent on the specific ecosystem being valued to the total number of the local population of the species being measured. Begon et al., (1996) defines a local population as the group of individuals of one species found within the defined area of study4.

Where information is available the ratio could be calculated for all known dependent species for the particular ecosystems under investigation. The higher the number of local populations of species that are dependent on an area the greater will be the value afforded to this attribute. However, the ecological information on the species present in an ecosystem may be unknown. In this case dependency may be measured by considering the local populations of commercial or endangered species dependent on the area, as more information is usually available about such species. Holling and Meffe (1996) suggest that there may be a subset of species that controls the dynamics and functions of ecosystems. However, Levin (1999) notes that identifying functional groups of species is not an easy task. Nevertheless, where functional groups of dependent species can be identified, the dependency formula would only need to include those groups. A similar approach is adopted when identifying sites

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3 Global averages for the net primary productivity of the major ecosystems have been estimated by Lieth and Whittaker (1975) but more detailed information would be required if systems within one of these regions were to be compared.

4Geneticists may define a population as “those individuals of a species that are close enough to each other for there to be at least occasional mating between them” (Begon et al., 1996, 569).
to be included on the list of *Wetlands of International Importance*. For example, a site is considered important if it “supports 1% of the individuals in a population of one species or subspecies of waterfowl” (Australian Nature Conservation Agency, 1996, 17).

**Biological Diversity and Organisation**

The third ecological attribute in Table 3.1 is biological diversity and organisation. According to Primack (1995) and Environment Australia (1998a) areas that sustain biological diversity (biodiversity) are considered valuable from a conservation perspective and this aim has become a primary goal in establishing protected areas. Environment Australia, (1998a, 16) defines biodiversity as “the variety of life forms: the different plant, animals and micro-organisms, the genes they contain and the ecosystems they form”. Species based approaches for biodiversity conservation are most commonly used and emphasise the value of individual species, usually by the presence of rare or endangered species (Johnson, 1995). Biological diversity may have intrinsic value but more important from an instrumental perspective is the interaction and transfer of energy and material between the variety of species that in turn provides essential services such as food production and waste assimilation. That is, while biodiversity has conservation value it is the organisation and function of biodiversity that has ecological value.

The species diversity of an area would be expected to increase in areas not subject to unpredictable changes and this is likely to be matched by a parallel increase in productivity and structural complexity (Putman, 1994). Pimm (1991) argues that this complexity or organisation can be thought of as the diversity of energy exchange pathways within the food web. Both Agardy (1994) and Ray (1988) support the view that it is the role that species play in the working of the system as a whole rather than the abundance of species that gives an ecosystem value. Further, as Agardy (1994) observes, it is the processes that maintain complex ecosystems and their immense variety of life forms that are important. For this reason, it is the organisation of the interactions of species, which continue to transform matter and energy into a form that is of benefit to humanity that should be included as an attribute of ecological value. This organisation is in turn reliant on primary productivity of the system and the provision of habitat critical for the life history of
the interacting species. Together these ecological attributes of productivity, dependency and organisation can lead to a definition of ecological value.

**Measurement of Biological Diversity and Organisation**

To measure biodiversity, diversity indices have been developed to produce mathematical indices of species richness and abundance in a community. The two most commonly used indices are the Simpson’s Diversity Index (D) and the Shannon Diversity Index (H) which take into account richness and abundance of species (Begon *et al.*, 1996). However, as noted above, it is not just the number and abundance of species that is important for ecosystem functioning but the way the species interact through energy and material exchanges.

These interactions can be measured through the process of network analysis. Network analysis is a statistical tool for analysing flows of energy or matter among systems components, which can be applied to ecology (Field *et al.*, 1989). Ulanowicz (1992) has encapsulated these networks of trophic interactions into a network ascendancy index. The ascendancy index is used to represent both the size and the organisation of the flows of energy or material into a single measure (Field *et al.*, 1989). Ascendancy is then the product of the total system throughput and the diversity of flows in a system (Field *et al.*, 1989). Ulanowicz (1992) suggests that as ecosystems mature, the ascendancy of the system should increase. The corollary of this suggests that if a system is disturbed this will be reflected by changes in the flows of energy or materials and the system’s ascendancy will also alter (Ulanowicz, 1992).

However, developing an ascendancy index requires data on all transfers occurring within a community. As a result, Ulanowicz (1992) states that fully quantified networks of ecosystems remain scarce. Given that in most circumstances information on species diversity is limited, as are the trophic interactions between species, measures of diversity and organisation will be determined by data availability.

The criteria used for the identification of protected areas have been categorised and reviewed to determine which are specifically related to the purpose of conservation
and those that are directly related to the ecological attributes of an area. The categories of representativeness and viability, although important, are not considered to relate specifically to the underlying ecological features of the ecosystem. The ecological attributes distinguished through this process are productivity, dependency, diversity and organisation. Methods for measuring these attributes have been suggested to aid in determining the appropriate value to assign to ecosystems possessing these characteristics. It is argued that the combination of these attributes can lead to a definition of ecological value and possibly a means of measuring this value.

3.3 A DEFINITION OF ECOLOGICAL VALUE

The attributes of ecological value identified in this chapter have been derived from those used to select the protected areas as presented in Table 3.1. The values identified in columns 1 and 2 are related specifically to conservation goals such as representativeness, uniqueness, naturalness and ecological integrity and health. These values are in turn dependent on the ecological attributes listed in column 3. Each of these attributes (productivity, dependency and organisation) has been examined to determine its contribution to ecological value. It has been argued that each of these attributes contributes to the continuing functioning of ecosystems through the capture and conversion of energy through food webs, via the interactions of species which have been provided with habitat to support their critical life stages. These attributes ultimately provide instrumental benefits for humans. Therefore, a system’s ecological value can be determined from and defined as its productivity, ability to provide habitats for its dependent species and the diversity of species and organisation it supports. This definition of ecological value reflects the requirements of Agenda 21\(^5\), Chapter 17 which requires that states should identify marine ecosystems exhibiting high levels of biodiversity and productivity and other critical habitat areas (Robinson, 1992). That is, it has been accepted at the international level that systems which exhibit these attributes of ecological value are considered valuable.

\(^5\) Agenda 21 was an outcome of the United Nations Conference on Environment and Development held in Rio de Janeiro in 1992. Chapter 17 outlines a program for sustainable development for marine and coastal areas. (Herriman et al., 1997).
3.4 ASSESSMENT OF ECOLOGICAL VALUE

As a result of the lack of a universal ecological value theory a consensus on a method for assessing the value of ecosystem and hence ranking them has not been developed. As Margules et al., (1988) assert, agreement on any given conservation assessment or on any index is unlikely. This is because it is difficult to remove personal bias from even the most complex of ranking systems (Salm and Clark, 1984). Similarly, Johnson (1995) states that there is no universally accepted scheme for establishing biodiversity priorities and Fairweather and McNeill (1993) argue that it is impossible to be entirely objective in choosing one area over others as being worthy of protection.

While it may be difficult to retain objectivity when placing differing values on ecological systems the task is still necessary if ecological value is to be incorporated into decision making. The purpose in developing an index of ecological value is to ensure that all the benefits provided by ecosystems are captured in resource allocation decisions. The intention of an index of ecological value is not to rank all the ecosystems of the biosphere. However, from the definition of ecological value it may be possible to develop an index which can rank similar ecosystems within a defined spatial scale. The attributes to be measured have been derived from a consensus on ecological attributes considered important from a conservation perspective. The relative value of ecosystems can be assessed if a method can be devised to value and integrate the attributes. A review of similar indices is given to determine if such a task is presently possible.

3.4.1 Indices of Integrity and Health

Although a method for ranking ecosystems based on their value has not yet been formulated there have been some attempts to measure other indicators of ecosystems such as ecological integrity and health. Karr et al., (1986) developed one approach for assessing the biological integrity of water resources, which is now widely used in North America and Europe. This Index of Biological Integrity (IBI) is an integration of twelve attributes of a segment of the biota (fishes) of a stream (Karr, 1992). By
incorporating data about the entire fish community twelve metrics are measured in
three categories (Karr et al., 1986). The value of each metric is compared to a
similar site which is considered to have had minimal human interference (Karr et al.,
1986). Values are given depending on how the assessed value deviates from this
standard (Karr et al., 1986). The values are then totalled and IBI scores assigned
(Karr et al., 1986). Karr (1992) considers that the use of the IBI is applicable for a
diversity of geographical areas and habitat types for a variety of taxa. While this
index provides valuable information on the integrity of water resources, it only
measures one group of animals, fish. It takes no account of the total diversity and
organisation of all species in the stream, nor does it measure productivity. That is, it
is not able to measure ecological value as it has been defined in this chapter.

Costanza (1992) has proposed that a Health Index (HI) can be formed by considering
a system’s vigour, organisation and resilience. This can be written as:

\[ HI = V \times O \times R \]

where:

- HI  is system health index
- V  is system vigour, a cardinal measure of system activity, metabolism or
  primary productivity
- O  is system organisation index, a 0-1 index of the relative degree of the
  system’s organisation, including its diversity and connectivity
- R  is system resilience index, a 0-1 index of the relative degree of the system’s
  resilience (Costanza, 1992, 248-249).

The concepts of vigour and organisation used in the health index are similar to the
ecological attributes included in ecological value. Resilience, which may be a
function of the underlying ecological attributes, is however considered to be an
attribute of viability. Costanza (1992) does not, however, provide any guidance on
the operational difficulties of measuring, combining and weighting the attributes or
the appropriate spatial scale for the assessment. This index thus provides a
theoretical construct for valuing the health of ecosystems but its application is not
demonstrated.
The purpose of developing an index of ecosystem health is to determine the change in one particular ecosystem over time. This concept is essential to determine if the management of the system is sustainable. However, this differs from the purpose of developing an index of ecological value, which is to include all the instrumental benefits of ecosystem in resource allocation decisions. As stated this may occur when a choice is to made between two systems. When the purpose of assessing ecological value is to determine the difference in ecological value resulting from some change in use of the ecosystem, the change in value may also reflect a change in the health, integrity or resilience of the system. This results from the fact that a system’s viability is determined by the underlying ecological attributes.

### 3.5 INDEX OF ECOLOGICAL VALUE

The definition of ecological value derived from the review of criteria used to identify protected areas included the ecological attributes of productivity, dependency and diversity and organisation. In order to determine the ecological value of particular ecosystems these values need to be combined in some fashion. This in turn requires encapsulating these attributes within an index. An index of ecological value would therefore need to capture the functional form of the attributes productivity, dependency and organisation.\(^6\) The form of such an index of ecological value could be \(IEV = f(P,D,O)\)

Where:

- \(IEV\) is the index of ecological value
- \(P\) is a function for the productivity of the system
- \(D\) is a function for the dependency of the system
- \(O\) is a function for the organisation of the system

using the definitions and methods of measurement as outlined.

Theoretically, the development of an index of ecological value would require choosing the appropriate spatial and temporal scales and setting the criteria and reference standards for each of the attributes: productivity, dependency and organisation as they have been defined. Following the measurement of the attribute,

\(^6\) Usher (1986) attempted to design an index of conservation value using a similar form.
the deviation from the reference standard would be determined to produce a value score. These scores would then need to be weighted and combined in some form to indicate the importance and interactions of the attributes to give a total value for the system.

For example, Odum (1978) argues that a mangrove/seagrass wetland ecosystem which is relatively unaltered by human influence would exhibit high productivity. The productivity of the wetlands makes them important habitat for a number of invertebrates, other vertebrates and fish communities (Mitsch et al., 1994). Reimold, (1994) for example estimated that over 90% of all species of fish that live in the coastal zone depend on wetlands in some way for their feeding and reproductive habitat. The dependency on wetland habitat for waterfowl, for feeding, resting, nesting and brood rearing areas have been recognised by management agencies (Reimold, 1994). It is also known that wetlands support a high diversity of flora and fauna (Williams, 1990a). A wetland that exhibited these attributes would be considered to have a high ecological value.

A mangrove/seagrass wetland ecosystem degraded through the accumulation of pollutants from runoff may not display the same ecological values. Excessive nutrients and sedimentation can increase mangrove mortality (Mitsch and Gosselink, 2000) and increase the growth of epiphytes leading to seagrass mortality (Shepherd et al., 1989). Increased turbidity resulting from sedimentation will also reduce the maximum depth of seagrass growth range and hence the area of seagrass (Abal et al., 1998). The resultant loss of plant biomass may then reduce primary productivity and reduce the structural support for dependent species. This may then further impact on the diversity of species within the ecosystem. The degraded wetland system will exhibit a lower ecological value and hence provides fewer services for human benefit.

The flow of benefits and services from the degraded wetland will also be reduced in comparison to the less disturbed systems. It is recognised that wetlands provide human benefits such as fish, fuelwood, recreation, water transport, flood control, 

\[7\] Excessive sedimentation may also increase available habitat and growth of mangroves.
storm prevention, external support for offshore fisheries and the existence value of endangered species (Barbier, 1993; Barbier et al., 1994). In the degraded system there will be a loss of fisheries and flood control and storm prevention will be reduced due to the reduced biomass and recreational values may be impaired as water quality drops below safe standards. Endangered species that are dependent on seagrass such as dugongs and green turtles may also be impacted. However, assessment of the economic value may only include those goods and services produced on-site which are characterised by market prices and will thus underestimate the full value of the wetland ecosystem (Barbier et al., 1997). The loss of ecological value will not be included in such calculations. However, the application of an index of ecological value may indicate that protecting the wetland (and hence preserving the full range of goods and services provided by the wetland) will ultimately provide greater benefits to humans. That is, by considering information on the ecological value of the system, the full benefits of the wetlands will be included.

3.5.1 Spatial and Temporal Scale and the Reference Standard

When assessing ecological value the chosen extent of the spatial scale will depend on the purpose of the ecosystem assessment. It is assumed that the ecosystems being compared will lie within a region that has distinct biological and physical characteristics. For example, in Australia coastal and marine areas have been categorised into 60 different bioregions and it would be expected that comparison of systems would take place within this meso-scale (100’s km) (Cresswell and Thomas, 1997, 18). At this regional level it is expected that differences of assessed value will be a result of the differences of the specific ecosystems under examination and not from general climatic or geographical conditions. In some instances, this meso-scale may be too small in terms of the number of comparative ecosystems found within the region. In these cases, the spatial extent of the scale of large marine ecosystems (LMEs) may be more appropriate.

Forty-nine large marine ecosystems (LMEs) have been identified globally and they are 200,000km² or larger in size (Sherman, 1994). The criteria used to delimit these areas geographically are bathymetry, hydrography, productivity and trophically
dependent populations (Sherman, 1993). However, only two LMEs have been identified for Australia, the Northwestern Australian Shelf and the Great Barrier Reef which limits the application of this spatial scale for Australia (Sherman, 1994). Sherman (1994) also notes that following adaptations to Costanza’s (1992) Health Index and Karr’s et al., (1986) Index of Biological Integrity methods are being developed to measure the ecosystem health of LMEs. While these studies are being undertaken to determine changes over time, the resultant information on the diversity, stability and productivity of these systems may allow comparisons between ecosystems.

It is proposed that each ecosystem would need to be assessed with respect to a reference standard. The standard chosen could be the maximum expected value for the attribute for the given bioregion or LME. This contrasts with the reference standard set for indices of integrity and health where the attributes of systems considered to be in a natural state are used as reference (Karr et al., 1986). When assessing ecosystem health a variety of sites subjected to known stresses is used as reference rather than waiting the hundreds of years that would be required to determine what is normal (Whitford, 1998). The index of ecological value is not designed to test if an ecosystem is in a natural or disturbed state nor the state of its health but to compare the attributes of systems. Scoring would be based on deviation from the maximum expected value. It is proposed that scores be given in the range of 0 to 1, following Odum’s (1978) approach for wetland assessments. A score of 1 would be given where the value of the attribute of the system to be ranked is equal to the maximum expected value at a regional level. Initially each attribute could be given equal weight within the assessment.

The scale of assessment, both spatial and temporal, would ideally consider that ecosystems are nested hierarchically across spatial scales and are dynamic, evolving and non-linear systems subject to potential thresholds and discontinuities over time (Holling, 1986; Peterson et al., 1998; Levin, 1999). Holling et al., (1995) argue that the range of services and benefits produced by ecosystems will also extend across spatial or temporal scales. Resource management based on producing human benefits however, often operates in isolation of spatial and functional organisation and may ignore the broader ecological context in which the exploited resources may
be embedded (De Leo and Levin, 1997). For the owner or manager of a resource, the benefits which accrue to the larger population across time may be ignored as these benefits lie outside their jurisdictional boundaries. The temporal scale of analysis is also important because an ecosystem that is considered valuable today in terms of the ability to produce commercial goods may quite quickly flip into another stability domain. For example, Holling and Meffe (1996) argue that management actions and resource exploitation can lead to collapses in fisheries, eutrophication of water bodies and the transformation of savannas into shrub-dominated semideserts. Ecosystem management based on the primary value of the ecosystem itself, as defined by ecological value, may avoid these errors as the focus moves from a single resource at the local level to the multifunctional features of the ecosystem that provide instrumental human benefits over wider scales.

### 3.5.2 Problems with application of an index of ecological value

Theoretically, an index of ecological value has identified the attributes of ecological value, a method of measuring each attribute, an appropriate spatial and temporal scale for undertaking comparative assessments and a reference standard. The application of an index of ecological value then requires an understanding of the relationships between the attributes so that they can be integrated without double counting. However, the ecological relationships between attributes are not well delineated. For example, it has been noted previously, productive wetland systems are also likely to have a greater diversity of species. Begon *et al.*, (1996, 888) notes that in general species richness is likely to increase with increases in productivity but that this is not universal. This is exemplified by the case of cultural eutrophication of aquatic systems where diversity is reduced with increased productivity (Begon *et al.*, 1996). Experimental work undertaken by Naeem *et al.*, (1994), Tilman *et al.*, (1996) and Loreau (1998) in examining the relationship between diversity, ecosystem functioning and productivity failed to produce conclusive results. Loreau (1998) argues that there is a lack of adequate theories and models for generalisations to be made about the relationship between the attributes. Until the complexity of the interrelationships of the attributes is more fully understood, an index of ecological value will not produce a useful tool for decision making.
A further problem with the application of an index, which has been alluded to in the discussion of other indices, is the problem of weighting the variables. Usher (1986) reports that previous investigation of weights given in conservation assessments has varied greatly for each evaluator. A similar situation is likely to arise when assessing the ecological value of a site. The literature is not able to provide guidance on the appropriate weightings to be attached to the attributes.

Another problem to consider when implementing an index-based approach is the availability of data. The range of values of primary productivity may be known for the major types of ecosystems of the world (Lieth and Whittaker, 1975) but are not known for each and every ecosystem. Similarly, the dependency of some particular species may be known for an area (in particular the endangered charismatic megafauna) but again detail studies for all species is lacking. Studies of the organisation of systems using network analysis are very limited and while some systems may have been studied for species richness, the abundance is not always known. In this case, it is not possible to value diversity using a diversity index nor is it possible to compare diversity between systems.

Usher (1986, 43) noted that the goal of producing a conservation index “is probably impossible to achieve” for the reasons listed above. These problems also apply to any proposed index of ecological value. However, with increases in ecological knowledge in the areas of data availability and the functional form of the interrelationships between attributes these problems can be overcome. An index of ecological value, as it has been developed, could then be used to compare ecosystems and their uses to provide information for decision-makers.

3.6 CONCLUSION

The need to improve environmental management through the inclusion of ecological values in natural area assessment has been identified. A clear definition of ecological value was not able to be distinguished from the literature but was derived from considering the ecological criteria used for the identification of protected areas. Indices of integrity and health were reviewed to aid in the development of an index of ecological value. The attributes of productivity, dependency and organisation
were found to reflect the essential ecological process and life support services provided by ecosystems and formed the bases of an index of ecological value. It was then possible to delineate the spatial and temporal scale and the reference standard that could be used when applying an index of ecological value. However, due to the present limits of ecological knowledge, it was not possible to determine the interrelatedness of the attributes. Thus, due to the current limitations ecological methods cannot provide a means of incorporating ecological and economic values.

The definition of ecological value does, however, provide an assessment of those attributes that should be included when undertaking ecosystem valuations. Although full information may not be available, it may still be possible to identify some of the attributes of ecological value for a particular ecosystem. If these values can be determined then it may be possible to include them explicitly in economic analysis and thus link ecological and economic value. A review of economic methods that may be able to capture ecological value is given in Chapter 4.
CHAPTER 4
ECONOMIC INSTRUMENTS AND THEIR ABILITY TO CAPTURE ECOLOGICAL VALUE

The purpose of this chapter is to evaluate the economic techniques commonly used to assess environmental benefits, with respect to their ability to capture ecological value. This task is undertaken as it was not possible to capture the economic benefits derived from ecosystems using an ecological approach. However, the ability of economic techniques to capture ecological value has not been directly addressed within the economic literature. To overcome this deficiency this chapter presents four criteria to assess the ability of techniques to measure ecological attributes. Three approaches for assessing non-market values, (direct linkage models, revealed preferences and stated preference models) are discussed with respect to their ability to assess ecological value. An alternative biophysical approach, namely an energy analysis, is also considered. Wetlands are considered as an example of an ecosystem exhibiting ecological values throughout the chapter. The review of these methods provides a possible approach for measuring ecological value using the economic method of contingent valuation.

4.1 CRITERIA TO ASSESS THE ABILITY OF ECONOMIC TECHNIQUES TO MEASURE ECOLOGICAL VALUE

Ecological value as it has been defined in Chapter 3 includes the attributes of productivity, provision of dependent habitat and biodiversity and organisation. These attributes will be present in an ecosystem, such as a wetland, to the extent provided by the interaction of the structure, processes and functions within the system. In some cases, it may be possible to value directly the output of functions such as commercial fisheries. However, most of the outputs of wetlands are not traded on markets, and thus these ecological services take on public good characteristics (Bishop and Woodward, 1995). Given this limitation, non-market techniques must be used to measure the value of these outputs.

The predominant methods of non-market valuation are direct linkage models, revealed preferences and stated preference models (Garrod and Willis, 1999). As
ecological value has been poorly defined within the literature, these criteria have
been developed specifically for this task. To guide the development of the criteria,
techniques which attempt to measure changes in the quality of environmental
amenity have been reviewed (for example, Kerry Smith and Desvousges, 1985,
Wilson and Carpenter, 1999). Studies that assess the value of changes in quality of
environmental amenities assume that environmental quality provides utility for the
consumer in the same way as an increase in a quantity of a marketed good (Bishop
and Woodward, 1995). This same assumption would need to hold for ecological
value, so that the ecological value of an ecosystem, such as a wetland, could be
measured by the increase (decrease) in utility as a result on an increase (decrease) in
ecological value.

Criteria 1
The first criterion for assessing if non-market economic techniques can capture the
ecological value is to determine if a change in the ecological value of a wetland is
reflected in willingness to pay (or the marginal implicit price) for the marketed good
linked to the wetland or for the preservation of the wetland itself. This is considered
important because the concern in economics is to measure changes in value at the
margin. Therefore, the technique must exhibit the ability to measure changes. The
question is are changes in the underlying ecological value actually reflected in
changes of value of the good being measured or are changes in value related to other
measurement factors? It is considered that methods that reflect changes in ecological
value through the good being valued are preferred.

Criteria 2
The second criterion assesses how many of the attributes of ecological value can be
linked to the production of the good being assessed. The greater the number of
attributes that can be linked to the good the greater will be the ability of the method
to capture ecological value. Further, as ecological value results from the interactions
of the attributes of value, methods that reflect this interdependence will be preferred.

Criteria 3
The third criterion is related to how closely the good that is being measured is linked
to ecological value. Consideration is given to the number of steps or links that need
to be taken from the good being assessed and the underlying ecological value of the
good. Revealed preference models by their nature will always be at least one link removed from the wetland. The fewer the steps or links the more likely that information on the wetland function and hence ecological value is reflected in the value of the good.

Criteria 4
The fourth criterion is the number of components of total economic value that can be assessed through the technique. It is thought that the ability of techniques to capture non-use value is important as ecological value provides for both use and non-use instrumental benefits.

4.2 ECONOMIC METHODS AND THE EVALUATION OF ECOLOGICAL VALUE

Value in economics is derived from the preferences of individuals for goods and services (Peterson et al., 1990). Goulder and Kennedy (1997) argue that if a good or service provides utility for the individual and they are willing to give up some other resource to obtain the good then it will have an economic value. For marketed goods with no public values, the inverse demand curve represents the marginal willingness to pay function (Anderson and Bishop, 1986). The total willingness to pay for a certain quantity of a good is then equal to the area under the inverse Marshallian demand curve. The net benefit is equal to the area between the inverse demand function and the marginal cost function (Kahn, 1995). However, in the case of environmental goods which have public good attributes there is no market price with which to estimate a demand curve. Also, as Kahn (1995) points out even if a market price (or proxy market price) is known the area under the inverse demand curve only represents the private benefits of the good and excludes the public benefits.

Therefore, to derive a value for the private benefits for non-market goods requires estimating a demand function for the non-market good to derive a total willingness to pay, or by directly estimating the total willingness to pay either for the good or discrete changes in the quantity or quality of that good (Kahn, 1995). Revealed preference models use the former approach based on estimating demand curves from

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1 The details of the total economic framework and examples with respect to wetlands have been provided in Chapter 2 and Figure 2.3.
observed behaviour (Anderson and Bishop, 1986). Stated preference models usually use hypothetical surveys to directly measure net benefits by estimating the Hicksian compensated demand curves (Mitchell and Carson, 1989).

With respect to measuring aspects of the total economic value of wetland benefits, Figure 4.1 expands on Figure 2.3 to include a range of techniques that have been used for wetland valuation\(^2\). However, Figure 4.1 illustrates that specific techniques may only able to measure specific components of economic value. Thus, the use of any one technique may only provide information on the partial value of the wetland.

The main methods of valuation of direct and indirect components of economic value can be undertaken through market analysis, physical linkage models, hedonic pricing and the travel cost method. In essence, each of these methods is linked to the observed behaviour of consumers. However, for the purposes of the discussion physical linkages models are examined separately from the observed market approaches of hedonic pricing and travel cost models. Estimation of non-use values and option values, which by their nature have no markets, can be assessed through the stated preference techniques of choice modelling and contingent valuation. All these approaches are discussed in terms of what would be the necessary conditions that would need to be fulfilled for each of the criteria as discussed in Section 4.1. An alternative biophysical approach, energy analysis, is also reviewed with respect to the criteria. The focus of the discussion is to assess if non-market techniques can capture ecological value. Each method has its own theoretical and empirical advantages and limitations but only those limitations which relate to this purpose are discussed.

\(^2\) The figure also includes examples of the various goods and services provided by wetlands as described in Chapter 2.
4.3 PHYSICAL LINKAGE MODELS

The value of environmental resources can be inferred from the direct purchase of marketed goods using the physical linkage models. Physical linkage models include methods which are referred to as production functions, damage functions or dose responses to determine value. Using this approach a natural resource, such as a wetland, is treated as an input into the production of goods such as fisheries products (Barbier, 2000). The economic impact of changes in the supply of the input on the
production of the good can then be estimated. This approach has been used to assess the value of coastal wetlands for supporting marine fisheries and other functions in the Gulf Coast fisheries in the US (Lynne et al., 1981; Farber and Costanza, 1987, Bell, 1997). Ruitenbeek (1994) undertook a related approach which included determining the value of multiple use mangroves under different management options in Bintuni Bay, Indonesia. A similar approach has been used by Kahn and Kemp (1985) to examine the effects of pollution on habitat-fisheries linkages. The first of these studies was undertaken by Lynne et al., (1981) who linked wetlands and human effort to blue crab fisheries production in Florida. This allowed the marginal value product of the wetland acreage to be determined.

However, as Lynne et al., (1981) note changes in the price of fish in the market place on which the marginal value of the wetlands are assessed may not necessarily reflect changes in the value of wetland on which the fish are dependent. For example, Lynne et al., (1981) demonstrated that the marginal value productivity of wetlands varied with alternative levels of wetlands and effort in the fishery. However, the change in the availability of wetlands produced a smaller response in terms of marginal product than did a change in effort. Thus, a limitation with using a direct linkage method is the ability to separate out the effects of human effort. Two more recent studies in Thailand and Mexico reported by Barbier (2000) demonstrated that valuing the support of mangroves for offshore fisheries was significantly affected by whether the fisheries are managed or subject to open access. These studies also illustrated that the value of wetlands for fishery support may be influenced more by the management of the fisheries than by changes in the area of wetlands. Morton (1990) avoided this problem in estimating the value of wetlands for fish production in Moreton Bay as he directly estimated the number and biomass of fish found in the wetlands. However, this method is also limited as the average value cannot be used to estimate the marginal change in fisheries production due to a decrease (increase) in wetland area as required for economic theory. However, this study could be replicated over time and related to changes in wetland area to provide an estimate of the marginal changes in productivity.

Further limitations to this model are also imposed when one system produces multiple outputs, as is the case for multifunctional systems such as wetlands. To
apply direct linkage models in this case requires that the relationship between function and economic activity is well understood (Barbier, 2000). Thus, for multiple use systems (such as wetlands) applications of this approach will be more problematic as ecological relationships need to be carefully constructed (Barbier, 2000). The change in the value of output of wetland due to a decrease in wetland size will not only affect fish but many also affect other services such as flood control and waste assimilation. Excluding the other services of wetlands from the analysis limits the usefulness of the method to fully reflect the changes in ecological value that may be produced by a change in wetland size. On the other hand, including the value of all services may result in double counting of the benefits, as some may be mutually exclusive (Barbier, 2000; Turner et al., 2000).

Nevertheless, as discussed in Chapter 2, the approach does provide a method for estimating the link between an ecological function and the value of the goods and services produced. There is only one link between wetland values and fish values assuming the costs of capture and marketing are reflected in market price. However, one more step is required to link wetland functions to ecological value. In terms of the criterion 2, the attributes related to the production of fish may encompass all the attributes of ecological value. That is, fishery production in wetlands is related firstly to the overall productivity of the wetland. Some fish are also dependent on the wetland at some part of their life cycle and are dependent on the existence of other species in the food chain for their own survival. So, if the value of the wetland is related to fishery production all the attributes of ecological value are included in that value. This value will also reflect the interdependence of the attributes. A major limitation of the model in terms of criterion 4 is that direct linkage models only have the ability to measure use values which can be linked to the environment.

In summary, if it were possible to assess the marginal value product for all wetland functions (with an understanding of the interdependence of these functions), and if the effects of human effort can be excluded from the analysis then direct physical linkage models may be able to capture the underlying ecological value of the system. The model directly links the wetland with output and may encompass all the attributes of ecological value. The model is, however, limited to measuring use values. Until further ecological information is available on the ecological
relationships between function and output physical linkage methods will be limited in their ability to capture ecological value.

4.4 OBSERVED MARKET APPROACHES

The advantage of using observed market approaches is that information can be obtained for the demand for a good, which is associated with the environmental good to be valued (Garrod and Willis, 1999). The hedonic pricing model, which most often uses the housing market to determine demand for some aspect of environmental quality and the travel cost method which uses travel as the observed purchase in the production of recreation are both observed market approaches. These two methods are outlined to determine their ability to capture ecological value. As both the methods utilise a link for the demand of the marketed product there are two conditions which are assumed to hold. The first is that the representative individual is assumed to have a utility function that is weakly separable (Hanley and Spash, 1993). In general, this assumption means that the marginal rate of substitution between good a and good b in the individual utility function is independent of the quantities of all other goods (Hanley and Spash, 1993). The second assumption is one of weak complementarity. For this condition to hold the demand for the market good and the environmental good must be at least weak complements (Mäler, 1974). This condition will exist if for a certain range of prices for the private good, the demand price for the environmental service is zero. If utility is derived from simply knowing the resource exists then this condition will not hold (Mäler, 1974). It follows that revealed preference models can only capture the use values of a resource.

4.4.1 The Hedonic Pricing Model

The term hedonic is derived from the Greek word, *hedonidkos*, meaning pleasure. In economic terms this pleasure is defined as the utility that is derived from the consumption of the goods and services (Streeting, 1990). Hedonic pricing techniques have been applied to studies related to air pollution, noise levels, location with respect to water frontage, water quality and wage differentials (Anderson and Bishop, 1986; Pearce *et al.*, 1989; Streeting, 1990; Lansford and Jones, 1995). Most common uses are in property value studies in which the price of the property is
thought to include the value of some environmental service (Hoevenagel, 1994). The differential of housing prices with respect to the environmental amenity allows estimation of the demand for the amenity.

To estimate ecological value using this technique would first require that there is a relationship between the attributes of ecological value in the neighbouring area and the price of a good (such as housing) for which a market exists. The observed market for housing is thus implicitly assumed to be motivated by the unobservable ecological values of the area (among other attributes) (Wilson and Carpenter, 1999). If two properties adjacent to wetlands are identical in all respects except for the ecological values of the wetlands then the differential between the housing prices is the implicit price paid for the difference in ecological value. That is, the technique is capable of measuring changes in ecological value. To provide an accurate assessment of the value it must by assumed that ecological value is perceived by house buyers (Hoevenagel, 1994). However, it is unknown if house buyers do perceive this difference (Mahan et al., 2000). The difference in property values adjacent to wetlands of differing ecological value may be related to other factors not included in the model. To ascertain that property buyers perceive ecological value would require undertaking a further survey of buyers’ attitudes. Relying on market data (as is the usual case for hedonic pricing) would not be sufficient to make this assumption.

To assess ecological value using the hedonic pricing technique would require including ecological value as an explanatory. For example, the price of a house fronting a wetland \( P_i \) will depend on several regressors such as site characteristics \( S_i \) (house size, lot size, distance from wetland), neighbourhood characteristics \( N_i \) (ethnic composition, schools etc) and ecological attributes \( E_i \) (productivity, dependency, diversity and organisation). A regression is then run to predict price from these variates:

\[
P_i = f(S_i, N_i, E_i)
\]  

From equation 4.1, the marginal value (or implicit price function) of ecological value
is estimated as $dP/dE$. A second stage of analysis is required to determine the marginal willingness to pay function. This requires combining the quantity and implicit price information (Freeman, 1995). The marginal willingness to pay function gives the maximum willingness to pay for an increase in $E_j$ holding utility constant and given the optimally chosen level of all other characteristics. If this function can be identified, then it can be used to estimate the change in welfare for an individual resulting from changes in ecological value, assuming other things are held equal (Freeman, 1995). However, Kerry Smith (1997, 167) controversially argues that “to [his] knowledge, no study has successfully estimated the WTP function as a second-stage model derived from the hedonic price functions”. Further, Mahan et al., (2000) note that willingness to pay functions for water resources have not been estimated using hedonic pricing due to the serious identification problem in estimating the function.

To apply the hedonic technique requires that housing is located adjacent to wetlands. Thus, the wetlands to be valued needs to be within an area that is used for residential dwellings. However, as Doss and Taff (1996) and Mahan et al., (2000) point out that while urban wetlands may provide amenities such as open space and the opportunity to watch wildlife, they also exhibit disamenities such as nuisance animals, insects and odours. Doss and Taff (1996) have utilised the hedonic technique to estimate the value of differing types of freshwater wetlands. Their results indicated that the implicit price for living an additional ten metres closer to a forested wetland is negative (-US$145) (Doss and Taff, 1996, 127). However, the implicit price was positive for wetlands, which exhibited a greater amount of open water. There is no information in the Doss and Taff (1996) study that indicates which wetland type would have high ecological value. Also it is noted that the hedonic technique only provides a lower bound for wetland values as it does not include public values such as water quality improvements, biodiversity, ground water recharge and recreation which may differ for the differing types of wetlands (Doss and Taff, 1996: Mahan et al., 2000). That is, the technique cannot capture the attributes of ecological value which provide benefits for society at large.

Similarly, Mahan et al., (2000) replicated the Doss and Taff (1996) study in Portland, Oregon and in contrast found that the marginal implicit price for reducing the
distance to the nearest wetland by 1,000 feet evaluated at the mean house value and an initial distance of one mile increased house value by US$436.17. No difference was found between wetland types. However, the marginal implicit price for reducing the distance to the nearest lake by 1,000 feet evaluated at the mean house value and an initial distance of one mile increased house value by US$1643.78. This may indicate that the increase in price is related to the amenity of being closer to water and not wetlands *per se*.

In summary, the hedonic pricing model could capture ecological value if it is known that the purchasers of housing were aware of the difference in the ecological value of the site, and willing to pay a premium for this difference. However, the behavioural link that allows amenity to be reflected in house prices cannot be extended to ecological value as the private value related to wetland amenity does not include all the attributes of ecological value (Doss and Taff, 1996; Mahan *et al.*, 2000). Therefore, with respect to criterion 1, it is unknown if changes in ecological value will be reflected by changes in housing prices. This information is essential if ecological value is to be successfully captured by the hedonic pricing method.

It is possible that there are other characteristics of the wetland (not captured in equation 4.1) that provide for different levels of utility. That is, the difference in value may be a result of omitted variables, or demand associated with differing segments of the housing market (Hanley and Spash, 1993; Freeman, 1995). These variables could include anything from mosquito control programs at the site, recreational species of fish present in the wetland, easy access for boating at one site and not another or perceived differences in aesthetics not necessarily related to ecological value. Thus, to determine the ecological value of wetlands using the economic hedonic approach would require controlling for all these factors. It may be quite difficult to find two (or more) wetlands, which have different levels of ecological value and all other factors, the same. Also, wetlands may actually be perceived to have a negative value in the choice of housing as they harbour disease-spreading species such as mosquitoes and midges. One of the causes of wetland loss, outlined in Chapter 1, has been their in-filling for urban dwellings which indicates they are not valued as prime water frontages.
With respect to criterion 2 there is no reason to assume that the hedonic technique will capture any of the attributes of ecological value or reflect their interdependence. This follows from the fact that there is no necessary association between housing markets and ecological value. It has been argued that the attributes of ecological value provide instrumental goods and services for humans but these services do not necessarily include desirable housing locations. As the hedonic technique is a revealed preference method there are at least two links between ecological value, the wetland and the demand for housing. With respect to the fourth criterion, the hedonic technique can only measure use values due to the condition of weak complementarity. That is, for necessary assumption of weak complementarity to hold, for a certain range of prices for housing the demand for a wetland location will be zero and there is no utility derived from simply knowing the wetland exists.

4.4.2 The Travel Cost Method

The travel cost method has been used most often to assess the economic value of the recreational consumption of resources. The method is based on the premise that the recreational experience is worth at least as much as it costs to travel to the site (Clawson and Knetsch, 1966). As people must transport themselves to a site in order to enjoy its services they pay an implicit price, the cost of travelling to it, which includes the opportunity cost of their time (McConnell, 1986). If benefit is derived from the site itself (and not the trip) then this value represents the net economic value, or consumer surplus, of the recreational resource (Menz and Wilton, 1983).

It is assumed in the travel cost method that individuals will react to travel costs in the same way they would react to an admission fee. That is, the more it costs to get to a site the less it will be used (Wilson and Carpenter, 1999). Thus, individuals or households will continue to visit a site until the marginal value of the last trip is equal to the cost of getting there. However, as Kahn (1995) notes the travel cost method can measure the use value of an area but is not well suited to measuring changes in quality of an area. This results from the fact that the simple travel cost method estimates value in a constant quality framework. All participants visit the same site, experience the same characteristics so there is no variation in quality to correlate to variations in travel cost. That is, quality is considered exogenous to the
model (Ward and Loomis, 1986).

The alternative is to use multi-site single-equations models of sites with varying levels of site quality (Ward and Loomis, 1986). Therefore, the first limitation in assessing ecological value using the travel cost is that more than one study must be undertaken. This repetition can be undertaken either spatially or temporally. Most studies that have assessed the quality of a site have compared visits and quality of a number of sites. For example, Kerry Smith and Desvousges (1985) used this approach to estimate the value of water quality and visits to 13 sites along the Monongahela River Basin, Pennsylvania.

Given that the multi-site travel cost method has been used to estimate the value of water quality changes at recreation sites it is possible that the technique could also be used to assess changes in ecological value of wetland sites (Wilson and Carpenter, 1999). This would require the assumptions that ecological value is reflected in recreational consumption. For example, if ecological value is a predictor of actual observed consumption behaviour, the number of trips to a site, the distances travelled or the cost incurred, then the number of visits to a site could be used to estimate the ecological value of each site (Wilson and Carpenter, 1999). Using the individual travel cost method

\[ V_{ij} = f(C_{ij}, E_j, M_i) \]  

Where \( V_{ij} \) is the number of visits to site \( j \) by individual \( i \)
\( C_{ij} \) is the travel cost of individual \( i \) to site \( j \)
\( E_j \) is the ecological value of site \( j \)
\( M_i \) is the income of individual \( i \)

It is assumed that the number of visits will rise when the ecological value of the site is improved. It would also need to be assumed that recreational demand is a proxy for ecological value. This limitation is less vexing for the travel cost than for the hedonic technique as it may in fact hold for some of the attributes of ecological value of coastal wetlands as defined in Chapter 3. As for the direct linkage model it is assumed that an area with high ecological value would sustain recreational fisheries.
Also, it would be expected that a system that exhibited these attributes would be capable of continuing to assimilate wastes thus ensuring good recreational water quality for swimming and boating. However, to make this link would require obtaining information from respondents about the characteristics that were important to them. As Hanley and Ruffell (1993) point out from their travel cost study of forest characteristics, there may be a difference between scientific and household assessment of characteristics. Similarly, the public may view the ecological values of a wetland differently from the ecologist.

However, Ward and Beal (2000, p 196) note that when considering multiple sites with quality changes using the travel cost there is no consensus in the literature on the way to specify quality into the multi-site demand model that is theoretically correct, policy-relevant and fits the data. This presents substantial limitations on the use of the multi-site travel cost to measure ecological value. Given these problems other approaches have been developed to account for variations in characteristics at recreational sites. These include hedonic travel cost methods which considers site quality and random utility recreational demand models which consider multiple sites. These methods will be reviewed to determine if they are less limited in their ability to capture ecological value.

*Hedonic Travel Cost Method*

The hedonic travel cost method was developed by Brown and Mendelsohn (1984) as an attempt to place values on the characteristics of recreational resources. Brown and Mendelsohn (1984) state that if there is a relationship between private expenditure and access to a public good and this varies among individuals, then the individuals face implicit prices on the characteristics of the public good. That is, people will be willing to travel farther to visit sites with higher levels of the characteristics under consideration (Kahn, 1995). Kerry Smith and Kaoru (1987) point out that an important maintained assumption of the model is that it is based on individuals selecting sites dependent on specific characteristics and that the perceived supply of the characteristics are continuously available at fixed rates of exchange. The characteristics are regressed on travel costs to determine the willingness to pay for changes in the level of the characteristics (Ward and Beal, 2000). For example, Brown and Mendelsohn (1984) estimated the average quality of
fishing per trip by comparing the fishing behaviour of people who faced low versus high prices (that is, lived closer or farther away from the site). It was shown that fishers would travel farther distance to obtain higher qualities of the three characteristics measured namely less congestion, better scenery and fish density.

If ecological value was considered a desirable characteristic then the hedonic travel cost method may be able to measure changes in ecological value between sites based on the costs of travelling to the sites. However, the marginal value of the characteristic is related to the extra distance that must be travelled. It may be that the distance to be travelled to a site of higher quality is an accident of nature and does not necessarily reflect the quality of the site (Kahn, 1995; Hanley and Spash, 1993; Ward and Beal, 2000). For example, if the site with higher quality was closer to home and the site with lower quality farther away, then using this method the higher quality site would have a lower willingness to pay (Kahn, 1995). Hanley and Spash (1993) note that negative prices have been found for characteristics which would be expected to have positive prices. To link ecological value to the demand for a site requires information on how these characteristics enter the household’s preferences, however, the hedonic travel cost method cannot provide this information (Kerry Smith and Kaoru, 1987).

Nevertheless, if it was known that demand for a site was related to its ecological value it may still be possible to use the hedonic travel cost method to measure ecological value if conditions are imposed on the distance travelled to sites of varying quality. If two sites, one of high and one of lower ecological value, are equidistant from an individual’s place of residence then the choice will not be influenced by distance but by the characteristic of ecological value. But, it would then be assumed that only the site with higher ecological value would be used, as this is the characteristic considered important by the individual. As distance is the explanatory variable in the method this condition would also impose further very restricted limitations on the sites that could be assessed using the hedonic travel cost method.
Random Utility Travel Cost Model

The random utility model is used to predict the probability of choosing a given site among all possible choices (Ward and Beal, 2000). Ward and Beal (2000) state that this model is suitable if substitute sites can be differentiated by quality by the recreationalist. The model treats each trip made to a recreational site as an independent decision and estimates the probability of choosing a particular site based on given travel costs, the quality of the site and designated individual characteristics that influence recreational demand (Kerry Smith and Kaoru, 1986; Kahn, 1995). The estimated parameters of travel cost are substituted back into the random utility function so that a measure of the net benefits per choice occasion can be estimated (Kahn, 1995). As changes in ecological value can be thought of (for present purposes) as synonymous with changes in quality, this model may be able to capture ecological value.

However, there are a number of limitations with the model. The estimation of value produced by random utility demand models only provide information about the benefit of a one trip occasion (Bockstael, 1995). Thus, as information is collected for single trips the method cannot explain the total number of trips per season or per annum nor the allocation of trips across sites (Ward and Beal, 2000). Thus, more than one study would need to be undertaken to assess the value of sites with varying ecological value. The model also assumes that each trip is independent of the number of trips taken. The same benefit is expected for each trip, however it would be expected that there would be a diminishing marginal utility in each successive trip (Kahn, 1995). It also assumes that each choice occasion is independent of each other choice occasion. It is expected however, that consumers will value variety in the choice of recreational sites (Kahn, 1995). That is, the choice to visit a site today will be a function of previous visits to the site and other sites.

In summary, the travel cost method provides the possibility to capture ecological value through assessing the demand for access to sites with varying levels of ecological value. However, there are major limitations with the approach. First, the simple travel cost model cannot be used as it is assessed in a constant quality framework. Multi-site models could be used to assess difference in sites based on
the ecological value of the site. However, it is unknown if ecological value is a
determinant in a visit to a recreational site. Further, the method is limited by the
difficulties in specifying quality into the multi-site demand model that is theoretically
correct (Ward and Beal, 2000). The hedonic travel cost model offers a better
prospect for measuring the value of characteristics, such as ecological value of a site.
However, the difference in value between sites may be an accident of nature rather
than a reflection of differing ecological value. It may be possible to control the
distance and other factors such as travel condition to be equal between sites but this
severely restricts the application of the model. Random utility models may also
reflect site characteristics but ignore marginal diminishing utility and variety as a
determinant of choice of recreational sites.

Thus, with respect to criterion one, the measurement of a change in ecological value
using the travel cost method would be extremely limited. The method does however
perform better than the hedonic pricing model with respect to criterion 2 as the
demand for recreation at a site may be related to the attributes of ecological value
and their interaction at that site. With respect to criterion 3 there are a number of
links which must be made. First the observed good is travel related consumption,
which is then linked to recreational demand and then recreational level of
consumption needs to be linked to ecological value. As stated the recreationalist
may perceive the link between recreation and ecological value but the model cannot
determine if this is the case. Finally, with respect to criterion four, as a revealed
preference model, all travel cost models can only measure use value. Non-use values
have zero marginal value.

4.5 STATED PREFERENCE MODELS

Stated preference models include contingent valuation and choice modelling. These
methods rely on respondents to surveys stating their theoretical willingness to pay (or
willingness to accept compensation) for a change in the quantity or quality of an
environmental good. These approaches offer advantages over observed market
techniques, as the researcher is able to formulate a scenario that includes the good to
be valued without reference to behavioural trails of demand. Stated preference
techniques can then determine both use and non-use values.
Contingent valuation surveys have been used to assess the monetary value of a range of goods including environmental or ecological goods. A common purpose of contingent valuation studies is to obtain an ex ante valuation of policy impacts that can be used in cost benefit analysis (Mitchell and Carson, 1989). Before reviewing contingent valuation the more recent techniques of choice-based stated preference methods are considered to determine if they would be appropriate to assess ecological value in terms of the criteria suggested in Section 4.1.

4.5.1 Choice Modelling

Choice-based experiment techniques are a class of stated preference techniques that are able to value environmental goods. Choice-based techniques have a long history in the marketing transport literature and applied decision research literature and are well accepted as methods for eliciting consumer responses to multi-attribute stimuli (Adamowicz et al., 1994; Boxall et al., 1996; Rolfe and Bennett, 1996a; Morrison et al., 1997; Morrison et al., 1998). However, they have only recently been used in the field of environmental valuations (Garrod and Willis, 1999). These techniques come under a number of differing titles and are often referred to as experimental or stated choice analysis, choice modelling or conjoint analysis techniques (Adamowicz et al., 1994).

In the case of conjoint analysis, respondents are asked to rank scenarios according to their preference and it is assumed that the ranking used is consistent between the different respondents (Rolfe and Bennett, 1996a). Choice modelling differs from conjoint analysis as it asks respondents to choose a scenario rather than ranking choices (Rolfe and Bennett, 1996a). This choice-based technique will be examined further to determine its suitability for assessing ecological value.

In brief, with choice modelling surveys, respondents are asked to choose between two or more goods or scenarios, where each alternative good or scenario has different levels of characteristics (Kahn, 1995). As each scenario contains a bundle of attributes the choice made will reveal preferences between bundles (Bennett and Adamowicz, 2001). A baseline line alternative which either corresponds to the status
quo or ‘do nothing’ is usually included in each set (Hanley et al., 2001). By using repeated experiments and statistical analysis of the data, the researcher is able to estimate the influence of the different attributes on choices and hence utility (Bennett and Adamowicz, 2001). By including price/cost as one of the attributes, willingness to pay can be indirectly estimated from the choices made. The ratio of the coefficients found from analysis of the probability of a choice being made allows the estimation of the marginal rate of substitution between attributes. The marginal rate of substitution between price and any other attribute indicates the part-worth or implicit price of the attribute.

As the researcher chooses the scenario it is possible to attempt to value the change in ecological value of a site directly using this technique. However, the choice modelling approach relies on the representation of a choice situation, using an array of attributes, rather than valuing specific changes in the good or service (Boxall et al., 1996). According to Boxall et al., (1996) and Adamowicz et al., (1999) with this approach there is less reliance on providing an accurate and complete description of the good in question and a greater emphasis on an accurate and complete description of the characteristics and features that describe the situation. That is, respondents are questioned about a sample of events, drawn from the universe of possible events rather than being questioned about a single event in detail (Boxall et al., 1996).

In designing a choice experiment a set of attributes and levels must be chosen from all the possible descriptions of the good. Garrod and Willis (1999) state that the attributes selected need to be familiar and relevant to the respondents and their levels should be measurable either quantitatively or qualitatively. With respect to ecological value, the ability to measure the attributes and the need to be familiar with them limits the application of the technique. Further, Blamey et al., (1998) suggest that prior attributes should not be included in choice modeling exercises. That is, only the outcome attributes (for example, fishing or waste assimilation) should be included.

As stated in Chapter 3, the ability to measure each of the attributes of ecological value is limited with respect to available information, although productivity can more easily be calculated. Also, in the case of ecological value, it has been argued that
respondents may not be fully informed about the ecological attributes that provide life-support services. In this case merely listing these attributes within a choice set may result in an under valuation. If respondents are unfamiliar with the concept of ecological value and its attributes it will limit the ability of choice modelling to capture this value. Indeed, the unfamiliarity with the concept of ecological value and its attributes is a limitation for all stated preference techniques. The ability of respondents to trade off ecological value with other resources will be determined (at least in part) by their understanding of the concept.

The various levels of the attributes are assigned to choice set by using some form of experimental design. If, for example, five levels of the three attributes of ecological value were included there would be 125 choices \(5^3\) to be presented to respondents using the full factorial. Therefore, it is suggested that a fractional of the factorial is chosen based on an orthogonal mains-effect plan (Adamowicz et al., 1994; Adamowicz et al., 1999). According to Adamowicz et al., (1999) and Garrod and Willis (1999) the main effects plan precludes the analysis of interaction effects between attributes and assumes that consumer preferences depend upon individual attribute levels alone and are not influenced by the combination of levels offered across different attributes. If there is interaction between attributes the main-effects design may produce biased welfare estimates. Garrod and Willis (1999) point out that when designing the choice sets, it is necessary to specify attributes that are independent of each other to avoid problems of multicollinearity. The requirement that the attributes be independent of one another is the major limitation of using choice modelling to measure ecological value.

For example, using a choice-based technique for determining the value of independent attributes such as the preferred colour, size, shape and price of a box of tissues within one survey provides useful information for the manufacturer. An analysis of such an experiment would provide information on the marginal rate of substitution between the colour and price or size and price for the box of tissues. However, it has been argued in Chapter 3 that the attributes of ecological value are known to be interdependent although the correlation between these attributes has not

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3 Multicollinearity will arise when two or more independent variables are correlated in a regression model (Pyndick and Rubinfield, 1981)
yet been fully explored within ecology. This limits the ability of choice modelling to measure ecological value. It is therefore not possible to present a realistic choice set where the level of one attribute changes while the others remain the same. Therefore, theoretically choice modelling can capture the value of the attributes of ecological value but it cannot, due to its requirements of independent attributes, measure the total value of or changes in the value of ecological value.

In summary, choice modelling appears to be able to measure changes in ecological value directly, fulfilling criterion 1. However, the method has been developed to determine the value of the attributes based on the change in the scenario rather than the value of the change itself. Thus, the technique is able to measure the value of the individual attributes of ecological value (depending on the limits to define them quantitatively or qualitatively) but is unable to incorporate the interdependence between attributes. This is the major limitation in using the technique. The technique could be used to measure directly ecological value but in reality the scenario presented would most likely be for example, a change in wetland quantity or quality, which means the scenario is still one link removed from ecological value. Finally, the technique can measure a range of components of economic value.

4.5.2 Contingent Valuation

The contingent valuation method has to date been the predominant stated preference technique used to measure society’s valuation of non-marketed goods (Kahneman and Knetsch, 1992; Kemp and Maxwell; 1993, Portney, 1994). The contingent valuation method establishes a hypothetical market to elicit an individual’s economic evaluation of a natural resource (Durden and Shogren, 1988; Hanley, 1989; Bishop and Heberlein, 1990; Diamond and Hausman, 1993; McFadden and Leonard, 1993). Mitchell and Carson (1989) state that the hypothetical market creates an auction situation for access to the public good for use in the future at a specified price. Durden and Shogren (1988) and Markandya (1992) note that the contingent market includes not only the good in question but also the institutional context in which it would be provided and the way it would be financed. The given willingness to pay (or willingness to accept compensation) for the change in the good is interpreted as
the income change that will leave the consumer just as well off as before the change (Mitchell and Carson, 1989; Diamond and Hausman, 1993).

Information is provided to respondents about the scenario under which the change will occur (Garrod and Willis, 1999). Their willingness to pay is then contingent upon the scenario. As the contingent valuation method does not rely on linkages to marketed goods it has the ability to measure both use and non-use values although considerable controversy exists about the measurement of non-use value (Kahneman and Knetsch, 1992; Diamond and Hausman, 1993; Knetsch, 1994; Kahn, 1995).

The presentation of the scenario is similar to that of choice modelling in that the researcher can define the quantity or quality change to be presented to the respondent. Thus, the method can directly measure the willingness to pay for a change to the good, in this case wetlands. This can then be linked to a change in ecological value. For example, respondents can be given a scenario for increasing the ecological value of a wetland through programs designed to reduce pollution, which is leading to degradation of the wetland, and hence impacting on ecological value. Boxall et al., (1996) and Adamowicz et al., (1999) point out that contingent valuation differs from choice modelling as the purpose of contingent valuation is to assess overall change rather than trying to assess the value of differing attributes. Thus, the interdependence of attributes does not become a limitation in applying the method. All the attributes of ecological value and their interdependence can be included within the contingent scenario.

Nevertheless, Hanley (1989) argues that the contingent valuation is still limited if respondents are unfamiliar with the good to be valued. That is, the major limitation of using contingent valuation studies to measure ecological value is related to information. If the respondent to the survey is unaware of the ecological value of a wetland, then willingness to pay for an increase in quantity or quality of the wetland will not reflect ecological value. The attributes of ecological value reflect the underlying structures, processes and functions of an ecosystem and thus differ from the resultant goods and services produced by the system. Respondents to a survey may be aware of the number of fish caught in the wetland or the recreational benefits
of good water quality but may not conceptually link these goods and services to the underlying ecological values of the wetland.

This problem can be addressed in two ways. The first is to choose a sample of respondents that are likely to be familiar with the good being valued. For instance respondents may either be users of the resource or those that live close to it (Hanley, 1989). This in turn will limit the geographical scope of the study. The second alternative is to provide information within the survey on the ecological value of the area under consideration. It is within the description of the good that ecological values can be incorporated into a contingent valuation survey. That is, if economic agents were informed about the importance of the underlying ecological values of the ecosystems, which provide them with goods and services, then their willingness to pay may be able to reflect both the ecological and secondary economic values of the ecosystem.

A potential problem resulting from the provision of information suggested by Cummings et al., (1986) is that information bias could result from the quality and quantity of information provided in the survey. However, Randall (1986) writing in response to this suggestion, emphasized that information which changes the structure of the contingent market could also change the choice made. Similarly, Freeman (1986) noted that if bids are changed in a systematic way because of changes in the description of the environmental good then respondents have altered their perceptions and valuation correspondingly. This effect was seen as being positive for contingent valuation rather than a potential bias (Freeman, 1986). Mitchell and Carson (1989) also dispute Cumming’s et al., (1986) claim that information bias is a form of systematic error in contingent valuation surveys. It is to be expected that responses will vary according to the information provided as the willingness to pay bids are intended to be contingent on the scenario (Mitchell and Carson, 1989; Hanley and Munro, 1992). Chilton and Hutchinson (1999) argue that it would be assumed that society would be better off if decisions taken were based on choices made by well informed people.

However, according to Willis (1995) and Garrod and Willis (1999) there are no exogenous criteria to specify exactly how much information or in what context
information should be provided to respondents in contingent valuation surveys. Munro and Hanley (1999) also argue that the contingent valuation literature has failed to define the optimal extent of information provision and what exactly is true and accurate information. There must however be limitations to the amount of information provided with respect to the time available in the survey, the ability of respondents to assimilate complex information and the prior information possessed by the respondents (Cameron and Englin, 1997; Carson, 1998). The survey is one avenue for the respondent to receive information about the good and when asked to make a choice is given an incentive to process the information (Carson, 1998).

Kahn (1995) states it is one of the important areas of research in contingent valuation that needs to be resolved empirically. Therefore, before the contingent valuation method can be tested in terms of its ability to measure ecological value, the information issue must be resolved.

With respect to the criterion 1, the contingent valuation method is able to measure changes in values, dependent on the scenario presented. Thus, it is only necessary to undertake one study to determine the willingness to pay for changes in ecological value which is an advantage over using the travel cost technique. The survey method also allows information to be collected on the respondents’ knowledge of the area and to provide them with the necessary information about the ecological values. This is considered an advantage over the hedonic method where a separate survey would need to be undertaken to determine if house buyers perceive and are willing to pay extra to purchase homes adjacent to a wetland that has higher ecological values than another wetland. With respect to criterion 2, the method can encompass all the attributes of ecological value and their interdependence within the information presented to respondents. The interdependence of attributes can be stressed so that respondents are aware that one cannot be traded for another. This is not possible using choice modelling. As for choice modelling, although ecological value can be valued directly using this method it would most likely be linked to the value of an area, such as a wetland. Thus, there would be at least one, but probably two links between the resource being valued and ecological value. Contingent valuation as a stated preference measure is able to measure all the components of total economic
As stated this is an advantage over direct linkage and revealed preference techniques.

4.6 ENERGY ANALYSIS

Given the difficulty in assessing the value of ecological service through the preferences of individuals, it has been suggested by Turner et al., (1995) that the value of ecosystem can be determined through biophysical approaches. These approaches are not strictly economic techniques, as value is not derived from willingness to pay for a service. However, these techniques are used within the discipline of ecological economics (Folke, 1991). The particular biophysical approach of energy analysis is reviewed as it explicitly includes one of the attributes of ecological value, namely productivity. Thus, it may be more suitable than non-market valuation methods for capturing ecological value. Also, energy analysis has been used previously to estimate the value the life-support function of wetlands. For example, Gosselink et al., (1974), Costanza et al., (1989) and Folke (1991) have used this approach in wetland valuation.

The approach first developed by Odum in 1971 places a single estimate of value on a natural area (Blamey, 1992). The aim of energy analysis is to quantify all energy flows inherent in a system. The method considers the total amount of energy captured by an ecosystem as an estimate of their potential to do useful work (Farber and Costanza, 1987). The first step in undertaking an energy analysis requires estimating the gross primary productivity (GPP) of the ecosystem. This value is then converted into fossil fuel equivalents. It is then assumed that there is a proportionality between energy inputs and Gross National Product (GNP) so that the ratio of GNP to total embodied fossil fuel equivalents provides an appropriate energy to dollar conversion (Farber and Costanza, 1987). That is, the primary productivity is converted to an equivalent economic value based on the cost to society to replace this energy source with fossil fuel as measured by the overall energy efficiency of

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4 It may not be possible to measure all components of total economic value in one survey. However, some surveys ask respondents to divide their willingness to pay into the components of value (for example, Kirkwood, 1988; Stone, 1992; Lockwood, 1993; Ogelthorpe and Miliadou, 2000).

5 It is necessary to determine how much raw biomass is required to produce one unit of more concentrated energy such as electricity. For example, Costanza and Farber (1987) consider that 20 calories of biomass is equal to one calorie of fossil fuel.
economic production. For example, using this technique Costanza et al., (1989) estimated the value of wetlands in Louisiana using both a willingness to pay approach and an energy analysis. The total value for willingness to pay estimates for fishing, trapping, recreation and storm protection was $2429 per acre and for the energy analysis the value was estimated to be between $6400 and $10600 per acre (all US$1984 with an 8% discount rate)⁶ (Costanza et al., 1989, 354).

The difference between the value of the wetland as calculated through the willingness to pay and energy analysis approaches may be a result of the fact that not all energy captured by the system provides useful benefits to society. Alternatively, the explanation may be that the full value of the system is not understood by society and thus willingness to pay is an underestimate (Blamey, 1992). From this latter perspective energy analysis may have a useful role to pay in measuring ecological value when full information on the values of the system is lacking.

While not all of the monetary value of an ecosystem as assessed by energy analysis may be perceived to be of value from a human perspective, the measure does give an indication of the life support functions of the ecosystem. However, with respect to criterion 1, the method is not able to directly measure changes in the ecological value of a wetland. As for the simple travel cost method, the assessment is undertaken in a constant quality framework. Therefore, to measure change would require more than one study either over time or using a number of sites. The changes measured would only, however, reflect changes in the primary productivity of the system, not changes in the total ecological value. Therefore, criterion 2 will not be fulfilled as only one attribute of ecological value could be measured using this approach. That is, the method cannot account for the dependency provided by the system or the organisation of species within the system (Folke, 1991). As a result, the method does not capture the interdependence of the attributes.

With respect to criterion 3, energy analysis would be able to measure the change in one attribute of ecological value, productivity, if repeated studies were undertaken.

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⁶ Costanza et al., (1989) noted however that the choice of discount rate made more difference to the final result than any other factor. For example, using a 3% discount rate produced values of $8977/acre for willingness to pay and $1700-28200/acre for the energy analysis.
The advantage of the model is that the link between the attribute of ecological value and the economic measure is direct. The criterion related to the technique’s ability to encompass all the components of economic value is not applicable as it only attempts to measure the life support functions of wetlands, not economic benefits.

4.7 DISCUSSION

Each of the methods reviewed is limited in its ability to measure directly ecological value. The choice between methods must thus be based on minimising their the limitations with respect to the criteria, which were introduced in Section 4.1. A summary of the limitations and an assessment of the ability of each method to fulfill the criteria is given in Table 4.1, using the examples provided in the text.

Examination of Table 4.1 indicates that stated preference models perform better at measuring the change in ecological value, as this can be included in the scenario presented. The stated preference methods are also theoretically able to value all components of ecological value, which is not possible with revealed preference techniques. Therefore, it is suggested that stated preference techniques are less limited than revealed preference techniques for capturing ecological value. Choice modelling offers advantages over contingent valuation in terms of offering respondents a choice of alternative states of the world, which avoids the problem of making money tradeoffs which some respondents find unethical (Kahn, 1995; Rolfe and Bennett, 1996a; Rolfe et al., 1998). Another stated advantage of choice experiments is the ability to estimate the value of alternative options within the one study. This is not possible with contingent valuation where a separate study is required for each policy option under consideration (Blamey et al., 1997).
Table 4.1 Assessment of criteria for capturing ecological value using non-market techniques

<table>
<thead>
<tr>
<th>ECONOMIC METHOD</th>
<th>Revealed Preference</th>
<th>Stated Preference</th>
<th>Bio-physical Methods</th>
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<td>Criteria</td>
<td>Direct linkage models (fish)</td>
<td>Travel Cost (recreation- boating and fishing)</td>
<td>Choice Modelling (wetland preservation)</td>
</tr>
<tr>
<td>1. Changes in value of good reflects changes in ecological value</td>
<td>Limited to valuing single use resource. Value may reflect human effort and not ecological value.</td>
<td>Limited, as the link between housing location and ecological value is unknown.</td>
<td>Requires multi-sites. Hedonic TC limited as distance travelled not necessarily related to value. Random Utility Models – limited to single trip</td>
</tr>
<tr>
<td>2. Number and interdependence of attributes captured</td>
<td>Yes</td>
<td>No</td>
<td>Perhaps</td>
</tr>
<tr>
<td>3. Number of links between good and ecological attributes</td>
<td>Two</td>
<td>At least two</td>
<td>At least two</td>
</tr>
<tr>
<td>4. Benefit categories that method can measure</td>
<td>Direct Use</td>
<td>Direct Use</td>
<td>Direct Use</td>
</tr>
</tbody>
</table>
Choice modelling may then be a superior method for measuring the value of individual attributes within a choice set. However, the method is limited in its ability to measure ecological value because the attributes are interdependent and this cannot be accommodated in the main-effects plan used in choice modelling studies. Thus, the only alternative is the contingent valuation method. The contingent valuation method offers the advantage of being able to directly assess the change in ecological value of a wetland provided all the attributes of ecological value and their interconnectedness are included within the scenario. The survey technique also allows information to be collected with respect to the importance of ecological value to the respondent. That is, no linkages need to be assumed as they do with the revealed preference techniques. However, as stated this method is limited particularly with respect to information. That is, the quality of the responses is dependent on the type of information provided in the survey. Nevertheless, if respondents to a contingent valuation survey are given information on the attributes of ecological value of a wetland it may be possible to capture this value. There are also a number of other methodological issues, which must be resolved before this method can be applied. These issues are addressed in the following chapter.

4.8 CONCLUSION

This chapter has reviewed the major techniques used in economics to value the benefits of environmental amenities. An alternative biophysical approach, energy analysis, was also considered. The review has been undertaken to determine the limitations of the methods with respect to their ability to capture ecological value. Four criteria were specifically developed to assess if the changes in the value of the good reflect changes in ecological value. Further criteria considered important were the ability of the techniques to capture the attributes of ecological value and their interdependence. Also, as the links between the attributes of ecological value and benefits provided by ecosystems are of major interest in this thesis the number of links between the good being valued and ecological valued were also determined. Finally, it was considered important that the method be able to capture all the components of economic value as ecological value produces benefits for all components.
Each of the methods reviewed was limited with respect to its ability to capture ecological value. However, it is considered that stated preference techniques, and contingent valuation in particular offer the best prospect for directly measuring changes in ecological value. Contingent valuation was preferred over choice modelling as it allows all attributes and their interdependence to be measured simultaneously. However, respondents may lack knowledge about the ecological value of a site under investigation. It is therefore necessary to investigate how the provision of information, both type and amount influences willingness to pay in contingent valuation studies. This issue is addressed in Chapter 5, which reviews the literature related to information effects in conjunction with some of the other limitations of the contingent valuation method.
CHAPTER 5

INFORMATION EFFECTS AND THE CONTINGENT VALUATION METHOD

This chapter focuses on the use of the contingent valuation method (CVM) for eliciting willingness to pay for the ecological value of a wetland. However respondents to a survey may not be familiar with the ecological values of the site being assessed and could be influenced by the information given within a contingent valuation survey. Thus, the purpose of reviewing the contingent valuation technique is primarily to determine the effects of information in a contingent valuation survey and in particular to review previous contingent valuation studies that have included information about the attributes of ecological value. This then raises the question of whether the provision of information about ecological value produces a differing willingness to pay response from that which occurs when information is given about other economic values. There are also a number of other limitations with this method identified from previous contingent valuation surveys. Consideration must be given to the choice of payment vehicle, bidding methods and income and substitution effects in terms of resolving potential difficulties. Other problems with this method that have been identified in the literature include the apparent disparity between willingness to pay and willingness to accept and the expression of moral or citizen values. These issues are reviewed in terms of the most suitable format for a contingent valuation study to capture ecological values.

5.1 PREVIOUS ANALYSIS OF INFORMATION EFFECTS

The contingent valuation method asks respondents to a survey for their theoretical willingness to pay (or willingness to accept) for a change in the quantity or quality of a good. Within the scenario presented to respondents information can be provided about the ecological values of the good to be valued. It is expected that the provision of information will alter willingness to pay bids, as these bids are contingent upon the scenario presented to respondents (Mitchell and Carson, 1989; Hanley and Munro, 1992). However, as noted in Chapter 4, there is no consensus within the literature with respect to the amount and type of information to present to respondents (Willis, 1995; Garrod and Willis, 1999; Munro and Hanley, 1999).
A number of studies have been undertaken to examine the effects of information provision in contingent valuation studies in terms of the characteristics of the good (Samples et al., 1986; Boyle, 1989; Hanley and Munro 1992; Tkac, 1998), resource quality (Blomquist and Whitehead, 1998) and services provided by the good (Bergstrom et al., 1990). Hoevenagel and Van der Linden (1993), Edwards-Jones et al., (1995) and Kenyon and Edwards-Jones (1998) have also examined the provision of information with respect to specific ecological attributes. From these studies it would be expected a priori that increasing the information provided about the services, characteristics or ecological attributes of a resource would lead to an increase in stated willingness to pay. What is not clear from their work is whether respondents are simply increasing their willingness to pay as a result of being supplied with increasing information about any aspect of the resource. It is possible that if the same amount of information were provided about any particular direct or indirect use value of the resource the willingness to pay would increase correspondingly. The question then becomes: does willingness to pay when given information about primary ecological values differ from willingness to pay when given information about secondary economic values?

In general it has been found that relatively large changes in the amount of information being provided will affect the willingness to pay bids given (Samples et al., 1986; Bergstrom et al., 1990; Hanley and Munro, 1992; Hoevenagel and Van der Linden, 1993; Kenyon and Edwards-Jones, 1998, Tkac, 1998; Blomquist and Whitehead, 1998). It is also noted that there is a limit to the amount of information that can be presented before respondents suffer from information overload. Information overload can be defined as “the emergence of confused or dysfunctional consumer behaviour caused by increased information” (Bergstrom et al., 1990, 617).

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1 Boyle’s (1989) study on trout fishing is an exception to this general conclusion.
5.2 INFORMATION CRITERIA

Some studies have concentrated on the psychological dimensions of incorporating new information into the value formation process (see Ajzen et al., 1996; Harris et al., 1989). Other contingent valuation studies have investigated the effects of providing varying levels of information about the services of ecological resources or characteristics of species (Samples et al., 1986; Bergstrom et al., 1990; Blomquist and Whitehead, 1998; Tkac, 1998). These services are in turn dependent on the underlying ecological attributes of the system under investigation. Service information describes the possible uses of a commodity, such as consumptive services or attributes associated with recreation (Bergstrom et al., 1990). Characteristic information relates to the particular physical characteristics of a species behaviour and endangered status (Samples et al., 1986). In each case, however, more detail of the same type of information was given. Further it was noted in each of these five studies that more research was required to determine the effects of information. The intended study is designed to further this research by determining the effects of providing information on different types of value, a subject which has not been extensively investigated in studies to date.

The studies outlined above will be reviewed to aid in the design of the survey for the case study area. The purpose of the proposed study is to investigate the effect of providing different information about the various values of an environmental resource on willingness to pay bids. In order to determine the potential effects of providing these differing types of information it is necessary first to review how this information may be assimilated from a psychological viewpoint. Studies that have tested the effects of providing increasing amounts of information about either the physical characteristics or services provided about a resource will then be reviewed to determine the appropriate amount of information to be given to respondents. Three studies which have tested the effects of providing information on the ecological attributes of the good are then examined. These studies which explicitly determine the willingness to pay for ecological features are the most relevant to determining if contingent valuation can capture ecological value.
5.2.1 Psychological Effects

Some psychological research raises questions with regard to the contingent valuation model. Harris et al., (1989) identified a number of issues that need to be resolved to ensure the validity of contingent valuation studies. These include the extent to which people have the ability to process the provided information correctly, the degree to which responses may be influenced in unknown ways and the lack of well-formulated values (Harris et al., 1989). The role of stress on the quality of people’s evaluative judgments also needs to be considered as too little or too much stress could lead to inadequate decision making (Harris et al., 1989).

The manner in which respondents assimilate information needs to be considered when designing contingent valuation surveys. Ajzen et al., (1996) examined the effects of personal relevance and the quality of the arguments presented as a potential source of information bias using a laboratory experiment. It was concluded that the nature of information provided in contingent valuation surveys could affect willingness to pay estimates. Ajzen et al., (1996) argue that as respondents in contingent valuation surveys may lack readily available monetary valuations for unfamiliar public goods then these must be constructed at the time of the survey. Monetary valuation will then be based on the information provided to respondents during contingent valuation surveys. However, this information may in fact contain arguments that are persuasive or produce attitude change (Ajzen et al., 1996). It was found that a detailed description more than doubled willingness to pay for the good (a campus movie theatre). It was also noted that the greater the personal relevance of the good the greater was the effect of argument quality (Ajzen et al., 1996, 52). In conclusion, it was argued that the provision of detailed information to respondents might not overcome problems of information bias. Extra information may not be assimilated if subjects are unfamiliar with the good or if it is of low personal relevance (Ajzen et al., 1996). It was recommended that valuations should be obtained for more than one information scenario to determine sensitivity to information bias (Ajzen et al., 1996).
To overcome the difficulties noted in this research several strategies can be suggested. First, all attempts should be made to avoid the use of emotive language that may produce an attitude change. Second, the description of the good should be confined to outlining well-documented particulars about its value. Third, to alleviate concerns about surveying respondents who are likely to be unfamiliar with the good, the geographical range of the survey can be limited to an area which is in proximity to the site to be evaluated. To reduce the potential for information bias a number of information scenarios can be presented to sub-groups in the sample. This may in turn increase the relevance of the good for some respondents. For example, if indirect uses of the good that the respondents may have been previously unaware of are presented, the relevance of the good may increase. However, the willingness to pay for differing values of the good may still rely on its personal relevance rather than the information presented.

5.2.2 Service and Characteristic Information

As noted, service information describes the possible uses of a commodity and characteristic information relates to particular physical characteristics of a species including endangered status. Samples et al., (1986) undertook a study to investigate the willingness to pay for endangered species and the effects of information provision. Samples et al., (1986) argue that the contingent market behaviour of participants could be modelled on the household production framework. Samples et al., (1986) asserts that in this case respondents first considered the productivity of the input in the production of animal preservation followed by the traditional utility maximization problem of marginal rate of substitution between income and preservation. If the information provided alters either the marginal rate of substitution or the marginal efficiency of investment then the willingness to pay amounts will be altered (Garrod and Willis, 1999). Empirical testing by Samples et al., (1986) on the effect of information disclosure about the physical characteristics, behaviour and endangered status of humpback whales showed that preservation bids were altered by the information given (Samples et al., 1986).

Another consideration in providing information to respondents in contingent valuation surveys is that they may already possess information about the ecological
good offered. To test the hypotheses that people who are well informed about an ecological good will place a higher value on it than those who have limited information Tkac (1998) replicated Samples et al., (1986) research. In addition, following Hoevenagel and van der Linden (1993) the hypothesis was tested that different descriptions of the same endangered species can result in varied levels of valuation (Tkac, 1998). Empirical findings confirmed that willingness to pay values were related to information disclosure and that selective information disclosure significantly influenced preservation bids (Tkac, 1998). It was therefore suggested that research be conducted into the setting of standards for information disclosure in contingent valuation studies (Tkac, 1998).

If information alters the marginal utility of an environmental commodity then it should effect willingness to pay for the good. Bergstrom et al., (1990) undertook a study examining the impact of the provision of service information on willingness to pay to investigate this effect. The contingent valuation survey was undertaken to determine willingness to pay for preservation of Louisiana coastal wetland by recreationalists (Bergstrom et al., 1990). The provision of service information, which described consumption services or attributes associated with recreational trips to the wetland, was conjectured to affect the perceived marginal utility of a given quantity of wetlands but not the objective character of it. That is, with or without service information the same commodity was being valued. It was argued that if participants in a contingent valuation survey do not consider all the beneficial consumption services or attributes they might underestimate the marginal utility of protecting wetlands (Bergstrom et al., 1990).

Bergstrom et al., (1990) considered that the provision of extra information would not result in information overload but this was not explicitly tested. The provision of service information did increase willingness to pay for wetland protection. It should be noted however, that the additional service information provided was of a beneficial nature and that if negative consumptive services were included the

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2 Tkac (1998, 1218) surveyed both economics and wildlife biology students under the assumption that economics students would be less familiar with the good, the Harlequin Duck. There was a significant relationship (p=<0.01) between information disclosure about the ducks and reported WTP values for the economics group. This was not the case for the wildlife biology students (p>0.05) who were assumed to already possess the disclosed information.
willingness to pay may be decreased. Finally, Bergstrom et al., (1990) concluded that the provision of information increases the completeness and accuracy of valuations and thus the information effect is desirable. They suggested that this is an area that requires additional research (Bergstrom et al., 1990). For instance, the study only examined the services or attributes associated with recreational trips, that is, use value. It is proposed to test whether if information related to the underlying ecological features of an ecosystem is provided this will also alter willingness to pay.

From the Bergstrom study it would be expected that willingness to pay will be higher where more information is provided. However, it is unknown if willingness to pay will be higher where information is provided about ecological values as opposed to other values. The ecological values of wetlands will differ between differing ecosystems due to the quality of the resource. The perceived quality of the resource may therefore also affect willingness to pay.

Blomquist and Whitehead (1998) consider that information about resource quality is important when imperfectly informed respondents perceive resource quality that differs from the true quality. This in turn requires that the information provided will allow respondents, who otherwise have little prior information, to become familiar with the resource quality of the good (Blomquist and Whitehead, 1998). Blomquist and Whitehead (1998) tested for information effects about four wetlands of varying quality with different information sets relating to characteristic and service information being presented to four different groups. Characteristic information provided included the wetland acreage, water regime and number of species. Service information included the flood control function and water quality improvement function.

Blomquist and Whitehead (1998) concluded that if additional resource quality information produces higher willingness to pay for higher quality wetlands then it is considered that the theoretical validity of the willingness to pay has been enhanced. Their results showed that wetlands which were described as being of higher quality received higher willingness to pay bids than those of lower quality. Therefore, resource quality information was seen to be a determinant of willingness to pay (Blomquist and Whitehead, 1998). That is, the provision of information about
wetland characteristics and services will differentiate wetlands on the basis of the quality of the wetland for survey respondents with incomplete information. In conclusion, they suggest that further research needs to be undertaken to determine when the benefits of additional information exceed the costs given the potential for information overload.

Another study which investigated the effects of providing increased information was undertaken by Hanley and Munro (1992). The survey used four information sets with each set providing increased amounts of information in terms of relative scarcity of heathlands and characteristics of rare flora and fauna present (Hanley and Munro, 1992). The comparison of willingness to pay bids between the first (basic) and final (full) set of information showed an increase of 79% in willingness to pay (Hanley and Munro, 1992, 20). It was noted however, that moving from the second to third set of information had no significant effect on willingness to pay (Hanley and Munro, 1992). This effect was interpreted as being a case of ‘weak information overload’ where information effects were positive but diminish at the margin (Hanley and Munro, 1992). The relevant question then became how much individual’s information sets can be increased before significant changes in willingness to pay occur (Hanley and Munro, 1992). This study again demonstrates that contingent valuation studies will be improved by the provision of additional information related to the characteristics of an area.

Studies undertaken which have included increasing information about the characteristics or services of a resource suggest that better informed respondents will increase their willingness to pay for environmental resources (Samples et al., 1986; Bergstrom et al., 1990; Hanley and Munro, 1992; Tkac, 1998; Blomquist and Whitehead, 1998). These studies, however, are concerned with evaluating the secondary values of the good. The question remaining is if supplying information about primary or ecological values will also alter willingness to pay bids.

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3 The first set provided only basic information, and the second set gave the basic information plus information on the relative scarcity of heathlands. The third set of information provided the basic information plus information on the characteristics of the rare flora and fauna of the area. The final set of information included information from the basic set, relative scarcity and characteristic information (Hanley and Munro, 1992).
5.2.3 Ecological Attributes

The studies which are most applicable to the proposed survey are those which relate to willingness to pay (WTP) for ecological attributes. Hoevenagel and van der Linden (1993) investigated the effect of supplying differing descriptions of an ecological good on willingness to pay bids. They noted that as ecological goods cannot be shown to respondents they must be described and in many cases this is the only information that is available to respondents on which to base their valuation. As a full description of the good may result in information overload, it is left to the researcher to select the partial information that will be provided. Even if respondents are familiar with the good, reference to the selected attributes within the survey may seem to increase their importance. Hoevenagel and van der Linden (1993) argue that if different descriptions of the good result in differing values this will produce an undesirable information effect. However, it could be expected that values will differ according to information provided as the bids are contingent on the scenario presented (Mitchell and Carson, 1989; Hanley and Munro, 1992). The resulting differing values are therefore, not necessarily undesirable.

The ecological good to be provided in the Hoevenagel and van der Linden (1993, 224) contingent valuation survey was “a clean environment around the year 2015”. Three sub-samples were surveyed. The first group received only the overall target and reference level, the second group received information about four ecological issues, which form part of a clean environment, and the third group received information about seven ecological issues that would be present in a clean environment. It was noted that a clean environment could be described in many other ways (Hoevenagel and van der Linden, 1993).

Their hypothesis was that if a large difference in the description of the good was provided to respondents, then it would have a significant effect on their willingness to pay. The results of the survey indicate that the difference in willingness to pay between the reference level and those given either four or seven issues was significant. However, the difference in willingness to pay between those who received four issues to those who received seven issues did not produce the same
This study demonstrates that descriptions of the good that provide more information are likely to increase willingness to pay. The results also suggest that providing information about more than four or five issues will not significantly increase willingness to pay and this finding may offer some guidance in choosing the appropriate amount of information to provide in contingent valuation surveys. The study does not, however, demonstrate if willingness to pay based on ecological attributes differs from willingness to pay for other use or non-use values of the good. This proposition still needs to be investigated.

In contrast to Hoevenagel and van der Linden (1993), Edwards-Jones et al., (1995) tested whether there was a significant correlation between the rankings of five sites based on willingness to pay and the ranking based on ecological assessment. They argued that if there was agreement between the relative evaluation of ecological goods by monetary and non-monetary techniques, the contingent valuation method should be supported for use in land use decisions.

The ecological value of the sites was determined by collecting quantitative data on the distribution and abundance of the flora, birds and selected macroinvertebrates. Given that that there is no universally accepted method for testing if one site is more ecologically important than another the attributes considered to constitute ecological value included only the species diversity, species rarity and the area of the site. This information was sent to ten expert ecologists who were then asked to rank the sites in order of ecological interest.

A face to face contingent valuation survey was conducted at each of the five sites. No further information with respect to the ecological data was provided. As the study sites were areas of open access, it was considered that respondents would be familiar with them. The respondents were asked their willingness to pay to preserve the areas and their willingness to pay if there was a 50% and 100% increase in species richness (Edwards-Jones et al., 1995, 220). It was found that there was no

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4 The difference in mean willingness to pay between the reference level and four issues and seven issues was significant (p<0.01). The difference in mean willingness to pay between four and seven issues was not significant (p>0.05) (Hoevenagel and van der Linden, 1993, 232).
significant difference in the willingness to pay bids for increased species richness. The attributes identified as important for respondents were the landscape, nice place to walk and open spaces. The mean willingness to pay bids were not statistically different between sites which suggested to the authors there was no difference in value to the respondents although ecological experts did value the areas differently. It was noted that a limitation of the study was that although the ecological experts valued all the sites the contingent valuation respondents were only asked to value one site (Edwards-Jones et al., 1995).

According to Edwards-Jones et al., (1995) the discrepancy between the expert and public valuation of the sites may have been a result of the respondents not possessing sufficient information to make a rational evaluation and willingness to pay bid. The problem still remains as to what is the necessary minimum amount of information which needs to be provided to respondents to value ecological goods (Edwards-Jones et al., 1995). The authors thus suggest that research should be conducted to produce a suitable ecological information set so that the difference between expert and public value could be rectified (Edwards-Jones et al., 1995). Defining ecological value in terms of measurable ecological attributes, as given in Chapter 3, is a first step in developing a consistent ecological information set which could be used to rank ecosystem if the ecological data were available.

The results of the Edwards-Jones et al., (1995) study indicate that respondents did value the secondary outputs of the environment, landscape for example, but they did not value the individual elements of primary value that contribute to this output. This conclusion allows conjecture as to whether respondents are either aware or care about the ecological attributes of the ecosystems they are being asked to value. It is possible if information had been provided on the landscape and open spaces which provide recreational opportunities for walking and wildlife viewing (direct secondary values) that the results would have been similar. Designing a series of questionnaires, which value the same environmental good but provide information on the differing values of the resource, will test this proposition.

Further research was then undertaken by Kenyon and Edwards-Jones (1998) to test the effects of providing information on the ecological value of sites to contingent
valuation respondents. Four sites were chosen that were ranked according to ecological experts and willingness to pay bids (Kenyon and Edwards-Jones, 1998). It was suggested that if each group ranked the sites in the same way then the substitution of contingent valuation for expert evaluation in land use decisions might be viable (Kenyon and Edwards-Jones, 1998). Six different categories of information, including textual, photographic and ecological information were provided to respondents to test which level of information was the most appropriate (Kenyon and Edwards-Jones, 1998). Kenyon and Edwards-Jones (1998) state that the textual information and the ecological information were taken from the full ecological survey reported in Edwards-Jones et al., (1995). Information was only provided on species richness and abundance. When respondents were provided with photographic, textual and ecological information they ranked the sites in the same order as the experts which suggests that this is the appropriate level of information (Kenyon and Edwards-Jones, 1998, 473). That is, the provision of ecological information increased the validity of the public valuation (Kenyon and Edwards-Jones, 1998).

Kenyon and Edwards-Jones (1998) note that respondents may have constructed their preference during the survey because of the provision of information about species diversity. It is therefore possible that as respondents were given the same ecological information as the experts their evaluation of the sites was also the same (Kenyon and Edwards-Jones, 1998, 473). This outcome may arise as a result of respondents using temporarily accessible information (Schwarz, 1997). According to Schwarz (1997) this information comes to mind because it has been used recently, often to answer a preceding question. This contextual effect is considered undesirable in survey work as it does not represent the underlying opinions of the population but rather those formed during the process of the interview (Schwarz, 1997). Nevertheless, the study did illustrate that the public are capable of assimilating and responding to ecological information and this resulted in more reliable willingness to pay bids (Kenyon and Edwards-Jones, 1998).

From the review of studies which have investigated the effects of information on willingness to pay for ecological attributes, some general guidelines can be inferred. First, respondents are capable of assimilating ecological information if it is written in
a non-technical manner and this can be further supported by photographic and textual material. From this, it would be expected that providing information about ecological value would produce a positive willingness to pay. Second, from Hoevenagel and van der Linden (1993) study it could be expected that including more than four or five ecological issues would not result in a significant increase in willingness to pay and may result in information overload. The level of information to provide with respect to ecological issues is, however, still undefined.

5.3 PAYMENT VEHICLE

Following an outline of the change to the environmental amenity to be valued, respondents are presented with the proposed payment system. Such mechanisms includes increased expenses per trip, income, sales and property taxes, sewerage fees, hunting and fishing licences fees, entry charges and trust fund payments (Anderson and Bishop, 1986; Garrod and Willis, 1999). The choice of vehicle will depend on the context of the survey. However, the payment vehicle should avoid emotional responses to the payment vehicle itself (Anderson and Bishop, 1986). That is, it should be neutral. If the respondent has some emotive response to the payment vehicle then the bid may be lower than the true value, resulting in payment vehicle bias (Hanley, 1989). To help overcome this potential bias the vehicle chosen must be plausible and be the actual method of payment if possible. Also, the payment vehicle should be perceived to be equitable and fair in terms of those paying for the good should be the recipients of its benefits (Diamond and Hausman, 1993; Garrod and Willis, 1999).

The choice of vehicle will thus to some extent depend on what is considered neutral and fair by respondents. This is likely to vary across nations, which will have different systems of taxes, entrance fees and environmental trusts. Evidence from Australia is reviewed to aid the choice of a payment vehicle for the intended study area. When testing the application of the contingent valuation in the Australia, Bennett et al., (1997) noted that respondents tended to believe that environmental projects were the government’s responsibility and needed to be convinced that the government has insufficient funds to carry out the project. As a result Bennett et al., (1997) found that there were more protest bids using income tax as the payment
vehicle. Conversely, another Australian study by Jakobsson and Dragun (1996) which tested for the effects of the payment vehicle on willingness to pay found that people were willing to pay more when the vehicle was tax rather than a conservation trust fund. However, Stone (1992) argues that employing a non-government organisation for the collection of payments is an appropriately familiar and neutral method of collecting donations in the Australian context. Given these conflicting perspectives it is essential that any objections to the payment vehicle that result in zero bids or refusal to answer are identified by follow-up questions in the survey. In this way protest bids can be identified and dropped from the analysis and payment vehicle bias can be avoided (Garrod and Willis, 1999).

The differing views with respect to a neutral payment vehicle make the use of focus groups and pre-testing essential before a full contingent valuation study proceeds. The issue of taxation has recently received considerable debate in Australia as the government introduced a new goods and services tax in July 2000. This may increase resistance to the use of tax as a payment vehicle given that all citizens face an increased tax burden. An alternative is to set up a levy on council rates for the collection of payments. However, if there are a number of councils in the survey area with inconsistencies between current environmental levies then this payment vehicle may not be seen as neutral by those already paying a levy. Thus, it is suggested that a voluntary fund set might be seen as more equitable and acceptable payment vehicle in the Australian context.

5.4 BIDDING METHODS

After information about the good has been provided, respondents are asked for either their willingness to pay for the good or alternatively their willingness to accept (WTA) compensation for loss of the good. Five principal elicitation methods have been used for contingent valuation assessments. These are open-ended questions, closed-ended questions, dichotomous-choice questions, payment-card formats and iterative bidding games (Anderson and Bishop, 1986). However, since the mid-1980s closed choice questions have gained widespread acceptance (Hanemann and Kanninen, 1999). Each of the methods has its own weaknesses and strengths and will be reviewed to assess which is most suitable in terms of rigour for eliciting bids.
for ecological value (Durden and Shogren, 1988, Hoevenagel, 1994). The method of
survey delivery and the purpose of the study will also effect bid elicitation choice.

The aim when eliciting a bid from respondents is to encourage a willingness to pay
bid that is as close as possible to the ‘true’ unknown value. In order to achieve this,
the possibility of bias must be minimised to the fullest extent possible. Biases may
exist if the respondent is unfamiliar with the good and formulating a value for it, uses
a given bid as a starting or anchoring point or engages in strategic behaviour (Willis,
1995). Strategic behaviour may be present if a respondent underestimates their true
willingness to pay as they believe the good will be provided anyway (‘free-riding’) or
overstates their true willingness to pay in order to ensure the good is delivered or
to influence policy.

The lack of familiarity with the good and valuation of it is particularly vexing for the
open-ended bid elicitation method. For example, if presented with an open-ended
question such as the maximum amount they would be willing to pay to enter an
unfamiliar national park it may be difficult for the respondent to formulate an answer
(Anderson and Bishop, 1986; Durden and Shogren, 1988). That is, this method may
not give respondents enough impetus to consider fully the value they place on the
resource. Given that ecological values are likely to be unfamiliar to respondents, an
open-ended format may underestimate the willingness to pay for ecological attributes
(Bateman et al., 1999). Open-ended formats do, however, continue to be used for
contingent valuation surveys because it allows the respondent to state a value without
the aid of additional information and hence this method is amenable to mail surveys
as well as personal interviews. Also, open-ended questions continue to be used in
pilot surveys to determine the range of values to offer for other methods.

The problem of uncertainty may be overcome to some extent by offering a range of
values for the respondent to choose from (closed-ended questions) or by offering a
‘take it or leave it’ (dichotomous choice) amount. The problem may then, however,
arise that the respondent becomes fixed or anchored to the initial bid given. This
influence is termed starting point bias. Anchoring or starting point bias is considered
a problem for the contingent valuation method as the willingness to pay amounts
offered influence the final willingness to pay of the good (Garrod and Willis, 1999).
That is, the initial bid is taken as an indication of what the value should be (Bishop and Heberlein, 1990). For instance in the case of closed-ended questions, respondents are presented with a number of willingness to pay options. The respondent is then asked to choose one value. This then anchors the answer to the range of values being presented although an ‘other’ category may also be provided (Garrod and Willis, 1999).

One method, which attempts to avoid starting point bias, but provides a range of payments for respondents to choose from is the payment card method. In this case, a card is presented which gives a range of potential contributions from zero to some upper limit. The card also shows information on the amount of tax the respondent contributes to other public expenditure, such as highways or public education given the respondent’s particular income/taxation cohort (Anderson and Bishop, 1986; Durden and Shogren; 1988, Garrod and Willis, 1999). This then gives respondents a frame of reference to help them in valuing the good (Garrod and Willis, 1999). This method does, however, require that personal interviews be conducted so that the payment card is tailored to suit the particular income/taxation group and this is likely to increase survey costs.

Iterative bidding games present an initial bid to respondents and then the amount is increased or lowered to find the maximum willingness to pay bid usually with a final open ended question (Anderson and Bishop, 1986). However, if individuals do not wish to engage in a lengthy bidding process, the low starting point may be related to low bids as respondents suffer from fatigue from the protracted questionnaire (Anderson and Bishop, 1986; Hanley, 1989). There is however, no agreement in the literature as to whether the final bids are over or under estimated with Mitchell and Carson (1989) and Bateman et al., (1999) offering contrasting views on the subject.

Using the first quartile, median and third quartile amounts from a pilot survey that used open-ended bids can help to establish appropriate starting points bid and overcome bias. Also, the number of iterations can be limited so that the bidding process does not cause fatigue and at the same time allows respondents to search their preferences thoroughly. This preference searching is desirable as it increases the probability of respondents offering a bid that reflects the worth of the good to
them. It is therefore argued that an iterative bidding technique is suitable for eliciting a willingness to pay for ecological value if the bids are derived from an open-ended pilot study and the number of iterations is limited.

Another approach is to present a fixed amount to the respondent with a take it or leave it option. This dichotomous choice or referendum evaluation technique asks respondents if they would pay a specified amount for the good. The respondent can then either agree or disagree. Thus, the method is open to starting point bias in much the same way as the iterative bidding technique (Garrod and Willis, 1999). The National Oceanic and Atmospheric Administration (NOAA) panel\textsuperscript{5} advocates the use of the dichotomous choice elicitation method for damage assessment as it is argued it more closely resembles the situation that consumers face in market transactions or in a political referendum (Arrow \textit{et al.}, 1993; Garrod and Willis, 1999). As a result this method of bid elicitation has come to dominate in contingent valuation studies in the US and elsewhere since the mid-1990s (Garrod and Willis, 1999). However, Willis (1995) argues that this form of voting is less prevalent outside the US and may not be familiar to respondents.

\subsection*{5.4.1 Dichotomous Choice and Iterative Bidding Formats}

Both the dichotomous choice and iterative bidding methods offer the potential for eliciting willingness to pay bids for ecological value. The advantages and disadvantages of each method are reviewed in terms of necessary sample size, analysis of resultant bids and purpose of study, in order to determine which method is most appropriate.

The use of the referendum format requires a larger sample size than open-ended or iterative bidding methods (Willis, 1995). For example, a sample of size of 160 –385 completed questionnaires in open-ended or iterative bidding methods is sufficient if it is accepted that willingness to pay to lying within 20% of true willingness to pay 90% of the time provides statistical reliability (Willis, 1995). In comparison Arrow \textit{et al.}, (1993) suggest at least 1000 surveys are required to limit sampling error to

\textsuperscript{5} The NOAA panel made recommendations to the US Department of Commerce’s National Oceanic and Atmospheric Administration (NOAA) with respect to the use of contingent valuation for damage assessment following the Exxon Valdez oil spill in 1989 (Arrow \textit{et al.}, 1993).
plus or minus 3% for a single dichotomous question. Further, a large open-ended pilot study must be conducted to determine the bids to be offered to different respondents in dichotomous choice surveys. However, there is no literature based consensus on the best method to use to facilitate the choice of bids to be presented to respondents. Garrod and Willis (1999) note current methods are complex and require considerable computational effort. It is suggested that a design where bid values are specified close to the true value of median willingness to pay and avoids placing bids in the tail of the distribution (e.g. the outer 12% of each tail for normal and logistic distributions) will work best (Garrod and Willis, 1999). Having determined the bid amounts different respondents are assigned different amounts at random (Anderson and Bishop, 1986).

Hoevenagel (1994) points out that dichotomous choice only reveals a discrete indicator of the maximum willingness to pay value. To calculate the mean of willingness to pay requires the use of logit or probit techniques because of the censoring and truncation of bids. These methods model the probability of the bid amount being rejected as a function of the bid amount and the explanatory variables such as income. The mean willingness to pay is given by the expected value of the cumulative probability distribution curve. These models are sensitive to the functional form of the model and to the truncation point applied to the data (Willis, 1999). Garrod and Willis (1999) note that there are no clear statistical grounds for selecting the specification of the model. Bateman et al., (1995) report an analysis of the effect of using differing truncation techniques for estimating mean willingness to pay amounts for a contingent valuation study of the Norfolk Broads. Three model were selected which truncated from 1) $\infty$ to $-\infty$, 2) to the limits of the observable data and 3) from 0 to $\infty$ (Bateman et al., 1995). The results demonstrated that the first and third truncation were essentially the same for the log-logistic model but that the third model produced a lower willingness to pay than the second model. The point of truncation (that is, the point at which higher bids are excluded) also influenced the mean willingness to pay measure (Bateman et al., 1995). Thus, the assumptions

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6 Arrow et al., (1993) note that sampling statisticians need to be consulted for appropriate design of CV surveys for legal or policy-making purposes.
about the mathematical form of the valuation function and the truncation method will influence the mean willingness to pay amount obtained as suggested by Mitchell and Carson (1989). With respect to differences in mean willingness to pay using the differing elicitation formats Garrod and Willis (1999) report that referendum responses provide mean willingness to pay bids that are between 1.5 and 4 times as large as those based on open-ended questions. Brouwer et al., (1997) in a meta-analysis of contingent valuation studies of wetlands also note that dichotomous choice estimates of willingness to pay are higher than iterative bidding which in turn is higher than open-ended amounts.

Garrod and Willis (1999) argue that an advantage of the dichotomous choice method is that respondents are unable to try to act strategically. This avoids the potential for free-riding or strategic behaviour and starting point bias. However, the difficulty remains as to the selection of the range of payments that needs to be presented to capture the value of the good which as noted requires a prior pilot study. Further, the dichotomous choice responses do not provide a direct measure of willingness to pay, which can be obtained using the open-ended formats (Hanemann and Kanninen, 1999). For the purposes of damage assessment or policy advice in the US the referendum technique may be considered the most appropriate but this does not necessarily extend to research related to other aspects of the contingent valuation method. As Willis (1995) notes, it is doubtful if contingent valuation studies in Britain should adopt only a referendum format. As the purpose of this research is to investigate the use of contingent valuation to assess the effects of levels of ecological information on willingness to pay the use of the dichotomous choice format would only increase the statistical difficulty in measuring the differences between willingness to pay for differing sub-samples. That is, the importance is not placed on providing information for damage claims but to compare willingness to pay amounts. Thus, given the technical difficulties in determining the mathematical form of the valuation function, the method does not offer any further advantage over the iterative bidding method for measuring ecological value.

In summary, Anderson and Bishop (1986) state that there is no consensus on the best method for asking contingent valuation questions or which format has a statistically significant effect on the results. Further, Willis (1995) notes that there is no
definitive evidence that referendum models perform better than open-ended, payment card or iterative bidding methods. As Willis (1995, 127) states it “there is no 24 carat gold standard against which results from different methods can be compared”. Given that the different formats will produce differing results the important issue is to use the same format for all sub-samples to be compared in a survey. However, as it is considered that ecological attributes may be unfamiliar to respondents and they are unlikely to have a predetermined value for them an open-ended format may make the valuation task difficult for respondents. Thus, it is not considered an appropriate method for bid elicitation. Both the dichotomous choice and iterative bidding methods reduce cognitive difficulties and require a pilot study using an open-ended method to determine the bids to be offered. The dichotomous choice then requires a sample of at least 1000 for each case to be presented to respondents. Where the purpose of research is to compare information across a number of sub-samples the dichotomous choice format would increase the costs of survey work considerably. Therefore, as the iterative bidding method can directly measure the mean willingness to pay, it is seen as preferable to the dichotomous choice, which requires complex assumptions. For these reasons, it is suggested that the iterative bidding method is the most suitable elicitation format for determining the willingness to pay for ecological attributes where the amounts to be offered have been determined from a pilot study.

5.5 INCOME AND SUBSTITUTION EFFECTS

It has been recommended by the NOAA panel that before bid elicitation takes place information should be provided to respondents that includes reminders about budgetary constraints and substitute goods (Willis, 1995). Problems can arise in contingent valuation estimations if respondents fail to consider the full range of substitute goods that are available and the full opportunity costs of their given willingness to pay. Therefore, respondents need to be reminded that substitutes exist and that they face contingent valuation budgetary constraints and that these should be taken into account when forming responses (Rolfe et al., 1998). The difficulty here is the amount of information and number of substitutes that can be conveyed to the respondents (Rolfe et al., 1998). Respondents also need to be mindful of their economic constraints, if not over-inflated answers may be interpreted as using ‘play
money’ (Rolfe et al., 1998). These value bids may however, be a result of high non-use values being expressed.

Bergstrom et al., (1989) have investigated the effects of information about income on willingness to pay and Whitehead and Blomquist (1991) have studied the effects of substitutes on willingness to pay. These studies are reviewed to determine the importance of providing this information in the assessment of ecological value. Bergstrom et al., (1989) defined information effects as changes in bidding behaviour resulting from changes in the contingent market information structure. The authors argued that changes in bidding due to this effect were neither good nor bad but needed to be understood to judge the desirability of specific information effects. Empirical tests were developed with respect to the provision of perspective, relative expenditure and provision cost information respectively7. An iterative bidding game was used with respondents allowed to alter their bids after information was given in relation to perspective income, relative expenditure information and finally the cost of providing the good. This produced small but not statistically significant changes for each successive bid (Bergstrom et al., 1989). However, the difference between the initial and final bid was statistically significant. This difference was seen as desirable by the authors as it was considered the provision of the three types of information produced more accurate estimates of willingness to pay (Bergstrom et al., 1989).

Whitehead and Blomquist (1991) investigated the effects of providing information about substitute environmental goods. The study investigated the effects of providing respondents with differing sets of information about lake reclamation following initial wetland losses and also information about a nearby wetland. The results showed that willingness to pay might be either overstated if information is not provided about substitute goods or understated if information about complementary goods is not given (Whitehead and Blomquist, 1991). This conclusion illustrates that the provision of extra information, including information on substitute and

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7 Information was provided about perspective income (the percentage of income the WTP bid represented), relative expenditure information (typical annual expenditures and the percent of income this equalled) and provision cost information (total cost of provision and that the stated bid would not be sufficient to cover this cost). Respondents were allowed to alter their bids after the provision of
complementary goods will alter willingness to pay. That is, the respondents do assimilate the information provided in the survey and use it in their value formation. Given this evidence, information about income and the availability of substitutes should be provided as fully as possible to respondents of contingent valuation surveys.

There is, however, a difficulty in defining substitutes for ecological value. Ecological information about the attributes of ecological value for differing ecosystems is presently unknown. It is unlikely that any two areas will provide for the same productivity, dependency and organisation of biodiversity. In this case, it is difficult to say if one ecosystem is a substitute for another in terms of its ecological attributes. If a similar ecosystem, for example wetland, exists within a bioregion and it is in a similar condition, (i.e. natural or degraded) then this could be called a substitute. There may, however, be no such substitute within a defined spatial scale. The spatial scale and identification of substitute sites with respect to ecological value will need to be determined on an individual basis.

5.6 THE DISPARITY BETWEEN WILLINGNESS TO ACCEPT AND WILLINGNESS TO PAY MEASURES

The contingent valuation method has the potential to ask respondents what they are willing to accept (WTA) for the loss of a good and the amount they are willing to pay to receive the good on offer (WTP) (Mitchell and Carson, 1989). Willig (1976) notes that theoretically these two measures of welfare change should be equal if there are no income effects. However, empirical evidence in a number of contingent valuation studies shows that willingness to accept values exceeds the willingness to pay measures (Anderson and Bishop, 1986; Hanley, 1989; Eberle and Hayden, 1991). A classic example of this divergence in found in a study by Hammack and Brown of duck hunters who were willing to pay an average of $247 to save a marsh area used by ducks but demanded $1044 to accept the identical loss (see Knetsch, 1994, 353; Brown and Gregory, 1999). Other studies, experimental and market based, also suggest that the compensation demanded to give up an entitlement far exceeded the comparable payment measure of value (Knetsch, 1994; Brown and
According to Markandya (1992) and Boyle and Bergstrom (1999) the reasons for the disparity between willingness to pay and to accept are not fully understood and remain enigmatic. A number of explanations have been given related to income effects, respondents’ lack of experience in revealing preferences, motivation based on the psychological arguments of loss aversion, endowment theory and that ethical beliefs are being expressed. To avoid the contradiction between the two measures many studies do not attempt to measure willingness to accept (Cummings et al., 1986; Knetsch, 1990; Eberle and Hayden, 1991). Mitchell and Carson (1989) note, however, there are strong theoretical grounds for assuming that willingness to accept cannot be replaced by a willingness to pay measure without biasing the findings. If large differences between the alternative values measures exist, then there are substantial implications for environmental policies and assessments (Knetsch, 1990).

The appropriate measure to use when attempting to measure ecological value will depend on the underlying property rights. If an individual owns the good then the compensation required for its loss is the correct utility measure. Garrod and Willis (1999) argue that if the individual does not own the good then the maximum they would be willing to pay for it is the correct utility measure. Ecological value has been defined as those characteristics of an ecosystem which exist prior to the production of goods and services which may be privately appropriated. When the ecosystem in question is held publicly there are no private property rights although the goods and services it produces may become private property. That is, if ecological attributes are considered to be public goods there can be no individual ownership and thus willingness to pay is the correct measure.

The disparity between willingness to accept and willingness to pay is of particular concern when the purpose of the valuation is to provide a measure to be used in cost-benefit studies that involve preservation or development of areas that are presently in a natural state (Blamey et al., 1995). However, the issue will not be as critical for the proposed purpose of conducting a contingent valuation, which is to assess if the

each successive bid and between the first and final bid.
method is capable of capturing ecological value. Also, it is proposed to present a scenario where the resource has been previously degraded through a range of community actions. In this scenario, there is no one agency which could be held responsible for paying compensation. A willingness to accept a compensation framework may thus appear implausible to the respondent.

5.7 PROBLEMS RELATED TO THE EXPRESSION OF MORAL OR CITIZEN VALUES IN CONTINGENT VALUATION SURVEYS

Diamond and Hausman (1993), among others, suggest that values given in contingent valuation surveys may be an expression of moral beliefs (Kahneman and Knetsch, 1992; Stevens et al., 1991; Blamey et al., 1995; Sagoff, 1988). Arguments that suggest moral values are being expressed in contingent valuation surveys are associated with problems related to scope or embedding, lexicographic preferences and consumers acting as citizens. These problems are briefly reviewed to assess their possible effect on the measurement of ecological value.

5.7.1 Scope and Embedding

If respondents do not give significantly different responses to changed amounts of the item being valued then they are said to be insensitive to scope (Rolfe, 1998). Scoping issues may occur if respondents find it difficult to isolate particular geographical areas from each other or if they do not differentiate between benefit sub-components. It may also occur if respondents treat the issue being considered as symbolic of a wider group of public actions and place their values on this wider category (Mitchell and Carson, 1989).

Kahneman and Knetsch (1992, 58) assert that the value of public goods as assessed by the contingent valuation method is arbitrary as the willingness to pay varies over a wide range depending on whether the good is being assessed on its own or embedded as part of a more inclusive good. The term the authors assign to this problem is embedding. For example, it was found that the willingness to pay estimate for protecting 2 000 birds was not statistically different from the willingness to pay estimates to protect 100 times more birds (Desvousges et al., 1993, 113). This
contradicts a basic tenet of economic theory that rational consumers prefer more of a
good or service to less. However, with respect to the proportion of birds to be saved
the program offered to save either 2% or 1% of the population (Desvousges et al.,
1993, 125,126). Garrod and Willis (1999) suggest that respondents may see little
difference between these two figures given the total of 8.5 million birds.
Nevertheless, Kahneman and Knetsch, (1992) hypothesise that responses to
contingent valuation surveys express a willingness to acquire a ‘sense of moral
satisfaction’, not the economic value of these goods. Similarly, Diamond and
Hausman (1993, 27), argue that sums given in contingent valuation surveys are given
in part from the ‘warm glow’ produced as a result of giving, in a similar way to
giving money to charity.

Knetsch (1994) also reports that contingent valuation responses, in the presence of
embedding, show a total lack of consistency. The value differed depending on how
the good is combined with or without other goods (Knetsch, 1994). Knetsch (1994,
362) states that “the contingent valuation answer is but an artifact of the question, not
an indication of the value of the good at issue”. To overcome these concerns it is
necessary to define clearly the good which is to be valued. With respect to
ecological value, it is necessary to state clearly the scope of the geographical area
where these values occur and to remind respondents that their willingness to pay
relates only to that area. However, it is difficult to assess if respondents are
expressing a willingness to pay for the particular wetland under investigation or if
they are valuing all wetlands in the process. Further possibilities to overcome this
problem may be to ensure that respondents are familiar with the particular wetlands
and are located in a nearby geographical area. A map that clearly outlines the area to
be valued could also be included. These measures should then ensure that the
respondent has a clear notion of the particular wetland that they are being asked to
value and avoid problems related to scope.
5.7.2 Lexicographic preferences

If individuals hold an ethical position where they refuse to trade off environmental quality for money they are said to hold a lexicographic position (Spash and Hanley, 1995, 192). A study which investigated the possibility that moral values or lexicographic preferences were being expressed in contingent valuation surveys was undertaken by Stevens *et al.*, (1991). The contingent valuation study was conducted to estimate the value of four wildlife species and included a number of follow up questions. Many of the respondents expressed moral beliefs about wildlife, which raised questions about the appropriate role of estimating existence value in monetary terms. Twenty-five percent of the respondents who were unwilling to pay for the preservation of the species stated their reason for making that choice was based on ethical reasons, claiming that wildlife should not be measured by monetary means (Stevens *et al.*, 1991). This raised the issue of whether individuals were acting as egoists as assumed by this neo-classical methodology or if they were acting as ethicists motivated by a commitment to the existence of wildlife. This led Stevens *et al.*, (1991) to the possibility that ethicists state their preferences according to a lexicographic rule where indifference and trade-offs between money and wildlife are undefined. That is, above a certain subsistence level of income, the individual acting as an ethicist will be unwilling to trade a decrease in the level of wildlife for income. They concluded that of the respondents who were unwilling to pay “used decision-making process that were inconsistent with the neoclassical paradigm of trade-offs between money and wildlife” (Stevens *et al.*, 1991, 399).

Similarly, Spash and Hanley (1995) undertook a contingent valuation study on the value of biodiversity and found that one quarter of respondents held lexicographic preferences. These respondents stated that animals/ecosystems/plants should be protected irrespective of the costs, and refused to give a willingness to pay amount. Gowdy (1997) hypothesises that biophilia may be part of the reason for lexicographic ordering of preferences for environmental protection. Biophilia has been defined by Wilson (1993, 31) as “the innately emotional affiliation of human beings to other living organisms”.
The expression of lexicographic preferences shown in these two studies does present problems for the assessment of non-use values. If respondents do not acknowledge the instrumental benefits that ecological attributes provide and see these attributes as possessing intrinsic value then they may be unwilling to measure these values in monetary terms. A common approach to this problem in contingent valuation survey design is to include questions about respondent’s attitudes towards the environment to assess if they hold ethical beliefs. Following bid elicitation, questions can also be included to determine if the respondent holds a lexicographic position by explicitly asking if they felt unwilling to place a monetary value on wetland preservation. The responses to these questions will allow the researcher to estimate the extent to which respondents hold lexicographic preferences.

5.7.3 Consumers acting as Citizens

Blamey et al., (1995) and others often argued that when answering contingent valuation questions, individuals no longer act as consumers trying to maximise utility, that is as egoists, but take on the role of the citizen where other ethical considerations predominate (Sagoff, 1988; Blamey et al., 1995). Sagoff (1988) argued that citizen choices should be seen as separate from and ethically superior to consumer choices and that environmental problems are “primarily moral, aesthetic, cultural and political” (Sagoff, 1988, 6). A similar argument is presented by Blamey et al., (1995) who reviewed the social/political attitudes of respondents elicited in a study of forest management in Australia. They concluded that responses to contingent valuation questions concerning environmental preservation were dominated by citizen judgements about desirable social goals rather than individual consumer preferences. They suggested that further research needs to be undertaken to understand the behavioural motivation involved in response making (Blamey et al., 1995).

In contrast, Rolfe and Bennett (1996b) argue that as economics is concerned with the range of preferences which people hold then these do not need to be confined to self-interested motivations. If an individual’s action makes someone better off then the resulting increase in utility is the benefit to that individual as well as the recipient. That this action may also make someone else better off only increases the benefit
(Rolfe and Bennett, 1996b). In this case it is not necessary to determine which motivation results in the associated change in utility. Thus, Rolfe and Bennett (1996b) argue that citizens’ values are part of the same factors that drive the willingness to pay variable. The treatment of attitudinal responses (as identified by citizen values) as independent of willingness to pay responses is rejected by Rolfe and Bennett (1996b).

Rolfe and Bennett’s (1996b) argument can be extended to the case of market goods. Rolfe (1998) notes that moral and ethical frameworks also impact on the choices individuals make for standard consumer items and economic studies rarely distinguish the source of satisfaction when assessing the demands for amenities. There is no simple dichotomy between preferences and ethical and moral frameworks. Preference formation includes the consideration of satisfaction or regret in terms of complying with ethical frameworks (Rolfe, 1998). For this reason, the expression of moral or citizen values within a given willingness to pay amount does not alter the validity of those bids.

For the case of ecological attributes which provide the basis for life support of all humanity, it seems conceivable that a respondent may include the value to society and themselves in their willingness to pay responses. If information is provided to respondents about a range of ecological and economic values of the environmental resource the resulting mean willingness to pay for each case may also reveal if respondents are acting as consumers or citizens. It could be considered that if the monetary value given for economic use values exceeds that for non-use or ecological value then the respondent is acting in self-interest as a consumer. If, however, the reverse is found respondents may be expressing citizen values. Questions can also be asked about respondent attitudes prior to the willingness to pay question to assess if they hold citizen or consumer preference with respect to wetland preservation.
5.8 SUMMARY AND CONCLUSION

The chapter has explored the potential for the contingent valuation method to measure ecological value if information about the ecological values of an area is included in the survey. The extent and impact of information in previous contingent valuation studies have been reviewed with respect to psychological effects, service and characteristic information and information about ecological attributes. Each stage of the contingent valuation survey design has been examined to identify the most suitable method for valuing ecological goods and to avoid problems that have been identified in previous studies. In summary, areas that have presented difficulties include the amount of information to provide to respondents, the choice of payment vehicle, the bidding method, the choice of measure and problems related to the expression of moral preference. By reviewing previous contingent valuation studies a number of strategies are suggested to overcome or avoid these difficulties.

Information provided to respondents should not evoke an emotional response and should be limited to verifiable facts. To ensure that respondents are familiar with the good and that it is relevant to them, the geographical scope of the survey should be limited to areas adjacent to the site which is being assessed. A consensus on the amount of information to provide to respondents has not been reached in the literature. However, from studies which have indicated an information overload, it is suggested that introducing more than four or five issues will not increase willingness to pay correspondingly. Studies that have directly assessed the value of ecological attributes indicate that respondents are capable of assimilating ecological information. This provides further impetus for attempting to assess ecological value using the contingent valuation method.

The choice of payment vehicle depends on the context of the survey and must be neutral to avoid protest bids related to vehicle itself. To assess ecological value in the Australian context it is suggested that either taxation or a voluntary trust fund would be the most appropriate vehicle. The final choice would need to be made in light of information gained from focus groups and pre-testing to determine public attitudes to either method.
Before being asked for a willingness to pay bid respondents must be reminded of their income constraints. The availability of substitutes may also alter willingness to pay. However, in the case of ecological attributes it is difficult to determine which areas could be considered substitutes due to a lack of ecological information.

There is a variety of methods available for eliciting bids from respondents. Problems of starting bid bias, strategic bias and lack of familiarity for the good using the closed-ended, and open ended formats suggest that the iterative bidding is the most appropriate for eliciting bids. There is also considerable debate within the literature as to the correct measure of value. Property rights will determine if respondents are asked either their willingness to pay or willingness to accept compensation. If a wetland is considered to be a public good and there is no one agency responsible for causing harm then the willingness to pay is the correct format.

Other problems that have been identified in previous contingent valuation studies relate to respondents expressing moral rather than self-interested preferences. If respondents express a value for a wider class of good than the one being assessed, problems of scope or embedding are said to arise. This can be avoided by clearly defining the good and reminding respondents that they are only being asked to value that good. If respondents hold moral beliefs such that they are unwilling to trade off money for non-use values they are said to hold a lexicographic position. Thus, it is important to identify what motivates respondents in their bid formation. This can be achieved by asking questions about respondent’s attitudes towards the environment and follow-up questions subsequent to the bid elicitation question. Respondents that express citizen’s values when answering contingent valuation questions need not be considered a problem. As long as the bid amount given increases their utility, the assumptions of economic behaviour have not been violated.
To develop a contingent valuation survey that includes both ecological and economic information requires identifying both the ecological and economic values of the case study area. Consideration must also be given to the status and management of the area to provide a credible scenario to offer respondents. Chapter 6 identifies the ecological and economic values of the case study area, Moreton Bay. The current condition of the Bay and the present management arrangements are also reviewed to direct the scenario to be presented to respondents.
CHAPTER 6
THE ECOLOGICAL AND ECONOMIC VALUES OF MORETON BAY,
AUSTRALIA

The purpose of this chapter is to describe and link the ecological and economic values of Moreton Bay, Australia. Moreton Bay has been selected as the case study area to assess whether the contingent valuation method can capture ecological value. This chapter documents the available information that can accurately be provided to respondents in a survey. A physical description of the Bay and the associated wetlands is provided. The ecological values of the Bay, in terms of productivity, dependency and biodiversity, are outlined within the limits of available ecological knowledge. The economic values that can be assigned to the wetlands of the Bay are then given. Finally, the legal and jurisdictional frameworks for the management of the Bay are presented. It is within this framework that an accurate and plausible scenario can be developed for the contingent scenario of providing improved water quality in the Bay and for the collection of theoretical payments.

6.1 PHYSICAL DESCRIPTION

Location
Moreton Bay lies in southeastern Queensland, Australia, (27 20’ S, 153 10’ E), immediately east and extending north east and south east of Brisbane, the capital city of Queensland (Map 6.1). It is a basin forming a semi-enclosed estuarine bay, roughly triangular in shape, enclosed on the west by the mainland and on the east by the large sand islands of North and South Stradbroke Islands and Moreton Island. It is one of only two large estuarine bays in Australia which are enclosed by barrier islands of vegetated sands (Perkins, 1996). Moreton Bay covers an area of about 1523km² (Dennison and Abal, 1999, 23). The water body is approximately 80 km long, 35 km wide in the north but less than 5 km wide in the south. The deepest water, which exceeds 40 m, is found off Moreton Island (Hekel et al., 1978, 7). Moreton Bay has a large variety of landforms including reef, tidal flat, supratidal flat, beach, tidal creek,

Map 6.1 Location of Moreton Bay

Geology
The geological history of Moreton Bay has been dominated by a succession of sea level changes which have produced a succession of differing sedimentary environments (Stephens, 1992). In Quaternary times (the last 2 million years), the Bay has existed only intermittently as a marine environment as advances and retreats of glaciers and ice sheets have progressed (Hekel et al., 1978). While the sea has returned to be close to its present level about every 120 000 years, most of the time sea level has been much lower than at present, by as much as 150 m during the last glacial ice age (Stephens, 1992, 4). During these low sea level phases, the present Bay formed a terrestrial plain traversed by stream valleys of the ancestral Brisbane and Pine Rivers and their tributaries. Sea level began rising about 17 000 years ago and when the post glacial marine transgression ceased 6 000 - 6 500 years ago the sea attained a level of 1 to 1.5 m higher than the present level (Flood, 1984,127). Since then fluctuations of about 1 m are indicated (Hekel et al., 1978). Evidence suggests that a contemporary rise of sea level of the order of 1 mm/year has occurred during the past 40 years (Flood, 1984, 127).

Catchment
It was argued in Chapter 2 that wetlands should be viewed as a component within the catchment. The wetlands of Moreton Bay are strongly influenced by activities and processes within the catchment as well as activities in the Bay itself. Inputs from the catchment include nutrients and sediments, sewage and toxicants, which are, discharged into the waterways leading into the Bay and the Bay itself.

A number of catchments of several large rivers and smaller creeks rise from the mountain ranges in the west (the Lamington plateau in the south, north along the Great Dividing Range to D’Aguilar Range) and drain eastward from the mainland into Moreton Bay and Pumicestone Passage. The major rivers of the catchments, the Nerang, Pimpama, Coomera, Albert, Logan, Brisbane, Pine and Caboolture Rivers are illustrated in Map 6.2. The catchment area of Moreton Bay (21220 km²) is dominated by the Brisbane River catchment (13100 km²) and compared to the area of the Bay (1523km²) represents a ratio of catchment to Bay of about 14:1 (Dennison and Abal, 1999, 23).
Climate

Moreton Bay is situated on a biogeographical boundary separating the tropical from the more temperate areas and has a maritime subtropical climate (Dennison and Abal, 1999). The region is predominantly influenced by air moving from the southwest, over the continent, during the winter months and from the east over the Pacific Ocean during the summer and autumn months when monsoonal low pressure systems bring rain to the region. Seasonal rainfall leads to periods of high runoff and occasional floods (Dennison and Abal, 1999). During flood and high runoff periods the major rivers in the catchment deliver silt, mud, sand and pollutants into the Bay.

Tides

The pattern of tidal currents in Moreton Bay is clockwise. Currents have a major influence as the East Australian Current brings warm tropical water (along with larvae and seeds) to Moreton Bay. Tides enter mostly from the northern entrance or squeeze through the passages between the other sand islands. The volume of water entering on each flood tide is about 15% of the total volume of water in the Bay. However, 95% of the incoming water went out on the previous tide. Hence, pollutants in the Bay may take from 30 -300 days to be reduced by 90% (Brisbane River and Moreton Bay Wastewater Management Study (BR and MBWMS), 1997). The flood tide water does not mix evenly in the Bay. Turbid, nutrient rich water from the Brisbane and Pine Rivers is delivered to the poorly flushed areas of Bramble and Deception Bay and Pumicestone Passage. Dennison and Abal (1999, 32) describe residence time as the length of time that a parcel of water will stay at a certain location. Map 6.3 illustrates the predicted residence times in Moreton Bay and demonstrates the poorly flushed areas.

Map 6.3  Residence time (in days) based on mean annual wind (1997)

Source: Dennison and Abal (1999, 32)
The normal tidal range is 1.7 metres. As the coastline is gently sloping with large, flat sediment banks, at some points the edge of water shifts hundreds of metres between tides, exposing the mud flats at low tide. Although the Bay is generally protected from ocean swells, the prevailing south-east winds can generate waves of up to 1.5 metres.

6.2 WETLANDS OF MORETON BAY

The definition of wetlands used in this thesis includes the swamps, lakes, mud flats and mangrove forests found in Moreton Bay and also includes seagrass meadows in marine water which do not exceed a depth of six metres at low tide (ANCA, 1996). The Moreton Bay wetlands are considered to be significant at a state, national and international level. The wetlands within and adjacent to the Bay together with the permanent lakes of the sand islands provide a diverse and rich suite of wetland habitats (ANCA, 1996). The freshwater wetlands on the islands will not, however, be considered further as they provide a different (but significant) range of ecological values from those that are influenced by saline waters. The features of the coastal wetlands of Moreton Bay and the species they support are outlined below.

Seagrasses

Seagrasses are marine angiosperms (flowering plants) which occur in sheltered, shallow coastal waters of tropical and temperate zones around the world. Hyland et al., (1989) point out seagrass meadows in the Moreton region occur in the intertidal zone and adjacent shallow subtidal zone. The seagrasses are susceptible to disturbance in these areas from changes in water quality, which result from increased nutrients and sediments derived from the catchment (Walker et al., 1999). Species of seagrass occurring in Moreton Bay and adjacent estuaries include: Zostera capricorni, Halophila ovalis, Halophila spinulosa, Halophila decipiens, Halodule uninervis, Syringodium isoetifolium and Cymodocea serrulata (Hyland et al., 1989, 4).

Hyland et al., (1989) undertook a survey of seagrass between Coolangatta and Noosa between August 1987 and December 1987. Seagrass communities were classed
according to the visually estimated ground cover as dense (50% ground cover), light
(10-50% ground cover) and sparse (10% ground cover). The occurrence of small
(0.5-3m diameter) patches of dense seagrass was also recorded. In the study area,
extending from the Broadwater to Caloundra there are approximately 14 100 ha of
seagrass meadows and another 11 200 ha of sparse or patchy seagrass (Hyland et al.,
1989, 42). Map 6.4 illustrates the distribution of seagrasses, mangroves and
saltmarsh in the Bay\textsuperscript{1}.

Map 6.4  Distribution of Seagrass, Mangroves and Saltmarsh in Moreton Bay


\textsuperscript{1} Map 6.3 is a colour copy of the map supplied in the contingent valuation survey of Moreton Bay.
It is noteworthy that the Wanga Wallen Banks is the southernmost limit of the range of *Cymodocea serrulata* and *Syringodium isoetifolium*. It is also the only place in the world where male flowers of *Cymodocea serrulata* have been found (Kirkman, 1975). Seagrasses occur mainly in the intertidal zone although Hyland *et al.*, (1989) reported that several large areas of either light or sparse growth occurred subtidally, mostly in less than 3 m of water at low tide. Seagrass was not found at depths greater than 10 m.

The distribution of seagrass within the Bay is considered an indicator of water quality. The depth of water in which seagrasses are able to grow is dependent on the penetration of light into the water (Abal and Dennison, 1996). Light availability is in turn affected by the amount of suspended sediments that flow down the rivers and into Moreton Bay. Inputs of dissolved nutrients can also lead to a proliferation of algae on seagrass leaves and stems which also reduces light availability to the seagrass itself (Walker *et al.*, 1999). Changes in seagrass depth range in Moreton Bay have been measured since 1992 and have been used to infer changes in water quality (Abal *et al.*, 1998). Between 1992 and 2000 there were declines in seagrass beds around the Bay, particularly in the southern and western areas (Grice *et al.*, 2000). Seagrasses had been lost at Bramble Bay prior to the 1992 survey, and were lost from southern Deception Bay in 1996, in the southern bay, near the mouth of the Logan River in 1995 and at the Northern Broadwater in 1999. It is estimated that since European settlement approximately 20% of the total seagrass area in Moreton Bay has been lost (Abal *et al.*, 1998, 269). These changes are illustrated in Map 6.5.

Once destroyed, seagrass regrowth and colonisation is very slow or often does not occur due to changes in the sedimentary environment and the poor dispersal capabilities of most seagrass species (Poiner *et al.*, 1992). For example, seagrass beds in Raby Bay had been described as rich stands in 1975 worthy of preservation but following the development of the canal estate only small stands were found by Hyland *et al.*, (1989).
Mangroves

Mangroves are the most conspicuous component of the wetlands in Moreton Bay. The term mangrove refers to tree, shrub and heath communities that occupy the land between the upper and lower tidal limits. They occur along the shoreline, on the banks of creeks and streams, in swamps, in areas of silt and mud at creek and river mouths and also around islands and in bays and other low-lying areas (Dowling, 1986). The mangrove communities of Moreton Bay are not continuous along all the shoreline but occur in general in the estuarine areas and other areas protected from strong wave action such as Pumicestone Passage, Caboolture River-Deception Bay, Hay’s Inlet-Pine River, Brisbane River mouth, Mud Island, St. Helena Island, Green Island and the bay south of Victoria Point (Dowling, 1978). (See Map 6.4). In 1987, the total area of mangroves between Coolangatta and Caloundra was estimated to be 14 457 ha (Hyland and Butler, 1988, 4).
Seven species of mangroves occur in Moreton Bay and its rivers. These are *Aegiceras corniculatum, Avicennia marina, Bruguiera gymnorrhiza, Ceripos tagal, Excoecaria agallocha, Lumnitzera racemosa* and *Rhizophora stylosa* (Dowling, 1986). These communities occur from just below low tide to near the high tide mark and penetrate into rivers and creeks (Abal *et al.*, 1998). *Avicennia marina* is the most widespread species and grows along river banks, the bay islands and along the intertidal fringes of the Bay (Abal *et al.*, 1998). The farther that it occurs towards the upper tidal limits the smaller and less vigorous it becomes (Dowling, 1986). *Avicennia marina* is replaced by *Aegiceras corniculatum* in the upriver, low salinity reaches of the rivers and creeks (Abal *et al.*, 1998). Figure 6.1 illustrates the distribution and growth of mangroves in Moreton bay and its rivers (Abal *et al.*, 1998, 273).

Figure 6.1 The zonation of mangroves in Moreton Bay and its rivers

![Figure 6.1 The zonation of mangroves in Moreton Bay and its rivers](image)

Many of the species occurring within Moreton Bay are at or near their southern limit of distribution. For example, *Ceriops tagal var. australis* and *Lumnitzera racemosa* are at the southern limit and *Rhizophora stylosa, Bruguiera gymnorhiza* and *Excoecaria agallocha* extend slightly further south into the Northern Rivers District of New South Wales (Dowling, 1978).

Until the 1970’s the mangroves in the study area had remained relatively undisturbed, however, due to development pressure from construction of port and harbour facilities, canal developments, marina developments and airport construction there has now been extensive clearing of mangroves along the western shoreline (Department of Environment and Conservation (DEC), 1989). It is estimated from 1974 to 1987, 1355 ha of mangroves were lost, while there was a natural accretion of 114 ha, resulting in a net loss of 1234 ha or approximately 10% of the mangrove area existing in 1974 (Hyland and Butler, 1988, 7).

**Saltmarshes**

The term ‘saltmarsh’ includes those areas between mangroves and terrestrial vegetation, which contain samphire sedges, salt couch, bare saltflats and stunted mangroves. As saltmarshes and claypans occupy the supralittoral fringe they are only inundated by sea water during high water of spring tides (Cribb, 1978).

The most extensive areas of saltmarshes in Moreton Bay are at the mouth of the Caboolture River, Boondall, Tingalpa Creek, Hilliard’s Creek, Eprapah Creek and in the southern bay islands. (See Map 6.4). In general, meadows of saltwater couch (*Sporobolus virginicus*) and *Juncus krausii* with scattered specimens of grey mangrove occupy the uppermost part of the intertidal area (Hyland and Butler, 1988). Seaward of this the plant community is dominated by fleshy chenopod shrubs (eg. *Suaeda australis* and herbs *Salicornia quinqueflora*). Following this is a band either devoid of vascular plants or only sparsely colonised by *Suaeda* and *Salicornia* (Cribb, 1978).

Salt marshes are known to provide important habitat for birds, particularly waders, and may also function as a source or organic material for detrital food chains (DEC,
Saltmarshes in Moreton Bay also provide habitat for fish (Thomas and Connolly, 2001). Hyland and Butler (1988) estimated that there were 5 010 ha of saltmarsh-claypan between Coolangatta and Caloundra. Between 1974 and 1987, 10.5% of the saltmarsh-claypans of this area were reclaimed. There was no evidence of any natural offsetting increase in this period (Hyland and Butler, 1988, 7). The remaining saltmarshes along the western foreshore are under threat from pressures for land fill, recreation and rubbish dumps.

**Corals**

Moreton Bay supports the southernmost true coral reefs on the east coast of Australia. (True in this sense refers to the fact that the corals grow on reefs of organically derived calcium carbonate and not on a substrate of terrigenous origin) (Johnson *et al.*, 1993). The coral communities are comprised of 45 species from 15 genera and seven families (Johnson *et al.*, 1993, 131). However, as noted earlier, although these corals may occur in less than six metres of water, thus falling within the wetland definition, due to the difference in the ecological functioning of these systems and other coastal wetlands they are not considered further.

**6.3 ECOLOGICAL VALUES OF THE WETLANDS OF MORETON BAY**

The diversity of coastal wetland ecosystems of seagrass, mangrove, saltmarsh and saltpans found in Moreton Bay provide all the attributes of ecological value as defined within this thesis, in Chapter 3. That is, productivity, dependency and biological diversity and organisation. Each of these attributes is outlined to demonstrate the ecological values, of Moreton Bay.
6.3.1 Primary Productivity

The primary producers of Moreton Bay include benthic microalgae, macroalgae, mangroves and seagrasses (Dennison and Abal, 1999). The fixation of carbon by these marine plants provides essential food for many species and provides structural habitat for the refuge and resting of faunal species. The estimated areal extent, productivity and biomass of these plants as assessed within the area of the ‘Moreton Bay Study’ are given in Table 6.1.

Table 6.1 The productivity and biomass of marine plants as identified by the Moreton Bay Study

<table>
<thead>
<tr>
<th>Plant Type</th>
<th>Areal Extent (kms²)</th>
<th>Biomass (tC)</th>
<th>Productivity (tC yr⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Benthic microalgae</td>
<td>315</td>
<td>1800</td>
<td>520 000</td>
</tr>
<tr>
<td>Mangroves</td>
<td>103</td>
<td>23000000</td>
<td>59 000</td>
</tr>
<tr>
<td>Seagrasses</td>
<td>181</td>
<td>11 000</td>
<td>36 000</td>
</tr>
<tr>
<td>Macroalgae</td>
<td>106</td>
<td>820</td>
<td>20 000</td>
</tr>
</tbody>
</table>


Benthic microalgae may be the most productive marine plants in Moreton Bay with the highest rates of productivity found in the Bay’s shallow coastal areas (Dennison and Abal, 1999). The greatest biomass of benthic microalgae is found at depths of less than five metres and is therefore within the definition of wetland area. Benthic microalgae are most common in the top 3 cm of aquatic sediments (Dennison and Abal, 1999). Recent work suggests that they may be a major food source for benthic feeders such as prawns, other crustaceans, bivalves and polychaete worms (Currin et al., 1995). In shallow water benthic microalgae may therefore form the basis of coastal food webs and exert a significant influence on sediment biogeochemistry through nutrient processes (Dennison and Abal, 1999, 164). They also increase sediment stability (Dennison and Abal, 1999, 162).

Table 6.2 summarises the contribution to total Bay primary productivity of seagrasses, mangroves and phytoplankton. From this table it is estimated that seagrasses and mangroves provide for nearly one-third (32.1%) of the primary

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2 The spatial scale of the Moreton Bay Study (Dennison and Abal, 1999) is less than the area defined within the Moreton Bay Marine Park and the areas included as part of the Moreton Bay Ramsar Site.
production occurring in Moreton Bay. Thus, it can be inferred that these wetland species exhibit the attribute of primary productivity which in turn is an attribute of ecological value.

### Table 6.2 Primary productivity of seagrasses, mangroves and phytoplankton in Moreton Bay

<table>
<thead>
<tr>
<th>Area (ha)</th>
<th>Primary Productivity (tonnes C/day)</th>
<th>Contribution total Bay primary productivity (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Seagrasses</td>
<td>25 000</td>
<td>105</td>
</tr>
<tr>
<td>Mangroves</td>
<td>13 604</td>
<td>96</td>
</tr>
<tr>
<td>Phytoplankton</td>
<td>140 000</td>
<td>424</td>
</tr>
</tbody>
</table>


#### 6.3.2 The provision of dependent habitat

The seven species of seagrass in Moreton Bay play an important ecological role in providing habitat for dependent species. For example, they provide habitats and nursery areas for commercially important marine organisms, food for dugongs and turtles, nutrient cycling and high nitrogen fixation (Poiner et al., 1992). They also trap and stabilise sediments (Poiner et al., 1992). For example, nine species of prawns and 31 other species of small shrimps lie in and around seagrass beds (Davie et al., 1998, 55). These species live almost to maturity in the shallow waters of the seagrass meadows and mangrove swamps before they migrate to spawn in deeper waters. The juvenile stages of many species are particularly abundant in seagrass meadows and Pollard (1984) has reported that the ten dominant fish families that commonly occur in seagrass habitats in the Australian region are Sygnathidae, Monocanthidae, Gobiidae, Scorpaenidae, Sparidae, Tetraodontidae, Terapontidae, Apogonidae, Ambassidae and Kyposidae. Young (1978) has identified seagrasses as being important for postlarval stages of penaeid prawns when they leave the plankton to adopt a bottom dwelling lifestyle. The seagrass meadows of Moreton Bay are important nursery grounds for the commercially important tiger prawn (*Penaeus esculentus* and *P. semisulcatus*), endeavour prawns (*Metapenaeus ensis* and *M. endeavouri*) and greasyback prawns (*M. Bennettae*) (Young, 1978). Other commercially important species that utilise the seagrass beds during their juvenile phase include the blue swimmer crabs (*Portunus pleagicus*) whiting species (*Sillago*...
ciliata and Sillago maculata) and snapper (Pagrus auratus) (Moreton Bay Task Force, (MBTF) 1997).

Some animals such as green turtles (Chelonia mydas) and dugongs (Dugong dugon) feed directly on seagrass, however the meadows provide a variety of other food sources such as epiphytes, detritus, microfauna and benthic fauna. Epiphytes such as macroalgae, microalgae and bacteria provide an important source of food for algal grazers including gastropods, amphipods, mysids, polychaetes, copepods, nematodes and forminiforans (Orth and Von Montframs, 1984). These animals are then fed upon by larger animals such as prawns (Metapenaeus and Penaeus species) and some fish such as whiting (Sillago spp.) (Poiner et al., 1992). Thus, seagrass is the base of the food chain with bacteria converting the indigestible plant food into food which is more readily assimilated (Poiner et al., 1992; Davie et al., 1998).

Detached seagrass leaves contribute organic material to the detrital pool (Klumpp et al., 1989). This is a source of food for bacteria and diatoms, which are fed on by other animals such as the greasyback prawn (Metapenaeus bennetae). Other animals associated with seagrass consume a variety of food, for example, the tiger prawn species (Penaeus esculentus), feed on polychaetes, gastropods, algae and seagrass seedcases (Hyland et al., 1989). In addition, seagrass beds are very important sites for fixing nitrogen via nitrogen-fixing bacteria, in the marine environment, with 0.2 to 0.4 kgN/ha/day being fixed in Moreton Bay seagrass sediments (Poiner et al., 1992, 43).

The seven species of mangroves in Moreton Bay act as nursery areas, prevent coastal erosion, provide habitat for many faunal species, provide landscape value and act as roosting sites for wader birds. The importance of mangroves is related to their detritus being an important food source. Also, the structural complexity of the plants and their pneumatophores and fallen branches provide refuge for smaller animals from predators and thus favour juveniles (Dennison and Abal, 1999, 177). For example, more than 30 species of crabs inhabit the mangrove forests (Davie et al., 1998, 26). Sesarmid crabs carry leaves into their burrows as well as consuming them on the forest floor (Robertson, 1991). Their burrowing activity displaces and mixes a large percentage of the top 20 –30 cms of mud every year. This brings organic
material to the surface and aids in the aeration of the mud. In north Queensland it has been demonstrated that sesarmid crabs process 32% of the annual leaf litter of Avicennia forests (Robertson, 1991, 439). Thus these crabs play an important role in the decomposition and export of particulate matter (mainly carbon) to adjacent habitats (Robertson, 1991, 439). The larvae of the crabs are also an important food source for newly recruited fishes (Robertson, 1991). Also, larger crabs found among the roots of the mangroves bring atmospheric oxygen and clean tidal water direct to tree’s root system (Davie et al., 1998, 27). For example, the largest of the portunid species of crab in the world, the mud crab (Scylla serrata) is found in the mangroves of Moreton Bay and is considered an important commercial species (MBTF, 1997).

The saltmarshes and saltpans adjacent to the mangroves are an integral part of the wetland system providing a source of material for detrital food chains and feeding and crucial roosting sites for wader birds. A recent study by Thomas and Connolly (2001) in the north of Moreton Bay, at Meldale in Pumicestone Passage has also demonstrated that the saltmarshes are utilised by fish species. Sampling was undertaken in both winter and summer when the saltflats were inundated and 15 species of fish from 9 families were caught (Thomas and Connolly, 2001, 278). Six of these species were of commercial importance (Thomas and Connolly, 2001).

In addition there are 285 species of macroalgae in the Bay found on rocky outcrops, and on mangrove trunks and pneumatophores, seagrass beds or in sediments (Dennison and Abal, 1999, 165). Cribb (1978) has noted fifty-five species of algae associated with the mangrove community of Moreton Bay. Algal films on leaves and the decay of leaves and branches supply nutrients both in situ, and where water currents carry them. This decay cycle forms part of the basis of the natural food chain of the Bay (DEC, 1989).
### 6.3.3 Biological Diversity

Moreton Bay lies in an area of transition between the southern (Peronian) and northern (Solanderian) faunas, which has produced a peculiar mix of both temperate and tropical species (Davie and Hooper, 1993). As a result, the Bay has a remarkably diverse fauna with relatively large numbers of endemic species (Dennison and Abal, 1999, 154). There are two main centres of biological diversity in the Bay. One is an inshore zone dominated by the estuaries and the other is the eastern region dominated by the ocean. Wetlands are present in both these regions. However, the greatest amount of information about the fauna of Moreton Bay is related to information about commercially important fish and crustacean species (Davie and Hooper, 1998, 329). The understanding of the underlying ecology of non-commercial species, many of which may play important roles in the ecology of the Bay, are less well known (Davie et al., 1998). Thus, it is not possible to document the organisation of the biodiversity of the Bay, although this is considered to be an important attribute of ecological value. However, estimates of the numbers of species found in the differing taxonomic groups indicate that Moreton Bay displays this important attribute of ecological value.

For example, it is estimated that there are over 3 000 species of free-living marine macro invertebrates living the Bay and 713 fish species (Davie and Hooper, 1998, 333). For example, there are 40 species of molluscs within the mangrove/saltmarsh zones of south-east Queensland and most of these are not found away from this environment (Davie et al., 1998, 27).

More than 273 species of birds from 65 different families have been recorded in Moreton Bay (Davie et al., 1998, 319). Moreton Bay is identified by the Australian Wader Studies Group as an area of international importance for 7 species of shorebirds and of national significance for 10 species of shorebirds (Watkins, 1993). Due to its sheltered nature, Moreton Bay is one of only four recognised sites of significance to wintering migratory birds along the eastern coast of Australia (ANCA, 1996). At low tide, birds can be seen feeding or resting on seagrass beds, sand banks and mud banks. Equally important are the high tide roosting areas which
includes mangroves, saltmarshes, saltwater couch grasslands, rock islands and the mainland foreshores.

More than 43 species of waders, including 30 covered under the JAMBA\(^3\) and CAMBA agreements, use the Bay. This includes 50 000 wintering and staging migratory waders which use the Bay during the non breeding season. The Bay is particularly significant for the wintering Eastern Curlew, *Numenius madagascariensis* (3 000-5 000 birds) and the Grey Tattler, *Tringa breviceps* (> 10 000 birds). Moreton Bay also has particularly large populations of water birds including cormorants and terns, white herons, storks, spoonbills, ibises and egrets (ANCA, 1996).

Moreton Bay also supports a large dolphin population. There are two species resident in the Bay, the bottlenose dolphin (*Tursiops truncatus*) and the Indopacific humpback dolphin (*Sousa chinensis*). The bottlenose is the more common and over 300 individuals have been identified in the Bay while 50 humpbacks have been identified (Preen et al., 1992). Also, as noted in Chapter 2, wetlands support a disproportionately high number of vulnerable and endangered species. In Moreton Bay, two vulnerable species that are directly dependent on the wetlands, particularly seagrasses as a source of food, are the dugongs (*Dugong dugon*) and green turtles (*Chelonia mydas*) (Poiner et al., 1992).

**Dugongs**

Moreton Bay is ranked among the top ten dugong habitats in Australia and together with the Gulf of Carpentaria and Torres Strait is considered one the most important areas for dugong in Queensland. Moreton Bay also represents the southern-most site in Australia with a large dugong population. Dugongs are of particular biological and conservation significance as they are the only herbivorous mammal that is strictly marine (Preen et al., 1992). It was estimated in 1996 that 800 dugongs (*Dugong dugon*) utilised Moreton Bay and that the populations were relatively

healthy (Lanyon, (1996) in MBTF, 1997). This situation is unique in that it is the only large dugong population known to survive in such proximity to a major city. Also, the dugongs of Moreton Bay tend to occur in relatively large herds of 50 to 200 while elsewhere they usually occur elsewhere only singly or in pairs (Preen et al., 1992). These marine mammals are over 3 m long and 400 kg in weight (Preen et al., 1992, 61). Dugongs mature between 7 to 10 years of age. They are slow breeding animals, and females usually bear only one calf and there may be 3 to 7 years between calving (Quinn, 1993). The principal food source of dugongs is seagrass and hence the most significant areas for the species in the Bay are Moreton Banks, Amity Banks and the adjacent channels in the eastern part of the Bay as shown in Map 6.6. There is also ready access to oceanic water, from this area, through the South Passage in winter when water temperatures fall below those favoured by dugongs within the Bay (Preen et al., 1992).

The dugong is listed as vulnerable under the Queensland Nature Conservation (Wildlife Regulation) 1994. A strategy, the Conservation and Management of Dugong in Queensland, 1999-2004 has been developed to achieve the protection and recovery of dugong in Queensland. Dugongs are threatened by a number of causes, including the loss of seagrass habitat, incidental boat strike, drowning in gill nets and otter trawl nets, and the disruption of feeding by boating activity. Longer term threats include the degradation of habitat and seagrass beds, urban runoff, sewage and other pollutants, including plastics which may be ingested. Grice et al., (2000) report that in 2000 there was the highest number of strandings and deaths recorded in the last 10 years. Although dugongs are protected by legislation, Aboriginal groups from the Moreton Bay region may catch a limited number under permit (Preen et al., 1992).
Turtles

Within Australia, Moreton Bay has the largest feeding concentration of endangered green turtles (*Chelonia mydas*) and the most significant concentrations of loggerhead turtles (*Caretta caretta*) (ANCA, 1996). The distribution of turtles in the Bay is illustrated in Map 6.6. Hawksbill turtles (*Eretmochelys imbricata*) are also resident in the Bay. However, no nesting of any turtle species takes place in Moreton Bay. The hawksbill and green turtle are considered to be endangered and the loggerhead is threatened in a world context. In Queensland, the loggerhead is listed as endangered under the *Nature Conservation Act (Wildlife Regulation) 1994* while the green and hawksbill turtles are listed as vulnerable.

Sea turtles are air breathing reptiles that have the capacity to remain submerged for lengthy periods (up to 60 minutes). Turtles spend the majority of their life in water on feeding grounds in low densities. Turtles take between 20 and 50 years to reach
sexual maturity and breed every two to eight years (National Research Council, 1990). Mature turtles migrate every two to eight years to nesting areas, which may be thousands of kilometres from the feeding ground (Miller, 1997). After egg laying they return to their feeding grounds and in four to six weeks the young turtles hatch and disperse into the sea. Green turtles are the most common turtle in Moreton Bay (60-65%) with a resident population of all sizes from small immature to mature adults of both sexes (Read et al., 1996, 262). Green turtles are the only herbivorous sea turtle and the most abundant vertebrate consumers of seagrass (Brand Gardner et al., 1999). They also feed on algae and occasionally jellyfish. Loggerhead turtles have an almost entirely carnivorous diet of crabs, crustaceans, molluscs, sponges, jellyfish and fish, while the hawksbill turtle is an omnivore eating a range of the above foods.

The ingestion of marine debris, including plastic, by turtles is known to occur but the effects are unknown (National Research Council, 1990). In Moreton Bay turtles are at risk of being struck by the propellers of boats, especially in the South Passage area (Dennison and Abal, 1999). They also face the threat of being entangled in trawler’s nets and drowning (Lutcavage et al., 1997). Habitat destruction poses a further threat for green turtles (Lanyon et al., 1989). Internationally, one of the greatest threats to turtles is from harvesting (Lutcavage et al., 1997). However, in Moreton Bay only indigenous groups are permitted to hunt turtles.

6.4 ECONOMIC VALUES OF THE WETLANDS OF MORETON BAY

The ecological values of the wetlands of Moreton Bay provide a range of goods and services for human benefit. The wetlands provide an aesthetic landscape for a range of recreational activities and have cultural and historical significance. They also provide a range of more tangible benefits such as fishery resources, waste assimilation and shoreline protection. Figure 6.3 illustrates the links between the two sets of values in Moreton Bay following the framework introduced in Chapter 2. Direct use values include commercial and recreation fishing and recreation. Indirect use values include the waste assimilation abilities of the wetlands and the shoreline protection they provide. Non-use values include bequest values, which are well articulated by the indigenous inhabitants, and the existence value of the vulnerable
species, turtles and dugongs that depend on the Bay. The purpose of reviewing these values is to facilitate the development of an accurate information set which can be presented to respondents in the survey of the case study area.

Figure 6.2 Examples of the relationship between the ecological and economic value of Moreton Bay

![Diagram of ecological and economic value components]

- **PROCESSES**
  - Photosynthesis
  - Denitrification
  - Water circulation
  - Nitrogen Fixation

- **STRUCTURES**
  - Geomorphology and Hydrology of Bay, Soils, Seagrass, Mangroves, Saltmarsh and associated Fauna

- **FUNCTIONS**
  - Hydrological
  - Flood Water Control
  - Biogeochemical
  - Water Quality Maintenance
  - Ecological
  - Food web support
  - Ecosystem Maintenance

- **ATTRIBUTES OF ECOLOGICAL VALUE**
  - Productivity, Dependency, Biodiversity and Organisation

- **DIRECT USE**
  - Commercial and Recreational Fishing, Recreation, Education

- **INDIRECT USE**
  - Water Quality
  - Improvements
  - Shoreline protection and stabilisation
  - Option for future use

- **NON-USE VALUES**
  - Existence Value for dugongs and turtles
  - Bequest Values for future generations

**COMPONENTS OF TOTAL ECONOMIC VALUE**
6.4.1 Direct Use Values

The economic use benefits of Moreton Bay include port activities, extractive industries, commercial and recreational fishing, recreation, tourism and boating. Of these activities, commercial and recreational fishing, recreation and education and to a lesser extent boating are reliant on the wetlands of the Bay\(^4\). The extent of these activities is reviewed and estimation of their contribution to the economic value provided by Moreton Bay is given where information is available. There have been no studies undertaken in the Bay to estimate all the components of the economic value of the wetlands (McDonald and Brown, 1992).

**Commercial Fishing**

Quinn (1993) states that of the total commercial fish catch in Queensland, approximately 75% by weight and 80% by value is derived from species that spend part of their life in shallow marine habitats such as mangroves. Thus wetlands and their ecological values are directly linked to this economic activity. There are about 350 commercial fishing boats licensed and operating within the Moreton Bay region. This includes 150 otter trawlers, 80 net fishing boats, 30 beam trawlers and 90 crabbing boats (MBTF, 1997, 50). In Queensland about 6000 people are directly employed in the commercial catching sector and there are 1150 people directly employed in the Moreton Bay region of whom 800 work within the Bay. There is also a significant workforce employed indirectly in fishery-related industries (processing and marketing) and industries servicing the fishing fleet (manufacturing, slipways, chandlers and fuel suppliers) (MBTF, 1997).

The most significant aspect of the fishery is that Moreton Bay accounts for only 3% of the State’s coastline but in 1988 it was estimated that Moreton Bay produced about 12.2% of the total fish products by value in Queensland estimated at $33m wholesale (Quinn, 1993, 49). Further figures for individual regions in the state are not presently available but it is estimated that the total value of wildcaught fish species for Queensland in 1999-2000 was $177.4m (Australian Bureau of Agricultural and Resource Economics, 2001, 20). If Moreton Bay maintained 12.2% of this total the annual value of the catch for Moreton Bay in 1999-2000 would be

\(^4\) Port activities and extractive industries are dependent on destroying wetlands.
equal to $22.175m wholesale. It is estimated that this value doubles by the time it reaches retail/export level (Quinn, 1993, 6).

However, these figures exceed the estimated value from the Department of Primary Industries data. Estimates of the catch and value of commercial fishing in Moreton Bay using these data as presented by Brisbane River and Moreton Bay Wastewater Management Study (BR and MBWMS) (1998a) for the years 1994–1997 are given in Table 6.3. However, the report suggests that the catch for 1997 was below normal and that the reliable long-term average value would be $13m wholesale or $33/annum retail (BR and MBWMS, 1998a, 16).

Table 6.3 Annual catch and value of commercial fisheries in Moreton Bay

<table>
<thead>
<tr>
<th>Year</th>
<th>Size (t)</th>
<th>Value ($'000)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1994</td>
<td>2 397</td>
<td>11 043</td>
</tr>
<tr>
<td>1995</td>
<td>2 644</td>
<td>12 018</td>
</tr>
<tr>
<td>1996</td>
<td>2 931</td>
<td>14 454</td>
</tr>
<tr>
<td>1997</td>
<td>2 003</td>
<td>9 452</td>
</tr>
</tbody>
</table>


Aquaculture for oysters also occurs in 120 hectares of Moreton Bay with an estimated value of $500,000 (BR and MBWMS, 1998a, 19). There are also 10 land-based aquaculture farms producing $7m worth of prawns annually (BR and MBWMS, 1998a, 20). However, as these operations occur on land this value cannot be attributed to the wetlands of Moreton Bay. The collective and cumulative discharges of nutrient enriched waters do however present a threat in terms of water quality (Crimp, 1993).

**Recreational Fishing**

Recreational and subsistence fishing in Moreton Bay has always been significant to indigenous people and to the Europeans since first settlement. A survey conducted for the Queensland Fish Management Authority in 1996, throughout Queensland (5000 respondents) indicated that 33.2% of Queensland households are fishing households. That is, there was at least one member of the household over 15 years of age who went recreational fishing in the previous 12 month period. The typical
Queensland fisher is a male between the ages of 15-49 years. They do not target any particular species and prefer saltwater fishing. The predominant purpose for fishing was for recreation (95.9%). Of those 4.1% who fish for other purposes, 49.1% do so for food, and 4.8% engage in fishing competitions (MBTF, 1997, 94).

All forms of recreational fishing take place in Moreton Bay including shell fish collecting for bait and food, estuarine shore, surf beach, rock, reef, boat, game, spear fishing and crabbing. Quinn (1993) states that almost one third of the entire state’s recreational fishing effort takes place within Moreton Bay and from the fringing islands. It is conservatively estimated that 300 000 fishers spending 1.5 million fishing days catching 2 000 tonnes of finfish per annum (Quinn, 1993, 6). There have been a variety of attempts at valuing the annual expenditure of recreational angler for fishing and boating in Moreton Bay using a range of assumptions. These estimates are outlined below. Overall, it is suggested that $200m/annum is a conservative estimate of the value of recreational fishing in Moreton Bay (BR and MBWMS, 1998a). This figure would be above the consumer surplus derived just from fishing as the activity also includes elements of the wilderness experience, relaxation, and recreation (Quinn, 1993, 8).

The BR and MBWMS (1998a, 16) report states that there are 29,637 registered recreational motor boats within Moreton Bay and its catchment. However, only boats with engines greater than 4hp are required to be registered (Neumann and Hundloe, 1986). The Queensland Fisheries Service estimates that a third of the fishing population owns a boat, which based on Quinn’s (1993, 6) estimate of 300 000 fishers equals 100 000 boats using Moreton Bay (Department of Primary Industries, 2001). Previous estimates from the Great Barrier Reef of the estimated average expenditure per year on recreational fishing and boating was $3700 per vessel ($A1993) (BR and MBWMS, 1998a, 16). For the 29, 637 registered vessels this amounts to $109.6m (BR and MBWMS, 1998a, 17). If the higher estimate of boat ownership is used this equals $370m (100 000x $3700) ($A1993).

The Department of Primary Industries estimates that throughout Queensland the annual spending on fishing is $400 m and the value of fish caught is approximately $50m (BR and MBWMS, 1998a, 17). It is estimated that 46% of the population of
Queensland lives in the study area and if they spend an annual amount consistent with other Queensland anglers it is estimated that they spend approximately $184 m (BR and MBWMS, 1998a, 17). Estimates from other Australian states put the average annual expenditure per fisher on recreational fishing in South Australia as $750, Victoria $1000 and Western Australia $650 (BR and MBWMS, 1998a, 17). If the same pattern of spending occurs in Queensland it is estimated the 300 000 recreational fishers in Moreton Bay would spend $240 million (BR and MBWMS, 1998a, 17).

Fishing occurs throughout the Bay especially in the sheltered estuarine areas and easily accessed places like the Pumicestone Passage (Quinn, 1993). General access to the more remote areas is improving with increased ownership of four-wheel drive vehicles and private boats (Quinn, 1993). With the expected increase in population of the area and increases in income and leisure time, the total number of fishing days is also expected to correspondingly increase, as will the pressures on the resource (Quinn, 1993). Comprehensive data about recreational fishing catches in Moreton Bay are not available and reliable data on commercial fishers have only been collected since 1988. These data demonstrate that the commercial fishery in Moreton Bay is experiencing declines in total catches, stability in catch per unit effort and decreasing numbers of commercial operations (MBTF, 1997, 91). For the recreational fishery there have been declines in the catch per unit effort and size of whiting and bream at key locations. The reasons have not been determined (MBTF, 1997, 91). It is thought that the decline is not just from increased recreational effort but also from urban and industrial development in the region. Changes in fisheries as a result of seagrass loss in the western side of the Bay have not been investigated (Connolly et al., 1999).
Recreation, Landscape and Educational Values

The wetlands of Moreton Bay and the barrier islands of Moreton and Stradbroke Islands provide a sheltered environment for a large range of recreational water-based activities. These activities include swimming, motor boating, sailing, water skiing, sailboarding and diving. The proximity of Moreton Bay to the urban areas of southeast Queensland provides opportunities for foreshore use such as picnicking and walking. The backdrops of the wetlands of Moreton Bay offer a wealth of visual landscape units such as estuaries and small enclosed bays, open coastline, semi-developed bay islands, low tidal islands and scenic channels and wooded sand islands and smaller sand cays. Some tourists visit Moreton Bay to view the diversity of wildlife, such as wader birds, dolphin, turtles, dugongs and the humpback whales which pass the outer islands. It is estimated 2000 people visit Brisbane each year specifically to watch the migratory wader birds (ANCA, 1996).

The Bay is an important environmental and historical education resource for schools, university and government agencies. The Bay’s range of undisturbed ecosystems and its proximity to educational institutions facilitates this role. For example, there is a research station on North Stradbroke Island used by the local universities for environmental research. A number of government research facilities are also located adjacent to the Bay. There are also State run field study centres for educating children on coastal and environmental matters (ANCA, 1996).

6.4.2 Indirect Use Values

The wetlands also provide many indirect use benefits related to their role in the protection and stabilisation of the shoreline and in maintaining water quality. These benefits are more difficult to quantify due to the fact that they accrue to the public in general. There are no available estimates of the value of the wetlands in terms of protection and stabilisation of the shoreline. The value of the waste assimilation of the wetlands can be inferred from current expenditures that are being undertaken to treat sewage (replacement costs) to a level previously provided for by the wetlands.
**Water quality**

Wetlands maintain water quality through their ability to absorb the sediments and nutrients released into the Bay (Dennison and Abal, 1999, 91). However, the ability of wetlands to continue to absorb the sediments and wastes created by a rapidly expanding population are now at their limit in some areas of the Bay. For example, there has been a total loss of seagrass in the Bramble Bay and Southern Deception Bay areas (Dennison and Abal, 1999, 64). Mangroves still exist in these areas but their capacity to continue to filter and absorb nutrients and sediments may be exceeded in some parts of the Bay (Moreton Bay Catchment Water Quality Management Strategy Team, (MBCWQMST), 1998, 29). Much of the western border of the Bay is now considered unsafe for swimming (Dennison and Abal, 1999). The wastes discharged into Moreton Bay come from both point and non-point sources. These discharges are reviewed to illustrate the nature of the assimilative capacity of the wetlands that previously provided good water quality in the Bay.

**Sewage and Wastewater Disposal**

Waterways have traditionally been seen as suitable sites for the deposition of wastes and Moreton Bay is no exception. Sewage and industrial effluent represent the largest contribution to pollutant loads in the Brisbane River and Bay. There are 25 major sewage treatment plants and 7 major industrial wastewater treatment plants discharging directly into the Brisbane River and Moreton Bay catchment. About 117 gigalitres (GL) of sewage a year from sewage treatment plants flow either directly into the Bay or into watercourses close to river or creek mouths (BR and MBWMS, 1997, 29). Map 6.7 illustrates the location of sewage treatment plants (STPs) in the Moreton Bay catchment.
All Brisbane’s sewage is currently treated to secondary level (MBTF, 1997). The largest sewage discharge is at Luggage Point at the mouth of the Brisbane River. This treatment facility serves a population in excess of half a million and discharges approximately a third of the total point source nutrient loading to Moreton Bay (Dennison and Abal, 1999; Moss et al., 1992). The major pollutants from this discharge include biodegradable organic matter, plant nutrients, nitrogen and phosphorus, and a range of micro-organisms. The total point source loads of pollution from the Brisbane river estuary are estimated at 700 tonnes/yr of sediments, 1100 tonnes /yr of total nitrogen and 350 tonnes /yr of total phosphorus (BR and MBWMS, 1997, 29).

Given the decline in ecosystem health of the western side of the Bay work has commenced on the upgrading of sewage treatment plants to biological nutrient removal standard. The aim is to reduce nitrogen output to 5mg/L N which is considered a sustainable level of point source nitrogen loads (Dennison and Abal, 1999). This follows scientific studies which have indicated that nitrogen is the
limiting nutrient within Moreton Bay (Grice et al., 2000). The cost of upgrades already undertaken or planned to be undertaken by 2005 to reach this target is $116.5m (MBCWQMST, 1998). For example, the Luggage Point STP has upgraded its nutrient removal standard which has led to a reduction from 30mg/L N to 8mg/L N at a cost of $24m (pers comm. Brian Perrot, BCC, 8/6/01). It is estimated that the cost of upgrading the Luggage Point and Oxley Creek STP to the target of potable re-use would cost an estimated $402.4m (1998-2020, 6% discount rate) (BR and MBWMS, 1998a, 43). The cost of these upgrades can be considered as the replacement cost of the waste assimilation functions previously supplied free by the wetlands before their assimilative capacity was exceeded.

Stormwater runoff

Changed land use patterns in the catchment have also contributed to increased sediments and nutrients from non-point sources. Before European settlement, the catchment area for Moreton Bay contained 93% forest and 7% of other natural areas. Now, the catchment has been cleared with grazing accounting for 75% of land use, cropping 3%, forest and other natural areas 14% and urban uses 7% (BR and MBWMS, 1997). The agricultural sector contributes wastes from over 1.5 m animals as well as increased sediment and fertiliser run-off (Dennison and Abal, 1999). It is estimated that rural/agricultural run-off contributes as much as urban run-off in terms of total nutrients and total suspended solids (Abal et al., 1998). Suspended solids directly impact on the viability of seagrasses and limit the growth of phytoplankton (Grice et al., 2001). Also, pollutants associated with sediments such as heavy metals, pesticides, effluent and herbicides can leave water toxic to fish or to the organisms on which fish feed (Moss et al., 1992). As virtually all pollutants enter on the western side of the Bay, pollution problems are most likely to arise in this section which, as discussed, is a depositional environment. Progressive clearance and urban development of catchments will continue to increase the mud loads from the catchments (Moss et al., 1992).
Option Value
Given the large number of uses provided by the wetlands of Moreton Bay it would be expected that individuals would be willing to pay to preserve the wetlands to retain the option of using them in the future. An estimate of this value has not been undertaken in Moreton Bay. Expenditures aimed at preserving the wetland resources for future use may be an indication of this option value. Where wetlands have been lost the option for future use has also been lost. However, it is unknown if option value was considered when allocating wetlands for other uses such as airport and port expansion.

6.4.3 Non-use Values

Estimations of non-use or passive value can be made in relation to wilderness areas or for species that one might never expect to visit or see. The value is related merely to the existence of these species (existence value) and the knowledge that they will also exist for future generation (bequest value). As noted in Chapter 4, non-use values can only be measured using stated preference techniques. However, there have not been any studies of this nature previously undertaken in the Bay to determine these values (McDonald and Brown, 1992). Nevertheless, from other studies it is suggested that existence values may be held for the vulnerable species in Moreton Bay. The expression of bequest values is well-articulated for the indigenous inhabitants of the area and may extend to non-indigenous inhabitants as well.

Existence value of Endangered Species in Moreton Bay
Loomis and White (1996) have reviewed a number of studies determining the existence value of endangered species. They state that threatened and endangered species which have previously been valued in terms of their non-use values includes grizzly bears, sea otters, bald eagles, sea turtles, humpback whales and gray wolves. Loomis and White (1996) suggest that humans appear to only value ‘charismatic megavertbrates’ but may include an implicit valuation for the underlying ecosystem. From an ecological viewpoint, these higher order species may not be crucial for the continued functioning of an ecosystem. In Queensland, species are listed as rare,
vulnerable or endangered in the *Nature Conservation (Wildlife Regulation) 1994*. This list includes the green turtles and dugongs found in Moreton Bay. If these species are considered valuable by humans due to the knowledge that they exist then these species can be said to provide non-use values. Therefore, the existence value of Moreton Bay could be measured by asking respondents to a survey if they are willing to pay to protect the habitat (seagrasses) that dugongs and turtles are dependent on.

*Bequest*

If current generations consider it is important to preserve the wetlands of Moreton Bay for future generations then they will hold bequest values for this resource. This value is perhaps expressed in the shared vision of stakeholder in the area that “Moreton Bay and its waterways will, by 2020, be a healthy ecosystem supporting the livelihoods and lifestyles of residents and visitors” (Dennison and Abal, 1999). Although, this vision includes only the next generation. Bequest values are more explicitly stated by the indigenous groups of the region. The groups who have traditionally lived in Moreton Bay (*Quandamooka*), include the Ngughi, Noonuccal and Koenpul people of Moreton Island (*Mulgumpin*) and North Stradbroke Island (*Minjerribah*). Other groups living adjacent to Moreton Bay include the Ngarangwal, Gubi and Gubi Gubi people. For these groups, environmental management practices are based on the respect, value and care of their country and resources (Tripcony and Anderson, 1996). The view of the Aboriginal people is that they belong to their country. The country is their ‘mother’ which provides them with their needs for survival (Tripcony and Anderson, 1996). The *Quandamooka* people have currently lodged a land claim on Moreton Bay through the National Native Title Tribunal. The basis of their claim is the “protection and preservation of *Quandamooka* for future generations” (Tripcony and Anderson, 1996, 23). This view exemplifies the concept of bequest value and suggests that this value is held for Moreton Bay, at least by the indigenous groups of the region.
6.5 THREATS TO THE ECOLOGICAL AND ECONOMIC VALUES OF MORETON BAY

The diversity of landscape features and the warm sub-tropical climate have made southeast Queensland a desirable place to live. As a result the Moreton region\(^5\) is experiencing the highest population growth in a metropolitan region in Australia. Almost two-thirds of the population of Queensland live in the region (Environmental Protection Agency (EPA), 1999a). In 1996 the population was estimated to be 2.2 million and it is expected to reach between 3 to 3.2 million in 2011 (EPA, June 1999a). The Moreton Region is the third or fourth fastest growing large urban region in the developed world over the past half century (Skinner et al., 1998). Brisbane is also the fastest growing capital city in Australia in the past decade (Skinner et al., 1998). It is estimated that the population in the Moreton region will double to reach 3.9 million by 2031. The Brisbane region can expect population to increase from nearly 1 million to 2.44 million in the same period (MBTF, 1997, 65). The principal component in growth has been internal migration (Skinner et al., 1998).

The associated intensive development in the region required to house this population is predominately occurring in proximity to the coast and thus has the potential to impact on beaches, estuaries, waterways and wetlands of the region (EPA, June 1999a). While the attributes of the Moreton Bay region attract this increasing population, it is argued here that this pressure poses the greatest threat to the resources of Moreton Bay. Threats from urban development include coastal and riparian development with increased effluent discharges and urban stormwater runoff (Quinn, 1993). The wetlands of Moreton Bay are considered a waterfront resource and it is likely that there will be further pressure to convert them to other uses such as urban dwellings, canals\(^6\), marinas and ports. This conversion may result in the

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\(^5\) The Moreton Region is defined as stretching from Noosa in the north to the New South Wales border in the south and extending westwards from the coast to the Great Dividing Range. This area is slightly larger than the Moreton Bay catchment area (Skinner et al., 1998).

\(^6\) There are now over 60 kilometres of canals and 4500 canal front properties on the Gold Coast (Nott, 1990) and over 200 completed major canal systems between Southport and Mooloolaba. As Johnson and Williams (1989) stated it, there are now canal developments on almost every available natural water body between Noosa and Byron Bay (NSW).
further loss of wetlands. The demand for housing is also expected to lead to an associated increase in extractive dredging\textsuperscript{7} from the Bay. Catchment degradation is likely to result due to vegetation removal, fertiliser use and soil erosion (Quinn, 1993).

Capelin \textit{et al}., (1998) argue that the potential acidification of estuaries due to the exposure of acid sulfate soils presents yet another threat resulting from development in the coastal strip. Acid sulfate soils are soils that contain iron sulfides, which when exposed to air, oxidise to form sulfuric acid. Sammut and Lines-Kelly (1996) note that along the Australian east coast these soils occur in areas with an elevation of less than five metres, in most mangrove and salt flat areas, and in river valleys. Capelin \textit{et al}., (1998) state that acid sulfate soils have been identified in the Moreton Bay region. The short term effects of sulfuric acid moving into local waterway and inshore areas are fish kills, fish disease, mass mortalities of microscopic organisms, increased light penetration owing to water clarity, loss of acid-sensitive crustacea and destruction of fish eggs (Sammut and Lines-Kelly, 1996). Sammut and Lines-Kelly (1996) identify the long term effects include loss of habitat, alterations to water plant communities, reduced spawning success owing to stress, reduced food resources, growth abnormalities, reduced growth rates, changes in food chain and web, damaged and undeveloped eggs, and higher water temperature owing to increased light penetration. Mosquito control for populations living next to the intertidal zone is a public health problem for local authorities, which in turn impacts on the environment\textsuperscript{8}.

The increased population will place extra pressure on recreational resources of Moreton Bay, with a corresponding increase in demand for marinas and tourist developments in the region. The population is also expected to have an increased demand for local seafood (MBTF, 1997). A larger population and hence workforce

\textsuperscript{7} The process of dredging sediments removes benthic flora and fauna and increases turbidity which smothers the sea floor, the remaining aquatic life and reduces light attenuation.

\textsuperscript{8} Recently, the main method of mosquito control has been to apply the organophosphate larvicides Abate\textregistered{} (BR and MBWMS, 1997). This has been shown to be highly toxic to crustacea and other aquatic fauna (MBTF, 1997). Physical modification of the wetlands through runnelling is also undertaken.
will encourage increased development of the port\(^9\) and industry which will have associated risks for the marine environment.

\textit{Lyngbya}  
A more direct threat to the wetlands and in particular seagrasses that is currently being experienced in Moreton Bay is from outbreaks of the toxic cyanobacteria, \textit{Lyngbya majuscula}. According to Grice \textit{et al.}, (2000) the outbreak of \textit{Lyngbya} is considered one of the greatest risks to ecosystem health in the eastern side of the Bay. \textit{Lyngbya} is a filamentous species forming strands of 10-30 cm long filaments. Dennison and Abal, 1999 point out that such outbreaks form dense mats that cover the sea floor, smothering underlying seagrass meadows. There have been anecdotal reports of \textit{Lyngbya} in northern Deception Bay since 1992 (Dennison and Abal, 1999). Further blooms have occurred over successive summers and covered an area of approximately 10 km\(^2\) in Deception Bay in late January, 2001 (Ecological Health Monitoring Program, (EHMP), 2001c, 1). The bloom in the summer of 1999/2000 also impacted on the eastern side of the Bay at Amity and Moreton Banks (Grice \textit{et al.}, 2000). Loss of seagrass in these pristine areas is considered detrimental to the feeding dugong and turtle populations.

Human health impacts of \textit{Lyngbya} include severe contact dermatitis, eye irritation and respiratory distress. \textit{Lyngbya} settles on seagrass beds. This deprives the seagrass of oxygen and light, leading to death. \textit{Lyngbya} is also toxic to fish (MBCWQMST, 1998). Following blooms large wracks of decaying \textit{Lyngbya} gather on beaches in the region causing a putrid odour and requiring removal by local authorities (Dennison and Abal, 1999). The outbreaks of \textit{Lyngbya} are correlated with the loss of seagrass in Deception Bay and it is suggested that there has been an associated loss of fisheries harvest (Dennison and Abal, 1999). It has been hypothesised that iron released from acid soils and groundwater may be a contributing factor to the outbreaks (Dennison and Abal, 1999).

\(^9\) Impact assessment of current proposed port expansion indicate that 230 ha of Moreton Bay seabed habitat will be lost to reclamation (Port of Brisbane Corporation, 2000). The planned extensions of port facilities will also result in the physical alteration of coastal environments, increased sediment from dredging, pollution by oil, chemicals, litter and antifouling paints (MBTF, 1997).
6.6 JURISDICTIONAL, LEGAL AND MANAGEMENT ARRANGEMENTS RELEVANT TO THE WETLANDS OF MORETON BAY

The legal and jurisdictional frameworks for the management of the Bay are presented to understand the institutional capacity for improving water quality in the Bay. Thus, the review provides a framework for developing an accurate and plausible scenario for the contingent scenario of providing improved water quality in the Bay and for the collection of theoretical payments.

The jurisdictional arrangements for the management of Moreton Bay are complex. This results from differing jurisdictional responsibilities between the tiers of government in Australia (federal, state and local). These arrangements are overlaid with the international agreements and conventions that the federal government is signatory to. The greatest powers with respect to the management of the Bay are within the state government and in particular the Environmental Protection Agency. The fragmentation of responsibility within the coastal zone has been acknowledged and been the subject of ten successive inquiries between 1988-1994 (Brown, 1995). The concept of integrated coastal zone management is, however, being developed within the Queensland planning system and through community involvement in integrated catchment management plans. One such integrated approach between federal, state and local government, industry and the community is the South East Queensland Regional Water Quality Management Strategy (Moreton Bay Waterways and Catchments Partnership, 2001). An outline of management responsibilities and relevant legislation for each jurisdiction is briefly outlined and illustrated in Figure 6.3. The South East Queensland Regional Water Quality Management Strategy is then reviewed to outline the current strategy to improve water quality in Moreton Bay.
Figure 6.3 Jurisdictional arrangements and major Queensland legislation relevant to the management of the coastal zone and wetlands of Moreton Bay.

Source: Adapted from Environmental Protection Agency, Queensland, (1999c, 5.9).

Notes:
Not to scale
*Normally local council areas stop at high water mark but Brisbane City Council and Redland Shire extend to low water mark
HAT - Highest Astronomical Tide
MHWS – Mean High Water Springs Tide
HLWS – Mean Low Water Springs Tide
LAT – Lowest Astronomical Tide
NM – Nautical Miles

*International Arrangements*
In recognition of the importance of the wetlands of Moreton Bay, an area of 113 314 ha was inscribed on the List of Wetlands of International Importance (Ramsar Site No 41) under the Ramsar Convention on 22 October, 1993 (Ross, 1996). The areas of Moreton Bay covered under the Ramsar Convention are outlined in Map 6.8. The broad aims of the Convention are to halt the worldwide loss of wetlands and to conserve, through wise use and management those that remain (ANCA, 1996). The listing of Moreton Bay thus places responsibilities on management agencies to ensure the aims of the Ramsar Convention are met. Other international agreements
that the government is signatory to include the CAMBA and JAMBA agreements. These agreements aim to protect the migratory species of birds in Moreton Bay and further illustrate the international importance placed on the wetlands of Moreton Bay. The Australian government is also a signatory to the Convention on the Conservation of Migratory Species of Wild Animals (Bonn, 1979), the Convention Concerning the Protection of the World Cultural and Natural Heritage (Paris, 1972) and the Convention on Biological Diversity (1992) which all impact on the management of the Bay.

Map 6.8 Areas of Moreton Bay covered under the Ramsar Convention

National level
The federal government’s jurisdiction extends from 3 nautical miles off the coast to
the extent of the exclusive economic zone (200 nautical miles). Thus, there is no
direct jurisdiction over Moreton Bay. However, the federal government has
responsibilities to ensure that the wetlands are managed in accordance with the
international agreements and conventions that it is signatory to. For example, the
National Strategy for the Conservation of Australia’s Biological Diversity is
designed to implement the provision of international agreements. Also, under the
Environment Protection Biodiversity and Conservation Act 1999 the Commonwealth
has an increased role in the management of wetlands particularly where they are
under international agreements. The importance of the wetlands of Moreton Bay has
been recognised at the federal level and they have been listed as a Wetland of
National Importance (No 134) as the area fulfills all the criteria for selection (ANCA,
1996).

State Level
The Queensland Government has primary responsibility for the management of
coastal resources with jurisdiction extending to 3 nautical miles seaward of the coast.
There are over 10 different government portfolios and more than 60 Acts of
Parliament that relate to the Brisbane River and Moreton Bay catchments (BR and
MBWMS, 1998b). These differing Acts of Parliament also provide for overlapping
jurisdiction. For example, Moreton Bay has been declared a Marine Park under the
Marine Parks Act 1982 which includes all state waters to the limit of the highest
astronomical tide mark (HAT). The Coastal Protection and Management Act 1995
(s11) on the other hand defines the coastal zone as “coastal waters and all areas to the
landward side of coastal waters in which there are physical features, ecological or
natural processes or human activities that affect, or potentially affect, the coast or
coastal resource”. In this case the extent of the coastal zone depends on establishing
a link with the coast or coastal resource. The Queensland Government has produced
a Strategy for the Conservation and Management of Queensland Wetlands
(Environmental Protection Agency, 1999b) to provide an integrating framework for
agencies responsible for wetland management.

Although management of the Moreton Bay Marine Park is undertaken by the Queensland Parks and Wildlife Service the management of fisheries in the Bay is undertaken by the Department of Primary Industries. The Department of Primary Industry’s Fisheries Group is responsible under the *Fisheries Act 1994* for the management, use, development and protection of aquaculture, marine plants, fish habitats and coral limestone. The Department of Primary Industries has control over the wetland flora as under the *Fisheries Act 1994* all marine plants (including mangroves and seagrasses) are protected. A permit must be obtained from the Department before these plants can be removed for any purpose. The Department of Primary Industries also has an advisory role under the *Environmental Protection Act 1994* and the *Nature Conservation Act 1992*.

Within Moreton Bay there are eight Fish Habitat Reserves and four Wetland Reserves protecting a total area of 52,927 ha which come under the jurisdiction of the Department of Primary Industries (Quinn, 1993). Fish Habitat Reserves allow for fishing and crabbing to take place, however, the destruction or disturbance of plant life, sedentary animals or banks is prohibited. Development within a Fish Habitat area is strictly limited (MBTF, 1997). There are also two fish sanctuaries at Swan Bay and Coombabah Lake where fishing is prohibited to protect juvenile stocks.

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10 *The licenses and permits under these Acts are being transferred to the Coastal Protection and Management Act 1995 and their processes are to be made consistent with Integrated Development Assessment System under the Integrated Planning Act 1997.*
The Department of Natural Resources and Mines is also involved in the management of Moreton Bay through its role in land planning, protection and management. For example, the department is responsible for the management of acid sulfate soils and integrated catchment management. Relevant legislation includes the *Land Act 1994, Water Act 2000, Vegetation Management Act 1999* and *Integrated Planning Act 1997*.

*Local Government*

Local government through its role in day to day land-use decisions and responsibilities for wastewater management (storm water, sewage, industrial, agricultural and domestic runoff) also plays a part in protecting and managing the wetlands of Moreton Bay. Councils with responsibilities in the Moreton Bay region include the Brisbane City Council, Caloundra City Council, Logan City Council, Gold Coast City Council, Redcliffe City Council, Caboolture Shire Council, Pine Rivers Shire Council and Redland Shire Council. The jurisdiction of these councils is illustrated in Map 6.9. These Councils have legislative responsibility for the waters and land adjoining Moreton Bay and are responsible for ensuring safe environmental practices in relation to wastewater management and land-use planning for Moreton Bay. The legislation administered by local government relevant to the coastal zone includes the *Integrated Planning Act 1997* and the *Local Government Act 1993*. Local Councils are also responsible for extensive areas of wetlands that are freehold and are managed with the assistance of community-based advisory committees. Examples include the Tinchi Tamba Wetlands on the Pine River and the Boondall Wetlands Reserve.
The SEQRWQMS is an interdisciplinary study of Moreton Bay and its major tributaries and was initiated to address water quality problems. The aims of the initiative are to preserve, protect and improve the quality and amenity of the Brisbane River and its associated tributaries by facilitating ecologically sustainable development in the catchment. The strategy supports the Waterways Management Plan (1998) (Brisbane River Management Group and Brisbane River and Moreton Bay Wastewater Management Study, 1998) which is an overall plan for the management of recreation, cultural activities, water entitlements, environmental flows, catchment land use, transport, tourism, noise and water quality of the Moreton Bay.
Bay catchment. The strategy has been conducted in stages. Stage 1 (1993-1995) reviewed the available information and produced a model for the strategy development (Lloyd et al., 2000). Stage 2 focussed on Moreton Bay and its major tributaries and Stage 3 (currently being undertaken) extends the geographic focus to include south east Queensland freshwater and catchment areas (Lloyd et al., 2000).

An important outcome of the Strategy has been the development of a comprehensive Ecological Health Monitoring Program (EHMP) for the regions waterways (Lloyd et al., 2000). Through the scientific studies undertaken for Stages 1 and 2 and the EHMP the strategy has been able to provide a better understanding of the ecosystem processes and effects of pollutants in Moreton Bay. Other initiatives include the development of ecological health indicators and the determination of sustainable point source nitrogen loads for different waterways (Lloyd et al., 2000). The strategy has also identified a number of actions, which should be undertaken over the next 20 years to achieve its aims. These include a framework for sewage and stormwater management (Lloyd et al., 2000). These actions provide a means of illustrating realistically and honestly, how water quality could be improved in Moreton Bay and hence protect the wetlands. Thus, the Waterways Management Plan with its contributions from both local and state governments can be used in the contingent valuation survey as the institutional mechanism for improving water quality in Moreton Bay.

To promote the strategy throughout the wider community, the Healthy Waterways campaign has been developed by SEQRWQMS as an inclusive identity to give a clear single focus to all actions being taken now and into the future. Other outcomes include the publication The Crew Member’s Guide to the Health of Our Waterways (MBCWQMST, 1998) which provides a report card for the health of Moreton Bay. The underpinning of the scientific research supporting these documents was published as The Moreton Bay Study: A Scientific Basis for the Healthy Waterways Campaign (Dennison and Abal, 1999).

11 South East Queensland Regional Water Quality Strategy was previously called the Brisbane River and Moreton Bay Wastewater Management Study.
6.7 CONCLUSION

The chapter has documented the physical and economic characteristics of the wetlands of Moreton Bay. It has been argued that all the attributes of ecological value, productivity, provision of dependent habitat and biodiversity are provided by the wetlands of the Bay. For example, available evidence indicates that the wetlands contribute 32.1% of total primary productivity in the Bay and provide dependent habitat for a range of fish, prawn and crab species of commercial importance. The Bay’s diversity is exemplified by the large number of species of fish, molluscs and birds that occur in the area. Less is known about the species, which in turn support these populations. The presence of vulnerable and endangered turtles and dugongs species has also been noted. This information gained from the scientific literature provides legitimate verification of the ecological values of Bay. This information can then be synthesised and presented within a contingent valuation survey.

The economic values provided by the wetlands have also been outlined within a total economic framework. There have been no studies undertaken in the region specifically to value the economic contribution of the wetlands. However, from existing data the evidence suggests that all components of economic value exist in the Bay. Use values can be inferred for the available figures for fishing (commercial and recreational) and the cost of upgrading wastewater treatment plants is suggested to be an indication of the indirect use value previously provided free by the wetlands. Similarly, there have not been any previous studies undertaken to estimate non-use values. It has been argued that non-use values for the existence of endangered species may however be held. Bequest values are clearly expressed by the indigenous inhabitants of the region. Based on this evidence two further sets of information on the economic values of the Bay (use and non-use) can be presented within a survey. Thus, the evidence presented within this chapter provides the necessary support for the development of three sets of information to be presented in a contingent valuation survey.

Threats to the wetlands of the Bay through conversion or degradation from declining water quality are expected to continue in line with population growth. These threats provide the basis of a scenario to be provided within a survey of the case study area.
The jurisdictional and management arrangements for the wetlands have been outlined to indicate the agency which would be considered appropriate for the management of water quality in the Bay. This information provides a credible institutional setting for the delivery of improved water quality and hence protection of the wetlands in the Bay.

From the information presented in this chapter, it is possible to provide accurate information to respondents of a survey about the range of ecological and economic benefits provided by the wetlands of Moreton Bay. The information can also be used to devise a contingent scenario for improved water quality in the Bay. The methods used to conduct the survey are provided in Chapter 7.
CHAPTER 7
CONTINGENT VALUATION SURVEY METHOD

The purpose of this chapter is to outline the methods and rationale for the design of a contingent valuation survey to be tested in Moreton Bay. A general outline of the contingent valuation technique, problems and the possible means of resolving them within a survey have been examined in Chapter 5. The design of the case study survey follows the guidelines established in that chapter. The ecological and economic values of the case study area have been presented in Chapter 6. The values identified are used as the basis for information to be provided within the questionnaire. This chapter describes the aims and hypotheses to be tested. This is followed by the description of the good, in four alternative formats. A scenario for provision of the good, proposed payment vehicle and bid elicitation methods are then given with follow up questions. Further questions relating to attitudes towards the environment, the existing level of knowledge of the good, and socio-economic details of respondents are then outlined. Finally, the sampling logistics and the mechanism for questionnaire delivery are provided.

7.1 AIMS AND HYPOTHESES

The purpose of conducting the contingent valuation study is to determine if ecological values could be captured using an economic valuation method. In particular, the survey was designed to test the effect on willingness to pay of providing ecological information to respondents. The survey also sought to establish if willingness to pay given ecological information differs from willingness to pay when given information about economic values such as use and non-use values as discussed in Chapter 4. To investigate these effects the survey has been designed with four versions of the questionnaire. Case A provided no extra information so that it could be used as a control. Case B provided information on the ecological values of the wetlands. Case C provided information on the use value of the wetlands (both direct and indirect). Case D provided information on non-use values (bequest and existence values) of the vulnerable and endangered species that are dependent on the wetlands in Moreton Bay for at least part of their life cycle. The
information provided was distilled from the review of the ecological and economic values of Moreton Bay as presented in Chapter 6. The full text of the survey is provided in Appendix D.

7.1.2 Hypotheses and Research Questions

The first question to be tested is: does the provision of different information in the four different cases produce a different mean willingness to pay bid? If the provision of different types of information did not have any influence on willingness to pay then it would be expected that the mean willingness to pay for each case would be equal (at the 5% significance level). If the alternative was true and the type of information did influence mean willingness to pay then it would be expected that for at least one case mean willingness to pay would not be equal to at least one other case. Thus to test for an ‘information effect’ the hypothesis to be tested is if the provision of a similar amount but different type of information will produce a differing mean willingness to pay bid. More formally,

1. \( H_0: \) Mean willingness to pay is equal for each case
   
   \[
   (\mu \text{ Case A} = \mu \text{ Case B} = \mu \text{ Case C} = \mu \text{ Case D})
   \]

If it is found that there is a difference in mean willingness to pay then it is possible to address the question of whether the provision of ecological information produces a different mean willingness to pay bid. In Chapter 3 it was argued that ecological value is a prerequisite for the provision of instrumental human benefits including use and non-use economic values. That is, ecological value is the primary value on which the secondary value of economic goods and services depends. Therefore, if a contingent valuation survey is capable of capturing ecological value it would be expected that mean willingness to pay for ecological value should exceed that of either use or non-use value. That is, if respondents to a survey also comprehend that the existence of ecological value occurs prior to the production of ecosystem goods and services then their willingness to pay for ecological value should exceed their willingness to pay for the individual components (use and non-use) of economic value. Therefore, the research question addressed here is does including information about the ecological values of wetlands in a contingent valuation survey produce a different mean willingness to pay bid from the cases where information is included.
on other economic values. The proposition is that if the mean willingness to pay for ecological value exceeds mean willingness to pay for other components of economic value then it suggests that the contingent valuation method can capture ecological value.

Thus, the hypothesis to be tested is:

2. \( H_0: \) Mean willingness to pay when provided with ecological information is equal to mean willingness to pay when provided with either no information, use information or non-use information (\( \mu \) Case B = \( \mu \) Case A, \( \mu \) Case B = \( \mu \) Case C, \( \mu \) Case B = \( \mu \) Case D).

The survey also included questions to assess if it met the requirement that respondents were familiar with the good they were being asked to value. These requirements include that respondents had sufficient awareness and knowledge of the Bay to form a value for the wetlands\(^1\). The attitudes of respondents with respect to wetland preservation and the economic values they provide were also sought. These values can then be compared with those resulting from the willingness to pay questions as an internal consistency check. Thus, further hypotheses to be tested were if:

3. Respondents possess a sufficient knowledge from use of the Bay and awareness campaigns to form a value for the wetlands.

4. Respondents’ attitudes with respect to wetland protection depend on the economic value being assessed. Also, did the particular information provided in each case produce any difference in attitudes expressed?

5. The attitudinal rankings of the importance of protecting wetland in terms of the ecological or economic values they provide was the same as the rankings of ecological and economic value derived from willingness to pay bids.

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\(^1\) Hanley (1989) notes that ideal conditions for conducting a CVM include that respondents understand and are familiar with the commodity to be valued.
7.2 INFORMATION PROVIDED TO RESPONDENTS FOR THE FOUR CASE STUDIES

A vital component of any contingent valuation study is a detailed description of the resource to be valued (Mitchell and Carson, 1989). The ‘good’ on offer in this survey was an increase in water quality, which would result in the preservation of wetlands in Moreton Bay. As noted in Chapter 6 there is a number of threats to the continued viability of the wetlands related to the increasing population in the area. Some of these include direct reclamation of wetlands for canal and port development. However, it is also expected that if no action is taken water quality will continue to decline due to increased loads of nutrients and sediments. The increased nutrient loading and decreased light availability due to turbidity have been clearly linked to seagrass decline. Thus, it is considered credible to directly link water quality to wetland preservation. As nearly all stormwater, wastewater and sewage arising from the catchment ultimately influences the water quality in the Bay there is no one agency that can be directly identified for causing the problem. Thus, the notion that improving water quality will aid wetland protection implies that it is a shared responsibility.

The challenge in defining this good is to translate the various values of the Bay into a form that is understandable by survey respondents. It is not possible to include full information about all the values of the Bay. For example, Chapter 2 has detailed information on the functions of wetlands. This information needs to be summarised in a form readily understood by the public. Thus, information was synthesised from the academic literature as presented in Chapter 2 and 6. The aim was to present the relevant information in way that was accurate but could be understood easily without causing information overload. As stated four cases were produced with the difference between cases being the type of information that was presented about the wetlands in terms of their various ecological and economic values.
7.2.1 Case A: No extra information

For Case A, only a baseline description of the good was provided. This sample was designed to be a control for the effects of information in total. Therefore, less information was provided in this case\(^2\). The review of the literature which has investigated information effects in contingent valuation surveys as presented in Chapter 5 suggests that the provision of extra information in the other cases may produce a higher mean willingness to pay that this baseline case. The other three samples were designed to test the effects of providing differing types of information, thus an attempt was made to keep the information the same in terms of length and the number of details provided. A summary of the information on which the text for each version was based is presented below.

7.2.2 Case B: The Ecological Values of the Case Study Area

The wetlands of Moreton Bay exhibit all the attributes of ecological value as defined in Chapter 3. It should be restated that throughout this thesis the definition of wetlands is taken from that adopted by the Ramsar Convention. This definition includes the diversity of wetland types found in Moreton Bay, including the coastal wetland ecosystems of seagrass, mangrove, saltmarsh and saltpans and corals. Taken together these systems provide all the attributes of ecological value, that is, productivity, dependency and biological diversity and organisation. A map was provided in the survey that highlighted the wetland areas of Moreton Bay\(^3\).

From the review of the scientific literature, the description of the relevant aspects of the ecological values of Moreton Bay presented to respondents included information about the productivity of the wetlands and its contribution to ecological processes. As noted in Chapter 6, it is estimated that the seagrasses and mangroves provide for nearly one-third of the primary production occurring in Moreton Bay (Abal \textit{et al.},

\(^2\) The only information included was a statement at the start of the survey stating that “the wetlands of Moreton Bay include areas of mangroves and seagrasses found along the foreshore and islands, and in shallow waters. These wetlands have been identified as being internationally important, especially for wader birds”. This information was also provided for the other cases.

\(^3\) A map was included of Moreton Bay for all cases which highlighted 41491 hectares of wetlands (18233 ha of seagrass and 23258 ha of mangroves and saltmarsh). A colour version of the map is provided in Chapter 6 (Map 6.3) and the black and white version is presented in Appendix E.
The ecological value of the dependency of many species on the wetlands for feeding, resting and reproduction was also given. This followed from the review conducted in Chapter 6 which highlighted the importance of wetlands for species of commercial importance (fish, crabs and prawns), endangered species (turtles and dugongs) and international important migratory wader birds as well as microfauna and benthic fauna. Respondents were also informed that as a result of the high productivity and dependency provided by wetlands there was a high level of biodiversity in the wetlands. This diversity includes the 3000 species of free-living macro invertebrates and 713 fish species living in the Bay (Davie and Hooper, 1998, 333). In addition, as noted the diversity of Bay also includes 273 bird species and 40 species of molluscs (Davie et al., 1998, 27, 319). It was then explained that the interaction of the species present provided a range of services such as food and waste assimilation. An outline of the biogeochemical and ecological functions afforded by wetlands, as given in Chapter 2, provided the basis for the comments. The connection between these services and the health and integrity of the Bay was highlighted. The information was intended to convey the notion that the aspects of ecological value as defined in Chapter 3 is dependent on the interaction of the attributes of productivity, dependency and biodiversity. The text as presented in the survey is given in Figure 7.1.

Figure 7.1  Information presented in Case B- Ecological Values

The large numbers of plants in the wetlands capture energy from the sun and convert this energy to a form that can be used by animals. This productivity provides the basis for all further biological activity. The wetlands of Moreton Bay also provide areas for animals to feed, rest and reproduce. Many animals, including fish, return to particular areas for breeding and so local populations of these animals are dependent on the existence of the wetlands. Migratory wader birds also use the wetlands to feed and rest before their return journey to the Northern Hemisphere. Due to the fact that the wetlands are productive and provide places for animals to feed, breed and rest, they contain a wide range of different plants and animals. The interaction between the variety of plants and animals produces food and helps assimilate wastes. This in turn helps to maintain the health and integrity of the complex wetland system.
7.2.3 Case C: Use Values of the Case Study Area

Case C presented respondents with the information related to the economic use values, both direct and indirect, of the wetlands of Moreton Bay as outlined in Chapter 6. From the large number of economic use benefits, respondents were presented with information about the recreational fishing and boating opportunities that are provided by the wetlands. As noted in Chapter 6, Quinn (1993, 6) estimates that 300 000 fishers engage in recreational fishing in Moreton Bay and it is also stated by Quinn (1993, 39) that 75% of commercial catch by weight is derived from species dependent on the wetlands in some way. The information presented in Chapter 6 suggests that the value of recreational fishing is worth at least $240 m. Thus this was considered to be an important activity in the region (BR and MBWMS, 1998a, 17). The wetlands also provide shelter for a range of water-based activities on the Bay. These recreational benefits were included. The aesthetic character of the wetlands and the opportunities to enjoy wildlife viewing were also highlighted. As noted in Chapter 2 and 6, Moreton Bay has been declared a Ramsar site due in part to the 50 000 migratory wader which use the Bay during the non-breeding season. It was also noted that it is estimated 2000 people visit the Bay specifically to watch these birds (Australian Nature Conservation Agency, 1996). The indirect use value of the wetlands was described as purification, which in turn maintained water quality so that it is safe to swim and consume food caught in the Bay. The purpose of including this information was to highlight that if the waste assimilation capacity of the wetlands was lost then water quality would decline. This would then have an impact on other recreational opportunities available on the Bay. Thus, it highlights the economic value of maintaining the wetlands. As noted in Chapter 2, Costanza et al., (1997, 11) estimate that the services of waste assimilation and protection may account for 80% of the economic value of wetlands. Finally, it was noted that in some parts of the Bay the wetlands were being degraded and destroyed so that these services would be lost. Map 6.4 has highlighted the areas where seagrass loss has occurred and Chapter 6 has outlined a number of threats which continue to endanger the wetlands. The text as presented in the survey is given in Figure 7.2.

4 The Moreton Bay Task Force, (1997, 42-47) notes that many of the fish and crab species in the Bay are targeted by both commercial and recreational fishers.
The areas of Moreton Bay sheltered by wetlands provide safe places for recreational fishing and boating. In fact, over three-quarters of fish caught by the 300,000 fishers who use Moreton Bay each year depend on the wetlands in some way. The sheltered waters also provide for a range of water related recreational activities such as sailing, motor boating, water skiing and diving. Others enjoy visiting the foreshores to view the wildlife such as the 50,000 migratory wader birds that visit the Bay each year. The visual character of the wetlands also adds to the pleasant scenery provided by the Bay. Also, as wetlands purify the water, it means that people can swim safely and eat the seafood they catch. However, in some parts of the Bay wetlands have been degraded and destroyed and they can no longer provide this service.

7.2.4 Case D - Non-use Values of the Endangered Species of the Case Study Area.

The information given in Case D was presented to derive an estimate of the non-use value of endangered species. Estimations of non-use or passive value in contingent valuation studies can be made in relation to wilderness areas or for species, that one might never expect to visit or see. As noted in Chapter 6 both dugongs (*Dugong dugon*) and green turtles (*Chelonia mydas*) are considered vulnerable under the *Nature Conservation (Wildlife Regulation)* 1994. Chapter 6 also presented information on the habitat requirements and reproduction cycles of dugongs. Of particular note for both species is that they are slow to mature and have low fecundity. From the available information about the habitat requirements and reproduction cycles of dugongs and turtles respondents were presented with factual information about the two species. This included their threatened status and their dependency on the wetlands of Moreton Bay. Chapter 6 has documented that both species are dependent on seagrass for food. The reproductive patterns of both species were outlined to demonstrate that the loss of adults could have an impact on the size of the population. The threats resulting from wetland loss as has been outlined in Chapter 6 were highlighted. It was also noted that theses species might not provide direct use benefits but that their existence was reliant on the wetland habitats of the Bay. This was to included to encourage respondents to consider the
existence value of the species rather than any benefit the species may provide for them directly. The full text presented to respondents is given in Figure 7.3.

Figure 7.3  Information provided in Case D – Non-use values of endangered species

| The wetlands provide a place to live for a number of threatened and endangered species such as turtles and dugongs. Sea turtles are air breathing marine reptiles. All the turtles in Moreton Bay are directly or indirectly dependent on the wetlands. For instance, green turtles feed directly on the seagrasses, which form part of the wetlands. Turtles may take between 20 – 50 years to reach sexual maturity. Dugongs (also known as sea cows) are harmless marine mammals. They only feed on plants such as seagrasses. Dugongs have a low reproduction rate, only producing a single calf every 3-7 years. As both turtle and dugong populations only increase very slowly, loss of adults can result in chronic declines in their population. Both these animals are threatened by habitat destruction, such as the loss of wetlands in Moreton Bay. These protected animals may not provide direct benefits to humans but their continued existence relies on the habitat provided by the wetlands in the Bay. |

7.3 SCENARIO FOR PROVISION OF THE GOOD

Following the description of the ‘good’ a contingent valuation survey needs to present to respondents the initial condition of the good and the proposed alternative condition following the implementation of the scenario (Mitchell and Carson, 1989). As noted the importance of the wetlands of Moreton Bay has been acknowledged at the international, national and state level. These wetlands maintain water quality through their ability to absorb the sediments and nutrients released into the Bay (Dennison and Abal, 1999, 91). Conversely, the poor quality of water directed to the Bay from the catchment threatens the continued viability of the wetlands. For example, as noted in Chapter 6, changes in seagrass depth range which are used to infer changes in water quality demonstrate that between 1992 and 2000 there were declines in seagrass beds around the Bay (Grice et al., 2000). These losses occurred particularly in the southern and western areas where for example, there has been a total loss of seagrass in the Bramble Bay and Southern Deception Bay areas (Dennison and Abal, 1999, 64). It is estimated that since European settlement
approximately 20% of the total seagrass area in Moreton Bay has been lost (Abal et al., 1998, 269). These changes are illustrated in Map 6.4. Mangroves still exist in these areas but it is suggested by the Moreton Bay Catchment Water Quality Management Strategy Team (MBCWQMST), (1998, 29) that their capacity to continue to filter and absorb nutrients and sediments may be exceeded in some parts of the Bay. Dennison and Abal (1999) note water quality has declined to the extent that much of the western border of the Bay is now considered unsafe for swimming. As noted in Chapter 6, the wastes discharged into Moreton Bay are from both point and non-point sources and include urban and rural stormwater and wastewater. Thus, the preservation of the wetlands is threatened by the poor quality of water discharged into Bay. Therefore, the initial condition of the ‘good’ is the wetlands of Moreton Bay that are being threatened by poor water quality and the proposed alternative, preservation of the wetlands of Moreton Bay was linked to improving water quality.

The alternative condition is that wetlands are preserved through improvements in water quality. As noted in Chapter 6, The South East Queensland Regional Water Quality Management Strategy has been designed to achieve this alternative condition. Within the strategy water quality objectives are designed to meet stated environmental values and objectives. Both community views and scientific information determined the environmental values and water quality objectives. Thus, it was considered that improved water quality would be important to respondents as it provided them with the community stated environmental values of ecological health, consumption of seafood and primary and secondary contact and visual recreation (BR&MBWS & BRMG, 1998b). These environmental values exemplify the economic values of the wetlands of Moreton Bay identified in Chapter 6. The strategy and Healthy Waterways publicity program have highlighted the losses of wetlands due to low water quality and the various ways that water quality can be improved in the Bay to meet water quality objectives. Funding has been

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5 Water quality is measured for a range of indicators including chlorophyll $a$, total nitrogen, total phosphorous, total carbon, suspended solids, denitrification efficiency, toxicants, turbidity, faecal coliforms etc depending of the particular objective (BR&MBWS & BRMG, 1998b).

6 The Moreton Bay Catchment Water Quality Management Strategy (1998) aims to identify management actions that will achieve its objectives for water quality in Moreton Bay (BR&MBWS & BRMG, 1998). These objectives include restoring and protecting water quality and ecological health.
provided for the highest priorities of the management strategy, such as the upgrade of sewage treatment plants but other programs, which would also result in water quality improvements, such as improving stormwater are yet to be funded. Willingness to pay for preservation of wetlands through water quality improvements can be directly linked to these strategies. Thus, the scenario for the provision of the good is realistic and honest. The wording for the provision of the good was the same for all four cases and is given in Figure 7.4.

Figure 7.4 Information provided on the provision of the good.

| The areas of marine vegetation in Moreton Bay are under threat due to the effects of low quality water coming from the rivers into the Bay. This poor quality water affects the mangroves and seagrasses which means that they can no longer function properly and in some cases results in their complete loss. To stop the loss of the wetlands the Local councils and State government have prepared a Waterways Management Plan. Some actions identified in the plan are being undertaken by Local and State governments. Other activities have not yet been undertaken due to a lack of funding. |

7.4 PAYMENT VEHICLE

There are a number of institutional methods for collecting willingness to pay bids. In the context of the wetlands of Moreton Bay the most feasible methods for collecting (actual) payments would be through the use of increased council rates, taxes or a contribution to a voluntary conservation fund which could carry out the actions identified in the *South East Queensland Regional Water Quality Management Strategy* (1998) (BR&MBWS & BRMG, 1998).

Chapter 5 outlined a review of previous contingent valuation studies in Australia to determine the most appropriate means of collecting payment. However, previous studies provided conflicting views on the most appropriate method for collecting payments and thus offer little guidance. However, it was noted in the Bennett *et al.*, (1997) study that there were more protest bids using tax as the payment vehicle.

This will in turn protect the wetlands which are being degraded as a result of poor water quality
Given that an additional tax was introduced in Australia, just prior to the survey, it was possible that asking for an extra tax payment would produce an undesirable response\(^7\). The possibility of using an environmental levy on local government rates was also investigated. However, as noted in Chapter 6 (Map 6.9) there eight local government areas adjacent to Moreton Bay. Presently, five of these councils charge an extra levy for environmental purposes but the amounts differ for each council. Thus, this would make the use of a local government levy problematic. Some residents presently pay nothing and others pay between $15 and $45 per property and asking the latter to pay a further levy may be resisted (Lusis, 2001). Also, the benefits of improving water quality in Moreton Bay may extend to others who do not live in the adjacent rate paying areas or who may not be ratepayers.

The alternative was to set up the contingency that the money could be collected by a non-government organisation such as a Conservation Trust Fund. The sole purpose of the fund would be to carry out works identified in the Management Strategy. The advantage of using a non-government agency was to avoid problems related to the public perception of current government taxation and spending. This, however, may have raised the possibility that the fund would be perceived as another layer of bureaucracy, which would require further funds to support the administration\(^8\). This possibility was investigated in a question following the willingness to pay question.

\(^7\) A 10% goods and services tax was introduced for all items with the exception of some foodstuffs in June, 2000.

\(^8\) In the pilot study only 9.7% of respondents who refused to offer a positive willingness to pay amount gave the reason as disagreeing with paying money into a fund. It was thus considered a suitable payment vehicle.
7.5 BID ELICITATION

From the variety of bidding methods that have been used for contingent valuation studies the iterative bidding technique and dichotomous choice were considered suitable means of eliciting a willingness to pay bid for ecological attributes. It was argued in Chapter 5 that if the initial bids for the iterative bidding technique were derived from an open-ended question in a pilot study this would be the superior method given consideration of costs and computational difficulties. Given that the purpose of the study was to compare subsamples, the iterative bidding technique was selected to elicit willingness to pay for improved water quality after determining initial bids from an open-ended pilot study. The full details of the pilot study are provided in Chapter 8.

In Chapter 5 it was noted that individuals may suffer from fatigue if a lengthy bidding processes is engaged (Anderson and Bishop, 1986). In order to avoid fatigue, the number of iterations was limited to two. However, the iterative technique still allows respondents to search their preferences more thoroughly than would be possible with a single dichotomous choice question. This is considered desirable as it increases the probability that the bid offered more truly reflects what the good is worth to the respondent before offering a hypothetical payment (Garrod and Willis, 1999).

The amounts offered to the respondents were determined by conducting a pilot survey of 398 respondents from the same electoral districts as the main survey. As described in Chapter 8, an open-ended willingness to pay question was asked to determine the first quartile, median and third quartile values. From a total of 56 positive bids the mean value of willingness to pay was $35.68 with a standard deviation of $36.12. The resulting first quartile, median and third quartile values were $10, $20 and $50. These values were thus selected to be presented to respondents in the iterative bidding game. Figure 7.5 illustrates the iterative process.
Figure 7.5 Sequence of Iterative Bidding Question

<table>
<thead>
<tr>
<th>Question</th>
<th>If Yes</th>
<th>If No</th>
</tr>
</thead>
<tbody>
<tr>
<td>Would you be willing to make a once off contribution of $20?</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Would you be willing to pay $50</td>
<td></td>
<td></td>
</tr>
<tr>
<td>What is the maximum amount you would be prepared to pay?</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Would you be willing to pay $10?</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

The wording of the bid elicitation question is presented in Figure 7.6. Following recommendations from Arrow et al., (1993) respondents were also reminded to consider their household budget constraints and other areas of environmental spending. Respondents need to be reminded of their income constraints to avoid over-inflated answers which could be interpreted as ‘play money’ (Rolfe et al., 1998).

Figure 7.6 Bid elicitation question.

A non-government Conservation Trust fund could be set up that specifically undertakes actions as identified by the Waterways Management Plan to improve water quality in Moreton Bay. If such a fund was set up to specifically undertake actions that would improve water quality and hence protect the wetlands of Moreton Bay, and keeping in mind your household budget and other areas of environmental spending, would you be willing to make a once off contribution of $20?

It was also noted in Chapter 5 that the use of a willingness to accept compensation bid is likely to produce a higher estimate of value than a willingness to pay bid. However, a willingness to pay question was used in the case study. As the area has already been degraded due to past community actions there is no agency, which
could feasibly be presented as being responsible to pay compensation. It is this same community which collectively contributes to waste discharges to the Bay that continue to cause the problems outlined in Chapter 6. It would thus seem unrealistic to ask the community what they would be willing to accept in compensation for the loss of wetlands, which they are degrading. In addition, the wetlands of Moreton Bay are a commonly held resource and thus there are no private property rights to them. In this case, the benefits associated with the preservation of the wetlands will accrue to the community at large, if they are willing to pay for them.

7.6 REFUSALS AND ZERO BIDS

There are a number of reasons why respondents may be unwilling to pay the amount asked for the good on offer. Contingent valuation researchers have thus developed the technique of asking respondents who gave zero bids their reason for doing so (Hanley, 1989; Mitchell and Carson, 1989). This allows those who truly hold no value for the good being valued to be distinguished from those who objecting to some other aspect of the survey. Mitchell and Carson (1989) note that protest bids can constitute a considerable fraction of zero bids. Protest bids are defined as statements of zero willingness to pay from individuals who presumably value the good positively (Lindsey, 1994).

Therefore, following bid elicitation, questions were included to determine if respondents who were unwilling to offer a positive amount truly felt the wetlands held no value for them or if they were protesting about some other aspect of the contingent scenario (Hanley, 1989). For example, respondents who couldn’t afford to pay or felt the wetlands were worth nothing could be classified as true zero bids (Jorgensen and Syme, 2000). Other respondents may not offer a positive willingness to pay bid as they are protesting about a lack of information or did not want to take part in the bidding game (Hanley, 1989). Spash and Hanley (1995) argue that others may believe it is immoral to place a monetary value on the environment. Protest bids can also be a result of objecting to the payment vehicle or disapproving about paying for something which the respondents perceive is currently provided at no cost as is often the case for public goods (Hanley, 1989).
From the answers given to these ‘follow-up’ questions, zero bids can be re-coded so that true zero bids are included in the means and protest bids discarded (Mitchell and Carson, 1989). This reduces the potential for strategic bias and allows the sample size to be maintained as not all zero bids are discarded. This in turn improves the representativeness of the results (Garrod and Willis, 1999). However, as Garrod and Willis (1999) point out there is no theoretical justification for removing these observations. Mitchell and Carson (1989) support this view arguing that the method for determining which zero bids and outliers to remove is currently ad hoc which raises the possibility that the researcher may engage in selective deletion to achieve the desired result.

It is possible that respondents who answer that they are unwilling to place a dollar amount on protecting wetlands may do so because they believe the wetlands should be protected no matter what the cost. That is they hold a lexicographic position as defined in Chapter 5. Individuals that hold a lexicographic position will be unwilling to trade off environmental goods for money contrary to the assumed utilitarian position of neo-classical economics (Stevens et al., 1991; Spash and Hanley, 1995). It will be assumed that if an individual does hold such a position then their response to questions with regard to their attitudes towards the protection of wetlands and the rights of animals and plants to exist would also reflect this.

Those respondents who were unwilling to offer a positive amount were presented with six statements to determine if they held zero value for the wetlands or were protesting about some other aspect of the survey. These statements were developed from examples given by Garrod and Willis (1999) and Jakobsson and Dragun (1996). The wording for the question used to determine protest bids is presented in Figure 7.7. It was considered that those who truly held a zero valuation would select the statements stating ‘the wetlands are worth nothing to me’ or ‘I couldn’t afford to pay’. The other statements relate to protests about information, the morality of valuing wetlands, the payment vehicle and paying for a public good respectively.

---

Figure 7.7 Question used to determine protest bids.

<table>
<thead>
<tr>
<th>Why were you unwilling to offer to pay any amount?</th>
</tr>
</thead>
<tbody>
<tr>
<td>Please place a cross in the box next to any of the reasons listed that apply to you (you may cross more than one).</td>
</tr>
</tbody>
</table>

- I did not have enough information to place a $ value on protecting the wetlands
- I did not want to place a dollar value on protecting wetlands
- I disagree with paying money into a fund
- The wetlands are worth nothing to me therefore I don’t want to pay
- The government should transfer spending from other areas and pay to protect the wetlands
- I couldn’t afford to pay anything
- Other (please specify) __________________________

7.7 ATTITUDES

The initial section of the questionnaire determined the attitudes of respondents towards the environment and the values of the wetlands. Following from the social-psychology literature it would be expected that attitudes are important predictors of behaviour (Kotchen and Reiling, 2000). It could be expected that those respondents who believe that it is important to protect the wetlands would be willing to offer a positive willingness to pay bid. Conversely, if the respondent holds a lexicographic position then their rights-based beliefs may prevail over utilitarian beliefs about the environment (Stevens et al., 1991; Spash and Hanley, 1995; Kotchen and Reiling, 2000).

A series of statements about the values of the wetlands were developed to encapsulate some of the components of total economic value described in Chapter 2. The statement relating to ecological value was rephrased as health. Chapter 3 has outlined the connection between ecological value and health. The term ‘health’ was a simpler concept to convey to the respondents. The statements were also related to the direct and indirect use values of the wetlands, bequest values and the existence values. Respondents were asked to indicate how important each value was to them.
They were then asked to allocate a fixed number of points (100) between the statements to indicate the relative importance of each to them.

It was anticipated that the responses to these questions could be used to determine if respondent’s attitudes to the wetlands were the same as their willingness to pay for the categories of values presented in the four versions of the survey. This would allow some assessment of the internal consistency of responses. It would also allow some assessment of attitudes in terms of the importance and relevance of the wetlands to the respondents.

The final question asked respondents if they were members of an environmental group. Previous contingent valuation studies have used membership of environmental organisations as an indicator of environmental attitudes. Kotchen and Reiling (2000) point out that within environmental groups individual members may have differing levels of concern and beliefs. The relationship between membership of an environmental group and willingness to pay will be explored to determine its effect and if any lexicographic preferences exist.

7.8 EXISTING USE AND KNOWLEDGE

Respondents to the contingent valuation survey may already possess information about the various values of the wetlands of Moreton Bay. It is assumed that they would use this information stock and the provision of any new information when forming their willingness to pay bids (Turner and Pearce, 1993). The issue of familiarity with the good is considered important because if the respondent is unfamiliar with the good it may lead to biases when formulating a value for it (Anderson and Bishop, 1986).

The population from which the sample was drawn resided in the area adjacent to Moreton Bay with the aim that this group would have some familiarity with wetlands. It was also assumed a priori that those who have visited the area, or used the area for bird watching or fishing would have some knowledge of the ecological and economic values of wetlands. There have also been a number of publicity campaigns recently undertaken in the area relating to the zoning of the Moreton Bay
Marine Park, the ‘Healthy Waterways’ program and as part of a commercial television station promotion of the ‘Great South East’ [Queensland]. The information derived from this question related to use and awareness of the Bay can then be used in the analysis to determine if existing knowledge alters willingness to pay bids.

Answers to questions asked about the use and awareness of the Bay may also give an indication of the recreational direct use value that the wetlands provide. An understanding of the activities engaged in by the respondents will also allow some analysis to be undertaken to determine if those who use the Bay have a differing willingness to pay from those who don’t.

7.9 SOCIO-ECONOMIC CHARACTERISTICS

To ensure that the selected sample was representative of the larger population a number of questions were asked in relation to the socio-economic characteristics of the respondents. These characteristics can then be compared with the characteristics of the population to gauge the representativeness of the sample. Aggregation of values derived from the study is only possible if the sample and the population possess similar socio-economic characteristics. Information on the socio-economic characteristics of residents of the chosen Commonwealth electoral districts was available from the 1996 national population census (Australian Bureau of Statistics). Questions were asked about sex, age, education and income to correspond with the categories used in the census data. This information can also be used to assess the effect of these characteristics on willingness to pay (Garrod and Willis, 1999).
7.10 SAMPLING LOGISTICS

This section outlines the rationale and methods used for the selection of the sample from the population, the sampling frame and the sample size.

*Population*

One of the criticisms of the contingent valuation method is that respondents who are unfamiliar with the good to be valued will have difficulty forming a monetary valuation for it. To overcome this problem the sample was selected from a geographical area that was adjacent to the study area. It was expected *a priori* that those who lived near the study area would possess some previous knowledge about it and this would improve their ability to form a value for it. It is also argued that this group would benefit from paying to protect the wetlands. Mitchell and Carson (1989, p. 264) consider this to be the optimal population choice. The population from which the sample was drawn was defined by the Commonwealth Electoral Districts that are adjacent to Moreton Bay. This includes the area from Caloundra to Southport and is illustrated in Map 7.1. The electoral rolls only include Australian citizens over the age of 18. The sampling unit was the individual respondent selected from these rolls.

*Sampling Frame*

The sampling frame was the Commonwealth Electoral Rolls for the electoral districts adjacent to Moreton Bay. This included the electoral districts of Fisher, Longman, Petrie, Bowman, Fadden, Griffith and Lilley as illustrated in Maps 7.1 and 7.2. Simple random sampling was used to draw a sample of 1600 units.


Map 7.2 Commonwealth Electoral Districts – survey included districts of Lilley and Griffith

Sample Size

The sample size for the case study needed to be large enough to provide sufficient statistical power to reject the null hypothesis (Mitchell and Carson, 1989, 357). Mitchell and Carson (1989, 357-365) provide tables to estimate the total number of observations needed for different levels of \( \alpha \) and \( \beta \) to avoid Type 1 and Type 2 errors, the coefficient of variation (V) and for the percent difference between the mean of the first experiment and that of the second experiment expressed as a percentage of the mean of the first experiment\(^{10} \) (\( \Delta \)). It is noted that the sample size can be considerably smaller if a one tailed test is used.

A pilot study of 398 respondents was undertaken prior to the main study. The full details and results are presented in Chapter 8. The coefficient of variation (V) can be calculated from the standard deviation divided by the mean. In the pilot study it was found that willingness to pay had a standard deviation of 36.1207 and mean of 35.6321, so V=1.01. If a level of \( \alpha=0.05 \) and \( \beta=0.05 \) for a one tail test is chosen with V=1 and a 30% difference between the mean of the first experiment and that of the second experiment accepted as the confidence interval a sample size of 242 useable responses would be needed (Mitchell and Carson, 1989, 365). The useable response rate for the pilot survey was 56.03%. Thus, to ensure 242 useable responses would require sending 432 questionnaires for each survey version. Taking these estimates and funding into consideration, 400 questionnaires were sent for each of the four cases. Thus, the overall sample size of the study was 1600 units. Each element within the sampling frame was given a number between 1 and 591 155. Then 1600 different random numbers were generated between 1 and 591 155. These numbers were then used to select the sample.

\[ \Delta = \frac{X_1 - X_2}{X_1} \]

\(^{10}\) That is, \( \Delta = \frac{X_1 - X_2}{X_1} \)
7.10.1 Questionnaire Delivery

A choice of questionnaire delivery needed to be made from the three major methods of face to face interviews, telephone interviews and mail surveys. Each has its own advantages and disadvantages in terms of control of sampling units, response rates, cost, potential for interview bias, questionnaire length and the types of questions that can be asked (Dillman, 1978; Lehmann et al., 1998; Malhotra, 1999). The three methods are outlined to illustrate why the final choice of a mail survey was made. The method used for the questionnaire design is then discussed.

Face to Face Surveys
Face to face or personal interviews have the ability to control which sampling units are interviewed and have recorded higher response rates than other methods but it is the most expensive method to collect data, particularly if respondents are spread over a large geographic range. Also, the presence of the interviewers may influence responses (interviewer bias), the interviewer may be left to interpret the response and assign it to a particular category, which may increase errors. Also, the interviewer may cheat to complete the required number of questionnaires. However, personal interviews allow a large amount of information to be collected, can provide visual stimuli and allow a complex sequencing of questions. Also, as the interviewer can encourage respondents to answer complex, open-ended and tedious questions. Problems can arise for personal interviews with respect to finding respondents at home and asking sensitive questions in the presence of the interviewer (Dillman, 1978; Lehmann et al., 1998; Malhotra, 1999).

Telephone Surveys
Telephone interviews allow control over the selection of respondents within sampling units, produce high response rates with costs lying between that of personal and mail surveys and a reduced potential for interviewer bias. Telephone surveys have the ability to reach large geographic range but may limited to where phone numbers are available. However, only a limited quantity of data can be collected and physical stimuli cannot be used. Telephone surveys do have the ability to control the sequences of questions, and are successful with open-ended tedious and sensitive
questions thus avoiding item non-response. Telephone interviews also have a high speed of implementation (Dillman, 1978; Lehmann et al., 1998; Malhotra, 1999).

Mail Surveys
Mail surveys have limited control over selection of respondents within sampling units, avoid interviewer bias and have a higher non-response but lower costs than other methods. Mail surveys also have an ability to collect information from respondents over a large geographic range. Mail surveys can collect moderate amounts of data and physical stimuli can be included, but have low success with open-ended and tedious questions, the use of complex branching, and no control over item non-response. However, due to the anonymity of respondents sensitive questions can be asked (Dillman, 1978; Lehmann et al., 1998; Malhotra, 1999).

Mitchell and Carson (1989, 111) suggest that in-person interviews are the technique of choice for contingent valuation surveys. However, they acknowledge that this is also the most expensive technique and that telephone and mail surveys may work equally as well when respondents are familiar with the good or if the scenario is relatively simple. The sample was selected to ensure (and test) that the respondents were familiar with the wetlands of Moreton Bay. However, as the respondents were spread geographically over a strip greater than 200 kms in length it would have been very expensive to conduct in-person interviews. A telephone survey could have overcome this problem but would not have allowed the inclusion of a map to convey the extent of the wetlands. It was also felt that a mail survey would allow respondents the necessary time to assimilate information and form their bids accordingly. This follows from the argument that when making a market purchase for an unfamiliar good a consumer would gather information from a range of source including others (Carson, 1998). It is not possible to obtain this type of extra information during a face to face or telephone survey. So, for reasons of expense, geographical range of respondents, the desire to include visual material and the assumed familiarity of the respondent with the good the survey was delivered by mail.

However, it is acknowledged that the potential for a low response rate remains and needs to be addressed. Also, the sequencing of questions, the avoidance of open-
ended and tedious question need be given special consideration with the use of this method. To overcome these difficulties the design features of the questionnaire have been guided by Dillman (1999) *Tailored Design Method* and its precursor *The Total Design Method* (Dillman, 1978). This method has been followed for the design of other Australian contingent valuation questionnaires such as Jakobsson and Dragun (1996) and Bennett *et al.*, (1997). Mitchell and Carson (1989) also note that it is through the methodological advances by Dillman (1978), and others in the design of mail surveys that response rates have generally improved. An outline of these methods and rationale is given below.

### 7.11 SURVEY IMPLEMENTATION

The survey implementation followed the rationale used by Dillman (1978, 1999). This rationale is based on considerations of social exchange that aim to maximise the response to surveys and to minimise errors. In order to get respondents to answer self-administered surveys they must be motivated. Therefore, it is necessary to increase the perceived rewards for responding, decrease the perceived costs and promote trust that the outcomes of the research are beneficial. To achieve this outcome Dillman (1999) provides detailed guidance on the preparation of the survey, from question wording and design of the questionnaire to the timing of mailings.

Following these guidelines, the wording of all questions was designed to minimise respondent misunderstanding and to be simple (but not subordinating) and unambiguous. The sequencing of questions was designed to maintain respondent interest and avoid measurement errors due to missed questions. The initial question was designed to be closed-ended, directly related to the stated purpose of the survey and easy for all respondents to answer. This was followed by a series of questions related to attitudes and awareness of use of the Bay respectively. Information was then provided for the differing cases and the threats to the wetlands. Willingness to pay was then elicited and follow up questions asked of those unwilling to pay. Questions related to income and socio-economic variables were placed last as they are often found objectionable by respondents (Mitchell and Carson, 1989).
It was suggested by Dillman (1999) that the preferred format for the questionnaire is in booklet form. This format is familiar to respondents, easy to handle and reduces the possibility of missing pages. In summary, Dillman (1999, 134) provides 28 construction principles related to a clear set of objectives about how the written page should look, instructions about how respondents should process information and implicit propositions about what visual features will achieve that objective. Following these guidelines, a ten page booklet, on A4 paper folded in half was produced. The full text of the questionnaire is presented in Appendix D.

The area of survey design which is argued to have greatest influence over the response rate, is the implementation procedure. Dillman (1999) suggests five contacts should be made with the respondents to maximise response rates. However, due to financial constraints only the three following contacts were made for both the pilot and main survey.

1. A questionnaire mailing, including a detailed covering letter, and stamped return envelope.
2. A thank you postcard sent a few days to a week after the questionnaire. The card includes a thank you for those who have returned the survey and asks others to return the survey.
3. A replacement questionnaire is sent to non-respondents 2-4 weeks after the initial mailing. The tone of the cover letter was altered to induce response.

Given Dillman’s (1999) emphasis on the importance of the personalisation of all correspondence each letter and postcard was personally signed. The text for the two covering letters and the postcard is given in Appendix C.

7.12 CONCLUSION

This chapter has outlined the rationale and method used to develop and implement a contingent valuation survey of the wetlands of Moreton Bay. The aims of testing the effects of information on willingness to pay, particularly for ecological attributes have been outlined. The methods used for random sampling, sample selection and sample size have been detailed. The development of the survey has been guided by the information presented in Chapters 5 and 6. This includes a description of the
good in four differing versions and an outline of the present condition of the good
and the alternative given the hypothetical scenario is undertaken. A method of
collecting a once only payment has been devised and the use of iterative bidding
technique justified. Further questions have been developed to understand the
attitudes of the respondents and their prior knowledge and use of the Bay. The
questions related to socio-economic status have been outlined. Finally, a review of
survey delivery and questionnaire design from a market research perspective has
been presented. The details of the pilot survey from which the bids were derived are
given in Chapter 8 and the results of the main survey are provided in Chapter 9.
A pilot contingent valuation study of the wetlands of Moreton Bay was undertaken prior to the main survey. The purpose of this chapter is to outline briefly the survey design, the aims of the pilot study, the sampling logistics and implementation. Analysis is provided for the survey response, the attitudes of respondents, their use and awareness of the Bay and their willingness to pay for wetland protection. Reason for refusals or zero bids are examined and the socio-economic characteristics of the sample are compared with that of the population. In conclusion, recommendations for the main study resulting from the pilot study are given.

### 8.1 Survey Design

The purpose in undertaking the contingent valuation study was to investigate further the linkages between ecological and economic value. In particular, the study was designed to assess if ecological value could be captured using this technique. The study was also designed to test if the provision of information about differing economic values of the wetlands produces differing willingness to pay amounts. To this end, four different versions of the survey were produced. Case A provided no extra information so it could be used as a control. Case B included information about the ecological values of the wetlands of Moreton Bay. Case C provided information about the economic use values of the wetlands in the Bay including direct and indirect use. Case D provided information about the non-use values of endangered species resident in the Bay that are dependent on the wetlands. The derivation of this information and its rationale are outlined in Chapter 7.

The theoretical considerations for the rationale of the survey have been outlined in Chapter 5. The specific issues that needed to be addressed with respect to valuing the wetlands of Moreton Bay have been given in Chapter 7. The wording of the pilot study and the information included in the four differing cases is given in Appendix B.
8.2 AIMS OF THE PILOT STUDY

The pilot study was undertaken for several purposes. The primary purpose was to determine the first quartile, median and third quartile values of willingness to pay bids from an open bidding technique. These values can then be presented to respondents in an iterative bidding game in the main survey. The purpose of determining these values was to minimize the possibility of starting point bias associated with bidding games. The pilot study was also undertaken to:

- Determine if mail delivery would produce an adequate response rate.
- Test if respondents had sufficient knowledge of the wetlands of Moreton Bay to formulate a monetary valuation. The geographical area from which the sample was drawn was chosen with the aim that proximity to Moreton Bay would increase use and awareness.
- Test the survey instrument for plausibility and consistency with respect to choice of questions, information and payment vehicle.

8.3 SAMPLING LOGISTICS

The sample was drawn from the same population as in the main survey. That is, the sample was drawn from the population that resided in the Commonwealth Electoral Districts that bordered Moreton Bay. The Commonwealth Electoral Rolls include Australian citizens over the age of 18. The geographical location of these districts extends from Caloundra to Southport and is illustrated in Map 7.1. It was considered that they who lived in close proximity to the Bay would be more likely to be familiar with the wetlands of Moreton Bay or at least aware of the Bay and its values.

Simple random sampling was used to draw a sample of 398 units. Each element on the rolls was given a number between 1 and 591 155. Then 398 different random numbers were generated between 1 and 591 155. These numbers were then used to select the sample.

As for the main study, four different versions of the survey were sent to respondents. This resulted in approximately 100 surveys being sent for each case.
8.4 IMPLEMENTATION

The survey was delivered by mail for the reasons outlined in Chapter 7. The implementation procedure followed the guidelines of Dillman (1999) however only three contacts were made. The first contact (11th May, 2000) consisted of mailing 398 surveys, including a covering letter and stamped return envelopes to the respondents. A week later (18th May, 2000) a thank you postcard with a reminder for those who had not responded was sent. Two weeks later (1st June, 2000) a replacement survey and letter was sent to the 206 respondents who had not responded to the earlier mailings. Given Dillman’s (1999) emphasise on the importance of the personalisation of all correspondence each letter and postcard was personally signed. The text for the two covering letters and the postcard was also used for the main survey and is given in Appendix C.

8.5 RESPONSE RATE

The survey achieved a response rate of useable replies of 60.3% which is higher than the usual range obtained by other contingent valuation mail surveys of 30-60% (Jakabsson and Dragun, 1996). The details are outlined in Table 8.1.

<table>
<thead>
<tr>
<th>Response</th>
<th>Number</th>
<th>Percent</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total questionnaire posted</td>
<td>398</td>
<td>100</td>
</tr>
<tr>
<td>Undeliverable or unwilling</td>
<td>158</td>
<td>39.7</td>
</tr>
<tr>
<td>Useable reply</td>
<td>240</td>
<td>60.3</td>
</tr>
</tbody>
</table>

Four different types of information were provided to respondents to test the effects of information provision and in particular the effect of providing ecological information to respondents. The response rate for each of cases for the useable replies received are presented in Table 8.2.
Table 8.2  Response Rate For The Different Cases

<table>
<thead>
<tr>
<th>Case</th>
<th>Number</th>
<th>Percent</th>
</tr>
</thead>
<tbody>
<tr>
<td>A - No information</td>
<td>67</td>
<td>27.9</td>
</tr>
<tr>
<td>B - Ecological Value</td>
<td>57</td>
<td>23.8</td>
</tr>
<tr>
<td>C - Use value</td>
<td>62</td>
<td>25.8</td>
</tr>
<tr>
<td>D - Non-use value</td>
<td>54</td>
<td>22.5</td>
</tr>
<tr>
<td>TOTAL</td>
<td>240</td>
<td>100.00</td>
</tr>
</tbody>
</table>

Table 8.2 indicates that there was little variation in the response received for each case. Given the good response rate the choice of survey delivery by mail was considered acceptable.

8.6 ANALYSIS OF RESULTS OF PILOT STUDY

The results of the pilot study were analysed to determine the general attitudes of respondents to wetland protection and their use and awareness of the Bay. Analysis of the willingness to pay bids for all case studies was then undertaken. The reasons for refusal or zero bids were then considered to determine if there were particular areas of protest that could be altered for the main survey. Finally, the socio economic characteristics of the respondents were considered to ensure the sampling method produced a sample with characteristics similar to the population from which it was drawn.

8.6.1 Attitudes

The first three questions in the survey were designed to elicit attitudinal information with respect to wetland values and protection. The first question asked respondents if they agreed or not that the wetlands of Moreton Bay should be protected. This question was designed to be easily answered by all respondents thus encouraging them to continue the survey. The item received a response rate of 97.5% indicating that respondents were able or willing to answer the question. 96.6% of respondents answered that they either slightly or strongly agreed that the wetlands of Moreton Bay should be protected.

The next question asked respondents to rank health, direct use, indirect use, bequest and existence values from extremely important to not at all important. This question
received a response rate of 96.7%, which again indicated that respondents understood it. Table 8.3 gives the mean ranks and significant differences between respondents’ values. In all cases over 90.6%\(^1\) of respondents considered the statements very important or extremely important with the exception of direct use (81.2%).

Table 8.3  Responses to attitudinal question– the importance of wetland values.

<table>
<thead>
<tr>
<th>Statement</th>
<th>Mean Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>Health</td>
<td>4.52</td>
</tr>
<tr>
<td>Direct Use</td>
<td>4.22</td>
</tr>
<tr>
<td>Indirect Use</td>
<td>4.52</td>
</tr>
<tr>
<td>Bequest</td>
<td>4.45</td>
</tr>
<tr>
<td>Existence</td>
<td>4.65</td>
</tr>
<tr>
<td>All Statements</td>
<td>4.47</td>
</tr>
</tbody>
</table>

Note: Scale 1 ‘not at all important’ 5, ‘extremely important’. Analysis conducted using Freidman’s non-parametric ANOVA indicated a difference in ranks (p<0.01). A Wilcoxon signed rank test was conducted between pairs to determine the order of ranking. There was not a significant difference for the pairs indirect use & health, bequest & indirect use and bequest & health. All other pairs were significantly different (p<0.01) with existence values ranked highest and direct use ranked lowest.

The third attitudinal question was a constant sum question, which presented respondents with the same five statements of wetland values as in the previous question. They were then asked to allocate 100 points between these statements to reflect the relative importance of each statement. Although this task may have required greater thought 97.1% of respondents attempted to answer it. Table 8.4 presents the mean value given for each statement.

\(^1\) Responses to statements that the wetlands were very important or extremely important were: health 92.7%, indirect use 92.2% bequest 90.6% and existence 96.6%.
Table 8.4 Mean value of points allocation for each statements

<table>
<thead>
<tr>
<th>Statement</th>
<th>Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>Health</td>
<td>22.17</td>
</tr>
<tr>
<td>Direct Use</td>
<td>16.39</td>
</tr>
<tr>
<td>Indirect Use</td>
<td>20.20</td>
</tr>
<tr>
<td>Bequest</td>
<td>17.08</td>
</tr>
<tr>
<td>Existence</td>
<td>24.39</td>
</tr>
<tr>
<td>All Statements</td>
<td>20.05</td>
</tr>
</tbody>
</table>

Note: A univariate analysis of variance was conducted of the arcsin of the statement value (F=22.24, p<0.01). Post-hoc tests for the least square difference (LSD) indicated that there was not a significant difference between direct use and bequest and Tukey honestly significant (HSD) indicate that there was not a significant difference between health and indirect use, health and existence and direct use and bequest.

These results indicate that existence, and indirect use are ranked higher than direct use value. It could thus be expected that willingness to pay for Case B (ecological value) and Case D (existence value) would also be higher than that of Case C (use values). This ranking of value can be compared with ranking of ecological and economic value derived from mean willingness to pay for wetland protection. Overall, there seems to be a preference for wetland protection by the respondents.

8.6.2 Awareness and Use of Moreton Bay

Respondents to the pilot survey may already possess information about the various values of the wetlands of Moreton Bay. It is assumed *a priori* that those who have visited the area, or use the area for bird watching or fishing would have some knowledge of the values of wetlands. There have also been a number of publicity campaigns recently undertaken in the area relating to the zoning of the Moreton Bay Marine Park, the ‘Healthy Waterways’ program and as part of a commercial television station promotion of the ‘Great South East’ [Queensland]. Therefore, questions were asked about use of and awareness of the Bay to determine the existing level of knowledge of the wetlands of Moreton Bay.
The counts for use and mean number of activities undertaken in Moreton Bay as reported by respondents are given in Table 8.5.

Table 8.5 Counts for use and mean number of activities on Moreton Bay

<table>
<thead>
<tr>
<th>Use Category</th>
<th>Count</th>
<th>Percent of total sample</th>
<th>Mean no. of activities in past 12 months</th>
</tr>
</thead>
<tbody>
<tr>
<td>Visited</td>
<td>185</td>
<td>77.4</td>
<td>39.84</td>
</tr>
<tr>
<td>Fishing</td>
<td>99</td>
<td>41.4</td>
<td>5.69</td>
</tr>
<tr>
<td>Bird Watching</td>
<td>34</td>
<td>14.2</td>
<td>7.98</td>
</tr>
<tr>
<td>Water Based Activity</td>
<td>118</td>
<td>49.4</td>
<td>12.34</td>
</tr>
<tr>
<td>Other Activity</td>
<td>67</td>
<td>28.0</td>
<td>6.96</td>
</tr>
</tbody>
</table>

This result indicates that 77.4% of respondents had made some use of Moreton Bay through visits in the past twelve months. From this, it could be concluded that respondents are familiar with the good. Thus, the sample is being drawn from the correct geographical area in terms of having some knowledge of the resource being valued. One difficulty encountered with the questions related to use arose for those respondents who considered that they lived at the Bay. In many cases they recorded their level of use as daily. The proximity to the Bay was not measured in the survey and thus the concept of ‘living at the Bay’ is subjective. To overcome this problem it was decided to include a question asking whether respondents considered that they lived at the Bay in the main survey.

To test if respondents were aware of the Bay through local media sources they were asked if they had read newspaper articles, heard radio programs, seen television programs about Moreton Bay and also if they were aware of the Healthy Waterways campaign\(^2\). The level of awareness of the Bay from these sources is given in Table 8.6. As 64.4% of respondents had seen television programs about the Bay and 61.1% had read newspaper articles it was considered that there was a reasonable level of awareness of the Bay within the sample.

---

\(^2\) The *Healthy Waterways Campaign* is an educational and publicity component of the South-east Queensland Regional Water Quality Management Strategy
Table 8.6  Level of Awareness of Moreton Bay from the local media

<table>
<thead>
<tr>
<th>Media</th>
<th>N</th>
<th>Percent</th>
</tr>
</thead>
<tbody>
<tr>
<td>Newspaper</td>
<td>146</td>
<td>61.1</td>
</tr>
<tr>
<td>Radio</td>
<td>73</td>
<td>30.5</td>
</tr>
<tr>
<td>Television</td>
<td>154</td>
<td>64.4</td>
</tr>
<tr>
<td>Health Waterways</td>
<td>84</td>
<td>35.1</td>
</tr>
</tbody>
</table>

The results presented in Tables 8.5 and 8.6 suggest that respondents had sufficient knowledge of the wetlands of Moreton Bay to formulate a monetary valuation. Thus, it is concluded that the geographical area from which the sample was drawn was appropriate to ensure some knowledge of the Bay.

### 8.6.3 Willingness to Pay

A primary purpose of conducting the pilot study was to establish a range of values to be presented to respondents in the main survey. This was achieved by asking respondents the maximum amount they would be willing to pay to improve water quality and hence protect the wetlands, as a one off contribution. The payment vehicle was the same as that used in the main survey. However, only 23.3% of respondents were willing to offer a positive bid. This may be the result of the open-ended nature of the question. Similar studies conducted by Stone (1992, 59) and Kirkland (1988, 109) reported a response to the willingness to pay question of 21% and 25.6% respectively which indicates the response to this question may not be unusual. Given that the purpose of the question was to establish the first quartile, mean and third quartile values, only positive bids were included in the analysis of the mean willingness to pay. The mean willingness to pay and standard deviation for each case appears in Table 8.7.
Table 8.7  Willingness to pay for all cases.

<table>
<thead>
<tr>
<th>Survey type</th>
<th>N</th>
<th>Mean</th>
<th>Std Deviation</th>
</tr>
</thead>
<tbody>
<tr>
<td>No information</td>
<td>13</td>
<td>40.75</td>
<td>40.34</td>
</tr>
<tr>
<td>Ecological Value</td>
<td>13</td>
<td>31.56</td>
<td>32.77</td>
</tr>
<tr>
<td>Use Value</td>
<td>18</td>
<td>36.11</td>
<td>36.84</td>
</tr>
<tr>
<td>Non-use Value</td>
<td>12</td>
<td>34.30</td>
<td>37.99</td>
</tr>
<tr>
<td>TOTAL</td>
<td>56</td>
<td>35.68</td>
<td>36.12</td>
</tr>
</tbody>
</table>

Note: Results of ANOVA conducted indicate that there is no significant difference between survey types (F=0.135, p>0.05)

From the range of values given the first quartile, median and third quartile values were determined. These values appear in Table 8.8. The estimates of the mean and standard deviation can also be used to estimate the necessary sample for the main survey, given the chosen confidence intervals.

Table 8.8  Quartiles for willingness to pay bids

<table>
<thead>
<tr>
<th>Percentiles</th>
<th>Value $</th>
</tr>
</thead>
<tbody>
<tr>
<td>25</td>
<td>10.00</td>
</tr>
<tr>
<td>50</td>
<td>20.00</td>
</tr>
<tr>
<td>75</td>
<td>50.00</td>
</tr>
</tbody>
</table>

The quartiles of willingness to pay bids given in the pilot survey provide the bids to be offered to respondents in the main survey. Therefore, the main survey will offer an initial bid of $20 to respondent for their willingness to pay. Those who agree to this amount will be asked if they would be willing to pay $50. Those who unwilling to pay $20 will be asked if they would be willing to pay $10. All respondents will then be asked for the highest amount that they would be willing to pay.

8.6.4  Refusals And Zero Bids

Following the willingness to pay question, respondents who were unwilling to offer a positive amount were asked their reason for doing so. A list of statements was presented to respondents and they could choose more than one reason. The results of this question can be used to indicate if respondents truly hold no value for the wetlands or if they are protesting about some other aspect of the survey. In particular, one purpose of the pilot study was to determine if the payment vehicle
was acceptable to respondents and to test if they had sufficient information to form a monetary value for the wetlands. The list of statements and the response for each for those 176 respondents who were unwilling to over a positive bid is presented in Table 8.9.

<table>
<thead>
<tr>
<th>Statement</th>
<th>N</th>
<th>Percent</th>
</tr>
</thead>
<tbody>
<tr>
<td>I did not have enough information to place a $ value on protecting the wetlands</td>
<td>35</td>
<td>19.9</td>
</tr>
<tr>
<td>I did not want to place a $ value on protecting wetlands</td>
<td>21</td>
<td>11.9</td>
</tr>
<tr>
<td>I disagree with paying money into a fund</td>
<td>17</td>
<td>9.7</td>
</tr>
<tr>
<td>The wetlands are worth nothing to me therefore I don’t want to pay</td>
<td>2</td>
<td>1.1</td>
</tr>
<tr>
<td>The government should pay to protect the wetlands using taxes already paid</td>
<td>111</td>
<td>63.1</td>
</tr>
<tr>
<td>I couldn’t afford to pay anything</td>
<td>64</td>
<td>36.4</td>
</tr>
<tr>
<td>Other –1 (comment)</td>
<td>23</td>
<td>13.1</td>
</tr>
<tr>
<td>- 2 (pensioner)</td>
<td>9</td>
<td>5.1</td>
</tr>
</tbody>
</table>

Note: Total percent greater than 100 as respondents were able to make multiple responses

It was considered that those respondents who indicated either that the wetlands were worth nothing to them or they couldn’t afford to pay truly had a zero value for the wetlands. Only two respondents felt the wetlands were worth nothing to them however, 36.4% of respondents felt that they could not afford to pay. The other responses are considered a protest to some other aspect of the survey.

Of all the respondents only 17 (9.7%) refused to offer a positive willingness to pay bid because they objected to the payment vehicle (a trust fund). In comparison, 63.1% of those unwilling to pay, felt that it was the government’s responsibility to protect the wetlands. This may be considered a protest about paying for what is perceived as a public good. It could also be concluded that asking respondents to contribute further to government funding through the use of taxes as payment vehicle would result in further protest bids. Given the much lower protest to the use of a trust fund it was decided to maintain this as the payment vehicle for the main survey. To reduce further the protest response to government spending the wording of the statement was altered to ‘the government should transfer spending from other areas
and pay to protect the wetlands’ to emphasis that a trade-off in government spending would be necessary to protect the wetlands. Also, the instrument design was altered slightly so that the questions to determine refusals were placed on the page behind the bid elicitation question rather than below it.

Of the respondents 19.9% felt they did not have enough information and therefore their unwillingness to pay was a protest about information. Of the 35 respondents who gave this answer 10 had received Case A of the survey (no information) and more interestingly 15 had received Case C (use information). There were five responses in the category for Case B (ecological information) and Case D (non-use information) respectively. However, as a proportion of the total useable replies (N=240) the percentage of respondents who felt they had insufficient information was only 14.6%. Given the potential for information overload no further information was provided in the main survey.

Those respondents who stated that they did not want place a dollar value on protecting wetlands may have been making a moral protest. That is, they were unwilling to trade wetland protection and may be considered to hold a lexicographic position.3 The ‘other’ category for refusal to pay was given by 18.2% of respondents. Of these 5.1% were pensioners who may be considered as unable to pay. The other 13.1% were comments about some other aspect of the survey.

In conclusion, the responses to the question of refusal to offer a positive willingness to pay bid suggests that the voluntary trust fund is an appropriate payment vehicle and that sufficient information was provided in the survey. To avoid the high response to the notion that the government should pay the wording of this statement was altered to stress governments would need to shift funding from other areas.

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3 The concept of lexicographic preferences has been discussed in Chapters 5 and 7. If individuals hold an ethical position where they refuse to trade off environmental quality for money they are said to hold a lexicographic position (Spash and Hanley, 1995, 192).
8.6.5 Socio-Economic Characteristics

Respondents were asked to indicate their socio-economic characteristics in terms of sex, age, education and income. The purpose of collecting this information was to determine if the sample has the same characteristics as the population from which it was drawn. This information can also be used to determine if there is a differing willingness to pay for the different groups in the main survey. The categories were chosen to correspond with data collected from the 1996 national population census.

These questions received the highest item non-response. From comments received from respondents it appears that some found these questions an invasion of their privacy and not relevant to the survey. For example, 15% of respondents refused to provide information about their income. Of those who answered the question 6.4% stated that they didn’t know their income. The response for each question is given in Table 8.10.

<table>
<thead>
<tr>
<th>Characteristic</th>
<th>N</th>
<th>Percent</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sex</td>
<td>236</td>
<td>98.3</td>
</tr>
<tr>
<td>Age</td>
<td>234</td>
<td>97.5</td>
</tr>
<tr>
<td>Education</td>
<td>229</td>
<td>95.4</td>
</tr>
<tr>
<td>Income</td>
<td>204</td>
<td>85.0</td>
</tr>
</tbody>
</table>

A difficulty arose with the education categories as they were confusing for respondents. For example, the category of post-bachelor degree was included but not bachelor degree. Also, the categories selected were not exclusive. For example, the question asked if they had left school between ages of 14-19. However, 85.62% of the population left school in this age group and had another education category. Therefore, it was decided to change the categories for the main survey although it would not be possible to compare these with the larger population.

The distribution of respondents by gender and age are consistent with distribution in the Commonwealth Electoral Districts that border Moreton Bay (p<0.05). The results are given in Table 8.11
Table 8.11 Comparison of the sample and population socio-economic variables

<table>
<thead>
<tr>
<th>Sex</th>
<th>Survey Frequency</th>
<th>Expected frequency from Census data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Male</td>
<td>122</td>
<td>114</td>
</tr>
<tr>
<td>Female</td>
<td>114</td>
<td>122</td>
</tr>
</tbody>
</table>

$\chi^2 = 1.08, (p<0.05)$

<table>
<thead>
<tr>
<th>Age</th>
<th>Survey Frequency</th>
<th>Expected frequency from Census data</th>
</tr>
</thead>
<tbody>
<tr>
<td>15-24</td>
<td>24</td>
<td>30</td>
</tr>
<tr>
<td>25-34</td>
<td>38</td>
<td>45</td>
</tr>
<tr>
<td>35-44</td>
<td>42</td>
<td>48</td>
</tr>
<tr>
<td>45-54</td>
<td>54</td>
<td>41</td>
</tr>
<tr>
<td>55-64</td>
<td>34</td>
<td>27</td>
</tr>
<tr>
<td>65-74</td>
<td>20</td>
<td>25</td>
</tr>
<tr>
<td>&gt;75</td>
<td>22</td>
<td>18</td>
</tr>
</tbody>
</table>

$\chi^2 = 10.84, (p<0.05)$

<table>
<thead>
<tr>
<th>Weekly Income</th>
<th>Survey Frequency</th>
<th>Expected frequency from Census data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Less than $79</td>
<td>11</td>
<td>6</td>
</tr>
<tr>
<td>$80 - $159</td>
<td>26</td>
<td>29</td>
</tr>
<tr>
<td>$160 - $299</td>
<td>39</td>
<td>37</td>
</tr>
<tr>
<td>$300 - $499</td>
<td>32</td>
<td>43</td>
</tr>
<tr>
<td>$500 - $699</td>
<td>31</td>
<td>36</td>
</tr>
<tr>
<td>$700 - $999</td>
<td>32</td>
<td>25</td>
</tr>
<tr>
<td>$1000 - $1499</td>
<td>16</td>
<td>10</td>
</tr>
<tr>
<td>&gt; $1500</td>
<td>4</td>
<td>5</td>
</tr>
</tbody>
</table>

$\chi^2 = 13.84, (p<0.05)$

Table 8.11 illustrates that none of the sample characteristics is significantly different from the census data at the 95% level. This confirms that the sampling technique used was able to draw a sample with characteristics similar to the population from which it was obtained.
8.7 CONCLUSION AND RECOMMENDATIONS

The pilot study was able to achieve the stated aim of determining the first quartile, median and third quartile amounts ($10, $20, $50) to be presented in an iterative bidding technique for the main study. As these amounts were derived from an open-ended question, the possibility of starting point bias will be minimised for the main survey.

The pilot study was also able to demonstrate that the survey instrument achieved a good response rate (60.3%) using mail delivery. There was also a good item response rate which indicates that the questions were understood and the respondents were motivated to answer them. The attitudes of respondents indicated a positive preference for wetland protection. The ranking of the ecological and economic values in the attitudinal questions allows further consistency checks with that of willingness to pay for the four cases. However, in the pilot study there was no significant difference between the mean willingness to pay for the four cases. The study also demonstrated that respondents have a high level of use and awareness and hence do value the Bay itself. This indicates that respondents have sufficient knowledge with which to form a monetary value for the wetlands. Further, it confirms that the geographical scope of the sample is adequate to ensure some familiarity with the good being valued.

The choice of payment vehicle, a trust fund, was well accepted by the respondents with only 9.7% of those who were unwilling to offer a positive bid protesting against its use. The provision of information also appears adequate with only 14.6% of the entire sample indicating that they did not have enough information to value the wetlands.

The pilot study also served to test and refine the survey instrument. Respondents who considered that they lived at the Bay recorded daily usage of the Bay, which may inflate the values for use. To overcome this respondents were asked if they considered that they lived at the Bay in the main survey. It also appeared that many those who were unwilling to pay (63.1%) were protesting about paying for what they
perceive as a public good. This may also be a protest about government spending in
general. To indicate that further funding is necessary to carry out the strategies
designed to improve water quality the statement was changed to indicate that the
government would need to transfer funding from other areas to meet this need. The
layout of the question was also slightly altered to encourage respondents to give
greater thought to the willingness to pay question before simply recorded a protest
response. Within the socio-economic questions, the education categories were found
confusing and non-exclusive. These categories were changed for the main survey.
CHAPTER 9
ANALYSIS OF RESULTS OF CONTINGENT VALUATION SURVEY

This chapter analyses the results from the contingent valuation survey on the protection of wetlands in Moreton Bay, which was outlined in Chapter 7. The results and recommendations from the pilot study have been presented in Chapter 8. A primary purpose of the survey was to determine if ecological values could be captured using this method. The survey was also designed to test if the provision of different information with respect to the values of the wetlands produced differing willingness to pay amounts. To assess these propositions and understand the motivation for willingness to pay responses the results are analysed to determine the attitudes of respondents with respect to protection of wetlands and the components of total economic value that were considered most important. As the effect of information was a focus of the survey, responses were also analysed with respect to previous knowledge and use of the Bay.

In order to place this analysis in context, the chapter begins with an outline of the response rate for the survey, use and awareness of the study area and socio-economic characteristics of the respondents. The responses to the follow up question for those unwilling to offer a positive amount are then discussed. This is followed by an analysis of the results from questions in the survey related to attitudes and willingness to pay. From the analyses the attitudes respondents expressed about wetland values are compared to their willingness to pay with respect to the provision of information in the surveys. A regression analysis using the Tobit method is then undertaken to investigate further differences in variables that may have influenced willingness to pay.

9.1 BACKGROUND INFORMATION

As background this section documents the response rate to the survey. This is followed by an analysis of results related to the previous use, knowledge and awareness of the Bay to determine if respondents possess sufficient knowledge to form a willingness to pay bid. The socio-economic characteristics of the respondents
are presented and compared to that of the population from which the sample was
drawn. Finally, the results of questions posed to determine protest bids are given.

9.1.1 Response Rate

A total of 1600 surveys was sent to respondents for the main survey. An equal
number (N=400) were sent for the four versions of information. That is, 400 surveys
for Case A (no information), Case B (ecological information), Case C (use
information) and Case D (non-use information) were sent. Following the first
mailing, a postcard was sent one week later. Table 9.1 provides an outline of
responses.

Table 9.1 Response to Moreton Bay wetland survey

<table>
<thead>
<tr>
<th>Response</th>
<th>Number</th>
<th>Percent</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total questionnaires posted</td>
<td>1600</td>
<td>100.0</td>
</tr>
<tr>
<td>Undelivered</td>
<td>99</td>
<td>6.2</td>
</tr>
<tr>
<td>No response</td>
<td>482</td>
<td>30.1</td>
</tr>
<tr>
<td>Usable reply or unwilling</td>
<td>170</td>
<td>10.7</td>
</tr>
<tr>
<td>Usable reply</td>
<td>849</td>
<td>53.0</td>
</tr>
</tbody>
</table>

From the first two mailings a total of 731 responses were received including 60
letters which were undeliverable. A third mailing of 815 surveys was undertaken
which produced 333 responses including 39 that were undeliverable. Thus, there
was a total of 849 useable replies and 99 undelivered surveys. The overall response
rate was 69.9%. However, the total useable response rate is calculated by including
all useable responses divided by the total number of surveys sent minus those which
were undeliverable or sent to deceased persons. The useable response rate was thus,
56.7%. An outline of unusable responses is given in and Table 9.2. The full text of
the survey, covering letters and postcard can be found in Appendices C and D.

1This terminology to describe the four different instruments will be retained for the discussion of the
results.
Table 9.2  Reasons for unusable responses

<table>
<thead>
<tr>
<th>Reason</th>
<th>Number</th>
<th>Percent</th>
</tr>
</thead>
<tbody>
<tr>
<td>Deceased</td>
<td>3</td>
<td>1.76</td>
</tr>
<tr>
<td>Too old or too ill</td>
<td>18</td>
<td>10.59</td>
</tr>
<tr>
<td>Didn’t speak English</td>
<td>6</td>
<td>3.53</td>
</tr>
<tr>
<td>Unwilling</td>
<td>143</td>
<td>84.12</td>
</tr>
<tr>
<td><strong>TOTAL</strong></td>
<td><strong>170</strong></td>
<td><strong>100</strong></td>
</tr>
</tbody>
</table>

The useable response rate of 56.7% is comparable with other contingent valuation mail surveys of wetlands. Kirkland (1988), for example, achieved a 54.1% useable response rate, Bergstrom, *et al.*, (1990) 62% and Sappideen (1992) 30.7%.

Table 9.3 gives a breakdown of the total number of useable responses by survey instrument. The response rate across the survey instruments was consistent at 25.6% with the exception of ecological information, which had a slightly lower response rate of 23.3%.

Table 9.3  Response Rate for the Different Cases

<table>
<thead>
<tr>
<th>Survey Type</th>
<th>Number</th>
<th>Percent</th>
</tr>
</thead>
<tbody>
<tr>
<td>Case A - No information</td>
<td>217</td>
<td>25.6</td>
</tr>
<tr>
<td>Case B - Ecological information</td>
<td>198</td>
<td>23.3</td>
</tr>
<tr>
<td>Case C - Use information</td>
<td>217</td>
<td>25.6</td>
</tr>
<tr>
<td>Case D - Non-use information</td>
<td>217</td>
<td>25.6</td>
</tr>
</tbody>
</table>

The high response rate provided sufficient information to conduct analysis of the responses for the entire survey and compare results across cases.

**9.1.2 Use and Awareness**

This section explores the hypothesis presented in Chapter 7 to determine if respondents possess a sufficient knowledge from use of the Bay and awareness campaigns to form a value for the wetlands. It was considered that respondents may have possessed previous knowledge of the wetlands if they had visited them in the past twelve months. It was expected that those who visit the Bay would have a positive willingness to pay for use values, however those that don’t may still hold non-use values. To further test previous knowledge and awareness of the wetlands
respondents were asked if they had heard information about the wetlands from various media and the Healthy Waterways campaign. If the media and the Healthy Waterways campaign have been successfully in making respondents aware of problems and possible solution to water quality issues in the Bay it would be expected that scenario presented was considered to be credible. This might in turn increase willingness to pay.

Cummings *et al.*, (1986) and Hanley (1989) argue that the ideal conditions for a contingent valuation study are present when the respondents understand and are familiar with the commodity to be valued. Where this is the case respondents will already possess information about the good to be valued. To comply with these conditions the sample was drawn from the electoral divisions that bordered Moreton Bay with the expectation they would have the required understanding and familiarity of the wetlands of Moreton Bay. To test if this was indeed the case respondents were asked about their use and awareness of the Bay.

It was assumed *a priori* that those respondents who have visited the study area or used it for bird watching, fishing or other water activities would have some knowledge of the wetlands of the Bay. Respondents were thus asked if any activities had been undertaken at or on the Bay and the frequency of use. Table 9.4 gives the counts for use of the Bay in the various categories, the percent of the total sample that had undertaken the activity and the mean number of activities undertaken in the past twelve months.

<table>
<thead>
<tr>
<th>Use Category</th>
<th>Count</th>
<th>Percent of total sample</th>
<th>Mean no. of activities in past 12 months</th>
</tr>
</thead>
<tbody>
<tr>
<td>Visited</td>
<td>648</td>
<td>76.3</td>
<td>23.45</td>
</tr>
<tr>
<td>Fishing</td>
<td>296</td>
<td>34.9</td>
<td>3.82</td>
</tr>
<tr>
<td>Bird watching</td>
<td>138</td>
<td>16.3</td>
<td>4.33</td>
</tr>
<tr>
<td>Water activities</td>
<td>360</td>
<td>42.4</td>
<td>6.25</td>
</tr>
<tr>
<td>Other activities</td>
<td>225</td>
<td>26.5</td>
<td>6.12</td>
</tr>
</tbody>
</table>

2 However, those who understand and are familiar with the resource may value it more highly than others and hence bias results.
The results given in Table 9.4 indicate that 76.3% of the respondents had visited Moreton Bay in the past twelve months. However, 61 respondents who had undertaken one or more of the activities listed did not indicate that they had visited the Bay although it is difficult to conceive how this was possible. It was considered that this group had some experience of the Bay. Therefore, the total number of respondents who had some experience of the Bay was 709, which is equal to 83.5% of all respondents. It would be expected that this group had some knowledge of the Bay prior to being asked their willingness to pay. This information should improve the reliability and validity of the survey results. It is considered that respondents who are familiar with the good will be able to formulate a willingness to pay bid that reflects the true worth of the good to them (Anderson and Bishop, 1986).

The percent of respondents who stated that they been have engaged in recreational fishing was consistent with the state average of 33.2% estimated by the Queensland Fisheries Management Authority (Moreton Bay Task Force, 1997, 94). It could be expected that those who enjoy the direct use value of the Bay would be willing to pay to preserve the wetlands on which 75% of the fish stock depends, that is, the wetlands (Quinn, 1993). This expectation could also be extended to those who engage in water based activities. Due to a decline in water quality, some areas of the Bay are now considered unsuitable for direct contact which decrease the opportunities for water based activities (Dennison and Abal, 1999, 139).3

One difficulty with the above estimates is that those who visit the Bay regularly and engage in other activities may also live at the Bay. This situation was indicated by responses to the question of use of Moreton Bay in the pilot survey. Thus, the rate of visitation may be confounded by the fact that those who live at the Bay consider that they visit it daily. This group would however be expected to have a good knowledge and understanding of the Bay. Thus, respondents were asked if in their view they lived at the Bay. The answer given is therefore, somewhat subjective. The total number of respondents who felt they ‘lived at the Bay’ was 490 equal to 57.7% of the sample. This group could be expected to have a high usage of the Bay. For

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3 Differences in willingness to pay between those who use and not use the Bay is explored further in Section 9.2.2 and 9.4.
example, all respondents who indicated they visit the Bay daily (n=27) also reside there.

Respondents might also have gained knowledge of Moreton Bay from the local media. To test if this was the case respondents were asked if they had become aware of Moreton Bay through newspapers, radio, television and the *Healthy Waterways Campaign*. A number of publicity campaigns have recently been undertaken in the area in relation to the zoning of the Moreton Bay Marine Park, through a commercial television station promotion of the ‘Great South East’ [Queensland] and the *Healthy Waterways Campaign*. It should be noted that Moreton Bay had received considerable media coverage prior to the survey due to large blue-green algae (*Lyngbya*) bloom (for example, *Courier Mail*, 5th April, 2000).

Table 9.5 presents the results from the awareness question and indicates that the majority of respondents had gained some knowledge of Moreton Bay from the newspaper and television. Also, 46.5% of respondents had heard of the *Healthy Waterways* campaign which as noted is aimed at increasing awareness of water quality issues in Moreton Bay. This level of awareness adds further support for the notion that respondents did have a knowledge and understanding of the resource they were being asked to value.

<table>
<thead>
<tr>
<th>Media</th>
<th>Percent</th>
</tr>
</thead>
<tbody>
<tr>
<td>Newspaper</td>
<td>62.3</td>
</tr>
<tr>
<td>Radio</td>
<td>26.5</td>
</tr>
<tr>
<td>Television</td>
<td>59.5</td>
</tr>
<tr>
<td><em>Healthy Waterways</em></td>
<td>46.5</td>
</tr>
</tbody>
</table>

Given that 76.3% of respondents have visited the Bay at least once in the past twelve months and 57.7% of respondents stated that they lived at the Bay, it can be concluded that a majority of respondents had made direct use of the Bay. There also seemed to be a high level of awareness of the Bay through the media and the *Healthy Waterways* campaign. This indicates that before being provided with extra

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4 The *Healthy Waterways Campaign* is an educational and publicity component of the South-east Queensland Regional Water Quality Management Strategy.
information, most respondents had some knowledge, use or awareness of the wetlands of Moreton Bay. Therefore, with reference to hypothesis 3, in Chapter 7 it is considered that respondents did posses sufficient knowledge from use of the Bay and awareness campaigns to form a value for the wetlands.

9.1.3 Socio-Economic Characteristics

To determine if the sample was representative of the larger population from which it was drawn (N = 591 155) information was gathered in relation to the socio-economic characteristics of the respondents. Questions were asked about the sex, age, educational level and income of the respondents. The categories for sex, age and income were chosen to correspond with those used in the National 1996 census conducted by the Australian Bureau of Statistics. The census education categories were used in the pilot study but were confusing for respondents. For example, the category of post-bachelor degree was included but not bachelor degree. The categories were thus changed but it is not possible to compare these with the larger population. It appears from comments received that many of the respondents felt that these question were an invasion of their privacy and not related to the survey purpose.

Chi-square tests indicate that the distribution of respondents by gender is consistent with distribution in the Commonwealth Electoral Districts that border Moreton Bay, at the 95% level. There are however differences with respect to age and income. The results are given in Table 9.6.

<table>
<thead>
<tr>
<th>Sex</th>
<th>Survey Frequency</th>
<th>Expected frequency from Census data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Male</td>
<td>384</td>
<td>402</td>
</tr>
<tr>
<td>Female</td>
<td>451</td>
<td>433</td>
</tr>
</tbody>
</table>

$\chi^2 = 1.53, (p<0.05)$
Table 9.6 illustrates that respondents are under represented in the 18–34 age categories and over represented in the 35-54 age categories. The 55-74 age category was also slightly over represented. With respect to income, the survey respondents are concentrated more in the higher income levels than in the census data. The lowest income bracket is slightly over represented and the $300-$699 is also under represented. The higher income brackets, $1000 - $1500+ are over represented. The income distribution may been have distorted as this question had the highest non-response rate (13.7%) and 3.8% of respondents stated that they didn’t know their income.

The lack of similarity between age and income for the respondents and the population from which they were drawn suggests that the results of this survey cannot be extrapolated for the larger population. Another difficulty, which arises when attempting to extrapolate values from survey data, is the potential for non-
respondent bias. This may arise where it is unknown if those who did not respond to the survey hold the same values as those who did. Mitchell and Carson, (1989, 282) note that it is likely that non-respondents to mail surveys may hold a lower value for the good compared to those who respond and have the same demographic characteristics.

9.1.4 Refusals and Zero Bids

If respondents were unwilling to offer a positive amount in the willingness to pay questions they were asked why this was the case. The purpose of the question was to determine if respondents truly felt the Bay was worth nothing to them or if they were protesting about some other aspect of the contingent scenario (Hanley, 1989). For example, if respondents gave a zero bid because they did not want to take part in the bidding game or did not agree with a voluntary trust fund then their zero bids are considered protests and not a true zero valuation of the resource (Hanley, 1989). Jorgensen and Syme, (2000) suggest that from the list of options given to respondents the options ‘the wetlands are worth nothing to me’ and ‘I couldn’t afford to pay’ were considered the only true zero bids. All other options are considered protest bids. For example, ‘not enough information’ is a protest about lack of information (Hanley, 1989). Spash and Hanley (1995) argue that the statement ‘did not want to place a dollar value on protecting wetlands’ indicates a moral or lexicographic protest. Hanley (1989) further argues that statements such as ‘disagree with fund’ is a protest about the payment vehicle and ‘the government should pay’ is a protest about paying for what is perceived to be a public good.

Table 9.7 presents the responses to this question. Note, respondents were able to list more than one reason for a refusal or zero bid thus there were 602 responses from the 377 respondents who were not willing to offer a bid.
Table 9.7 Reasons for refusals and zero bids

<table>
<thead>
<tr>
<th>Reason</th>
<th>Number</th>
<th>Percent</th>
</tr>
</thead>
<tbody>
<tr>
<td>Not enough information</td>
<td>43</td>
<td>7.18</td>
</tr>
<tr>
<td>Did not want to give $ value</td>
<td>36</td>
<td>6.01</td>
</tr>
<tr>
<td>Disagreed with fund</td>
<td>82</td>
<td>13.69</td>
</tr>
<tr>
<td>Wetlands worth nothing to me</td>
<td>10</td>
<td>1.67</td>
</tr>
<tr>
<td>Government should pay</td>
<td>216</td>
<td>36.06</td>
</tr>
<tr>
<td>Couldn’t afford to pay</td>
<td>134</td>
<td>22.37</td>
</tr>
<tr>
<td>Other – a) pensioner</td>
<td>14</td>
<td>2.34</td>
</tr>
<tr>
<td>b) other comment</td>
<td>67</td>
<td>10.68</td>
</tr>
<tr>
<td>TOTAL</td>
<td>602</td>
<td>100</td>
</tr>
</tbody>
</table>

Anderson and Bishop (1986) suggest that protest and extreme bids should be disregarded using subjective judgement. Diamond and Hausman (1993) argue that zero bids should only be included if they truly represent zero valuations. Similarly, Hanley (1989) suggests that other zero bids which represent people who didn’t want to take part in the bidding game, or disapproved of having to pay anything for something which is currently available free of charge should be discarded. The follow-up questions to differentiate between true zero and protest bids were originally included for this purpose.

In contrast, Kahneman and Knetsch (1992) point out that while extremely high responses and zero bids may have a considerable effect on the mean there is no agreed way in which to draw a line beyond which responses will be rejected. Jorgensen and Syme (2000) point out the danger in deleting protest responses is that the sample may no longer be representative of the population to which the aggregate value are to be generalized and rather over-represent those who support paying for the proposed program. The problem according to Jorgensen and Syme (1995) is that the behavioural intentions of the respondents are unknown. Thus, if answers to willingness to pay questions are to be censored, Jorgensen and Syme (2000) argue it should not be on the basis of the payment offered. Given that there appears to be no universal agreement on a model for omitting responses, all bids have been included in the mean willingness to pay figures presented thus far.
The reasons for refusal to pay do, however, give some information with respect to the attitudes of the respondents. For example, only 6.01% of respondents expressed any moral objection to placing a monetary valuation on the wetlands. This group could thus be considered to hold a lexicographic position. This is a relatively small group compared to the studies of Stevens et al., (1991) and Spash and Hanley (1995) which found a quarter of respondents held such views.

The largest number of protest bids (36%) in this survey was given for the response that the government should pay. Jakobsson and Dragun (1996) and Kirkland (1988) had a 26.5%-37.2% and 15.6% response to a similarly worded question respectively. Jorgensen and Syme (2000, 259) found a similar agreement in studies on stormwater pollution abatement. Similarly, Bennett et al., (1997) in a study of South Australian wetlands found that there were more protest bids if income tax was used as the payment vehicle compared to a trust fund. The response to the statement that the government should pay can thus be considered as a protest against paying for a what is considered a public good and not an indication of the value of wetlands to the respondents. There was also a protest against the use the trust fund by 13.69% of respondents who were unwilling to pay which indicates some distrust that the money would be used wisely. Of those that stated they couldn’t afford to pay, 91.4% have incomes under $500/week, which indicates that they truly could not offer to make a payment regardless of their attitudes towards wetland protection.

9.2 COMPARISON OF INFORMATION TREATMENTS

The provision of differing information in each of the four cases is examined with respect to the attitudes of respondents and willingness to pay bids. The information treatment is compared on three measures: ranking, constant sum distribution and mean willingness to pay.

This section first examines the attitudes expressed by the respondents about the protection of wetlands in Moreton Bay. These expressions are directly related to the

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5 In the pilot study this statement read as ‘the government should pay to protect the wetlands using taxes already paid’. As over 63% of respondents who answered this question gave this response, the wording was altered to ‘the government should transfer spending from other areas and pay to protect the wetlands’.
components of total economic value discussed in Chapter 2. It was hypothesised that respondents’ attitudes to wetland protection may influence willingness to pay. In this question the variable, maintaining the health of the Bay, was used as a proxy for the ecological values of the wetlands. It was expected *a priori* that those respondents who felt that it was important to protect the wetlands to maintain the health of the Bay would be more willing to pay for protection given the contingent scenario. The differences in mean willingness to pay for the four cases is then presented and compared to the ranking of values from the attitudinal questions.

### 9.2.1 Attitudes

The first three questions in the survey were designed to elicit attitudinal information with respect to wetland values and protection. The first question asked respondents how strongly they agreed or disagreed with the statement that the wetlands of Moreton Bay should be protected. This question was designed to be easily answered by all respondents thus encouraging them to continue the survey. 90.7% (n=725) of respondents answered that they either slightly or strongly agreed that the wetlands of Moreton Bay should be protected. The next two questions were included to address hypothesis 4 in Chapter 7, that is to determine if respondents have a difference in the their attitude toward wetland protection depending on the economic value being assessed. Also, the notion that the particular information provided in each case may produce a difference in attitudes was explored.

*Ranking*

The second question asked respondents to rank health, direct use, indirect use, bequest and existence values from extremely important to not at all important. A Likert scale was used where not at all important =1 and extremely important =5. In all cases at least 92.2%[^7] of respondents considered the statements very important or extremely important with the exception of direct use (82.4%). Table 9.8 gives the mean ranks of respondents’ values.

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[^6]: In the pilot study only 9.7% of respondents gave this reason. It was thus considered a neutral payment vehicle.

[^7]: Responses to statements that the wetlands were very important or extremely important were: health 94.1%, indirect use 93.7% bequest 92.2% and existence 94.8%.
Table 9.8  Responses to attitudinal question– the importance of wetland values.

<table>
<thead>
<tr>
<th>Statement</th>
<th>Mean Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>Health</td>
<td>4.51</td>
</tr>
<tr>
<td>Direct Use</td>
<td>4.27</td>
</tr>
<tr>
<td>Indirect Use</td>
<td>4.55</td>
</tr>
<tr>
<td>Bequest</td>
<td>4.53</td>
</tr>
<tr>
<td>Existence</td>
<td>4.66</td>
</tr>
<tr>
<td>All Statements</td>
<td>4.50</td>
</tr>
</tbody>
</table>

Note: Scale 1 ‘not at all important’ 5, ‘extremely important’. Analysis conducted using Freidman’s non-parametric ANOVA indicated a difference in ranks (p=0.000). A Wilcoxon signed rank test was conducted between pairs to determine the order of ranking.

There is not a significant difference between direct use and health, bequest and indirect use and bequest and health. All other pairs were significantly different (p=0.000) with existence values ranked highest and direct use ranked lowest. These results indicate that existence value is considered the most important reason for protecting the wetlands of Moreton Bay while the direct use benefits are considered the least important. This result can then be compared to willingness to pay for the four cases to determine if the ranking of values is consistent.

**Constant Sum Allocation**

The third attitudinal question was a constant sum question, which presented respondents with the same five statements of wetland values as in the previous question. They were then asked to allocate 100 points between these statements to reflect the relative importance of each statement. Table 9.9 presents the mean value of points allocation given for each statement.
Table 9.9 Mean value of points allocation for each statement

<table>
<thead>
<tr>
<th>Statement</th>
<th>Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>Health</td>
<td>22.08</td>
</tr>
<tr>
<td>Direct Use</td>
<td>16.18</td>
</tr>
<tr>
<td>Indirect Use</td>
<td>19.71</td>
</tr>
<tr>
<td>Bequest</td>
<td>17.99</td>
</tr>
<tr>
<td>Existence</td>
<td>23.88</td>
</tr>
<tr>
<td>All Statements</td>
<td>19.97</td>
</tr>
</tbody>
</table>

Note: A univariate analysis of variance was conducted of the arcsin of the statement value ($F=55.355$, $p<0.01$). Post-hoc tests for the least square difference (LSD) and Tukey honestly significant difference (HSD) indicate that all statements are different from each other. There was no interaction between the statements and the different cases ($p>0.05$).

There was significant difference between the means of the five statements. Reference to the mean of each statement indicates that the statements from highest to lowest ranking are existence, health, indirect use, bequest and direct use.

The statements presented to respondents were designed to assess the attitudes towards the various components of use and non-use economic value and ecological value. However, in order to compare the five statements in the constant sum question with the three different types of values given in the four versions of the survey they were then combined into three statements. That is, responses to the statements direct use and indirect use were combined into use values, bequest and existence were combined into non-use values and health was unaltered\(^8\). Table 9.10 presents the mean value for each of the combined statements.

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\(^8\) Where health is considered a proxy for ecological value
Table 9.10 Mean value of points allocation for combined statements

<table>
<thead>
<tr>
<th>Combined Statements</th>
<th>Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>Health</td>
<td>22.08</td>
</tr>
<tr>
<td>Use</td>
<td>17.95</td>
</tr>
<tr>
<td>Non-use</td>
<td>20.93</td>
</tr>
<tr>
<td>All Statements</td>
<td>20.32</td>
</tr>
</tbody>
</table>

Note: Univariate analysis of variance was undertaken of the arcsin of the statement value (F=41.67, p=0.000). Post-hoc LSD test found a significant difference between all three statements but the Tukey HSD found that there was no difference between health and non-use value. The interaction between survey type and statement was also significant in this case (F=2.349, p<0.05).

These results indicate that the respondents consider the health and non-use values of the wetlands to be more important than the use values. The ranking of the combined statements from highest to lowest is health, non-use value and use value except for Case B where non-use values are ranked higher than health. As the interaction between survey type and statement is significant this suggests that the particular information provided in each case did produce a difference in attitudes expressed. Thus, the analysis of respondents attitudes indicate that the most important reasons to protect the wetlands are to preserve the health of the Bay and provide for non-use values such as bequest and existence values. These results are in accord with the Stevens et al., (1995) study which demonstrated that the most important reason to protect wetlands was for wildlife habitat and biodiversity, followed by flood protection and then hunting and recreational values.

The results from the attitudinal questions demonstrate that respondents have a difference in their attitude toward wetland protection depending on the economic value being assessed as proposed in Chapter 7. It also appears from Table 9.10 that the information provided in the different cases did influence the attitudes expressed. The results also demonstrate respondents’ preferences for a healthy ecosystem and the non-use values of the rights of both future generations and animals over human use values. From these results it could be concluded that respondents understand the importance of the underlying ecological values and the importance of species dependence and diversity in the wetlands and that these are a necessary prerequisite for use values. Thus, it would be expected that this would be reflected in the mean
willingness to pay for the four survey versions. This issue of whether the attitudinal rankings of the importance of protecting wetlands in terms of ecological or economic value is the same of the value derived from willingness to pay is considered further in Section 9.2.2.

9.2.2 Willingness to Pay Analysis

The iterative bidding technique developed by Davis in 1964 (Mitchell and Carson, 1989) was used to elicit willingness to pay for improved water quality in the Bay that would lead to increased protection of the wetlands. The bid amounts offered to respondents were derived from the first quartile, ($10) median ($20) and third quartile ($50) values obtained from an open-ended question in the pilot survey. Respondents were initially asked if they would be willing to pay $20. If they answered yes they were then asked if they would pay $50. If they were unwilling to pay $20 they were asked if they would pay $10. All respondents were then asked for the highest amount that they would be prepared to pay.

A positive willingness to pay bid was offered by 55.6% of respondents. From hypothesis 1 and 2 in Chapter 7, the questions of interest are if the provision of different types of information produced a differing mean willingness to pay bid and if the mean willingness to pay for ecological value exceeds mean willingness to pay for other components of economic value. The numbers of respondents, mean values and standard errors for each cases (A, B, C and D) of the survey are presented in Table 9.11.

<table>
<thead>
<tr>
<th>Case</th>
<th>Number</th>
<th>Mean</th>
<th>Std error</th>
</tr>
</thead>
<tbody>
<tr>
<td>No information – Case A</td>
<td>217</td>
<td>13.05</td>
<td>1.39</td>
</tr>
<tr>
<td>Ecological information – Case B</td>
<td>196</td>
<td>14.79</td>
<td>1.98</td>
</tr>
<tr>
<td>Use value information – Case C</td>
<td>217</td>
<td>19.22</td>
<td>2.97</td>
</tr>
<tr>
<td>Non-use value information - Case D</td>
<td>217</td>
<td>11.41</td>
<td>1.26</td>
</tr>
<tr>
<td>ALL CASES</td>
<td>847</td>
<td>14.73</td>
<td>1.01</td>
</tr>
</tbody>
</table>

See Chapter 8 for further details of the pilot study.
In answer to the first hypothesis presented in Chapter 7, the provision of information did have an effect on mean willingness to pay with respect to use values and no information or non-use values. An analysis of variance indicated that there was a significant difference between the survey versions (F=2.724, p<0.05)\textsuperscript{11}. Post-hoc LSD tests indicate that use information was significantly different and higher than no information (p<0.05) and non-use information (p<0.01). No information and non-use information were not significantly different from each other. The value for ecological information was not significantly different from any other value. In relation to the hypothesis that information has an effect on willingness to pay, the results shows that there was an information effect.

However, with respect to the second hypothesis presented in Chapter 7, as ecological information was not significantly different (p>0.05) from any other type of information, it may be concluded that respondents did understand that this value was as important as all other values. That is, the connection between the underlying ecosystem value and the ultimate services provided is recognised. However, the conclusion that respondents would provide a higher willingness to pay bid for the survey version that included ecological information cannot be made. Thus, it cannot be stated that respondents considered that ecological values occur prior to the provision of secondary economic value. The results can only demonstrate that willingness to pay for ecological value is not different from other values. This may imply that ecological information was considered to be just another component of economic value.

The provision of use value information produced a higher willingness to pay compared to no information. These findings are similar to the results of Bergstrom \textit{et al.}, (1990), Edwards-Jones \textit{et al.}, (1995), Hoevenagel and van der Linden (1993) and Hanley and Munro (1992). Bergstrom’s \textit{et al.}, (1990) study, found that where respondents were only supplied with service information (similar to the use information provided in this study) they increased their willingness to pay for wetland protection. However, in this study there was no significant difference

\textsuperscript{10}Interestingly, three respondents sent the money with their surveys which indicates that the scenario was considered realistic.
between the provision of no information and information about ecological values. Nor was there any difference between no information and non-use information. That is, the information effect described by Bergstrom et al., (1990) is only evident in the case of use information. It does not hold when extra information is presented about other aspects of ecological and economic value. The results from this study are comparable to the study undertaken by Edwards-Jones et al., (1995) which found that respondents valued the secondary outputs of the environment, such as landscape but they did not value the underlying ecological values defined by the authors.

The results from this study also contrast with results from Hoevenagel and van der Linden (1993) who showed an increase in willingness to pay between those respondents who were presented with a baseline description of an ecological good and those who also received information about four ecological issues. The provision of extra information about ecological and non-use value, in this study, did not result in a significantly higher willingness to pay (p>0.05). It would seem from this result that respondents are more concerned about those aspects of the environment that provide them with direct and indirect benefits as described in the survey than they are with the underlying ecological attributes of the system or the non-use values of endangered species. That is, in the case of ecological and non-use information an ‘information effect’ could not be discerned. From this it could be concluded that it is the type of information that is presented to respondents rather than the provision of extra information that alters willingness to pay bids.

With reference to hypothesis 5, in Chapter 7 the rankings of the mean willingness to pay bids compared to the attitudinal rankings of importance of protecting wetlands in terms of value they provide are clearly not the same. As discussed in Section 9.2.1, attitudinal responses suggested that health and non-use values were more important to respondents than use values. However, the willingness to pay results implied that use values are the most important. This is illustrated in Figure 9.1.

11 However, this result needs to be treated with caution as neither the assumption of normality of homogeneity of variance could be met.
This juxtaposition of use value is of particular concern from an economic theory perspective. This result may be interpreted by considering that respondents act as citizens when placing importance on the values of wetlands but when asked to pay for those values act as consumers. For the consumer, those values that provide the greatest utility will produce the highest mean willingness to pay. Given the high numbers of respondents who already receive benefits from direct use of the Bay (through visits, fishing etc) it is not surprising that use value is ranked highest.\endnote{12}

Sagoff (1988), Stevens et al., (1991), Spash and Hanley (1995), Blamey et al., (1995, 1996), Rolfe and Bennett (1996b), Gowdy (1997) and Nyborg (2000) have examined the distinction between respondents acting as citizen or consumer. For example, Sagoff (1988, 6) argues that citizen choices should be seen as separate from and ethically superior to consumer choices and that environmental problems are “primarily moral, aesthetic, cultural and political”. Similarly, Blamey et al., (1995) argue that respondents to contingent valuation surveys act primarily as citizens as studies are increasingly placed in a political setting with the use of the referendum format. Further, Blamey et al., (1995, 285) note that the more realistic the problem is to the respondents the greater the chance that “citizen responses will be elicited in place of, or in addition to, consumer responses”. It is therefore possible that

\endnote{12} To determine if this high level of use or knowledge of the Bay altered the mean willingness to pay amount ANOVA’s were conducted including only those cases where respondents had/had not undertaken the activities listed, lived/didn’t live at Bay, had/had not read, heard or seen information about the Bay and those who were/were not members of an environmental group. There was a significant difference between the mean willingness to pay for the group who didn’t go fishing (n=551), those who had not engaged in other activities (n=622) and the group who had not heard of the Healthy Waterways Campaign (n=452). However, the rankings of the mean willingness to pay for the each case was the same as for the entire sample.
respondents to my survey may have been acting as citizens when reporting their attitudes about wetland protection.

The disparity in the ranking of use values may be a result of respondents acting as both citizen and consumer during the survey as suggested by Blamey et al., (1995). Respondents expressed moral (or citizen) attitudes with respect to the non-use values of the rights of future humans and species existence in their ranking of the statements presented in the earlier attitudinal questions. This may indicate that importance was placed on those issues important for collective society. However, when asked to pay to preserve the wetlands respondents answered as consumers placing greater value on the survey version which provided them with use value and hence greater individual utility. In their meta-analysis of contingent valuation wetland studies Brouwer et al., (1997) inferred the behavioural motivation of respondents to be both consumer and citizen based on the payment vehicle.

Brouwer et al., (1997) note that the use of voluntary trust funds as a payment vehicle are more likely to induce a positive consumer response than the use of a tax which might be viewed from a citizen perspective. Further evidence for the view that respondents held first citizen and then consumer views comes from the examination of the question following willingness to pay. If the citizen or social view extended to the willingness to pay question it would be expected that those respondents who held moral values about the rights of species would not have been willing to trade money for these rights. That is, if they continued to act as citizens they would hold a lexicographic position. In which case their reason for not offering a positive amount would have been that ‘I did not want to place a dollar value on protecting wetlands’. However, as shown in Table 9.7 only 6% of respondents gave this as a reason for not offering a positive amount.

This result perhaps lends credence to Gowdy’s (1997, 32) claim that “private decisions about available market choices cannot capture collective choices”. In other words, respondents may hold citizen views relating to preservation of a public good but these views will not be disclosed if they are asked to act as private consumers and provide a willingness to pay bid. This dichotomy may then lead to further questions as to the role of contingent valuation in a cost benefit situation for public
goods. However, for the present purposes the dichotomy offers a possible explanation of the survey results.

### 9.3 Regression Analysis

A simple comparison of the means of willingness to pay for the four cases does not allow potentially confounding variables such as experience or socioeconomic variables to be held constant. Thus, in order to improve the reliability of the result a multivariate regression analysis was undertaken. The purpose of the regression was to test if there was a significant difference in the provision of differing types of information and if willingness to pay for ecological value was significantly higher than for the other cases. Regression analysis also provides further information about the independent variables, which influenced willingness to pay bids. A Tobit regression analysis was undertaken to account for both zero bids and positive willingness to pay bids. Halstead *et al.*, (1991) show that the use of ordinary least squares would yield biased and inconsistent estimators where the dependent variable is limited to zero for those unwilling to offer a positive amount. A double bounded dichotomous choice model utilising the results of the first two rounds of bidding was also considered. However, this may have raised problems in relation to starting point bias.

#### 9.3.1 Theoretical Framework of the Tobit Model

McDonald and Moffitt (1980) show that the Tobit coefficients referred to Figure 9.2 can be decomposed to determine the effect of a change in the *i*th variable on changes in the probability of making a positive bid and the expected percentage change in willingness to pay amounts of those who have offered a positive bid. Thus the Tobit method provides closer examination of the effects of independent variables on the current non-zero willingness to pay and on the estimated proportion of zero bidders who would make a positive bid based on changes in independent variables. Following McDonald and Moffitt (1980) the stochastic model underlying Tobit may be expressed by the relationships provided in Figure 9.2.
Figure 9.2 The Tobit Model

\[ Y = X\beta + u_t \quad \text{if} \quad X\beta + u_t > 0 \]  
\[ = 0 \quad \text{if} \quad X\beta + u_t \leq 0 \]  

(1)

The estimates of the Tobit model are derived from

\[ E(Y_i) = X_i\beta F(z) + \sigma f(z) \]  

(2)

And the expected willingness to pay conditional on a positive bid is given by

\[ E(Y^*_i) = X_i\beta + \frac{\sigma f(z)}{F(z)} \]  

(3)

Where:

- \( X \) = a vector of regressor variables
- \( \beta \) = a vector of unknown coefficients (Tobit coefficients)
- \( e \) = a vector of independent and identically distributed normal random variables assumed to have mean zero, and constant variance \( \sigma^2 \)
- \( E(Y^*) = E(Y \mid Y > 0) \)
- \( z = X\beta/\sigma \), normalised index
- \( f(z) = \) the standard normal density function
- \( F(z) = \) the cumulative standard normal distribution function

The basic relationship between the expected value of all observations, \( E(Y_i) \), the expected value conditional upon being above the limit, \( E(Y^*_i) \) and the probability of being above the limit, \( F(z) \) is

\[ E(Y) = F(z)E(Y^*) \]  

(4)

The model can then be decomposed by considering the effect of a change in the \( i \)th variable of \( X \) on \( Y \):

\[ \frac{\partial E(Y_i)}{\partial X_i} = F(z)[\frac{\partial E(Y^*_i)}{\partial X_i} + E(Y^*_i)]\frac{\partial F(z)}{\partial X_i} = F(z)\beta \]  

(5)

Where the marginal effect of a change of the \( i \)th variable on expected willingness to pay conditional on a positive bid is given by:

\[ \frac{\partial E(Y^*_i)}{\partial X_i} = \beta[1-zf(z)/F(z) - f(z)^2/F(z)^2] \]  

(6)

The marginal effect of a change in the \( i \)th explanatory variable on the probability of observing a positive bid is given by

\[ \frac{\partial F(z)}{\partial X_i} = f(z)\beta/\sigma \]  

(7)

As noted by Adesina and Baidu-Forson (1995, 3) equation (5) demonstrates the disaggregation of the model to determine the effect of a change in $i$th variable on changes in the probability of making a non-zero willingness to pay bid $(F(z)[\partial E(Y_i^*)/\partial X_i])$ and the marginal effect of a change in the $i$th explanatory variable on the probability of observing a positive bid $(E(Y_i^*)[\partial F(z)/\partial X_i])$. If (5) is multiplied by $X_i/E(Y_i)$ on both sides, the relation can be converted into elasticity forms.

$$[\partial E(Y_i)/\partial X_i]X_i/E(Y_i) = F(z)[\partial E(Y_i^*)/\partial X_i] X_i/E(Y_i) + E(Y_i^*)[\partial F(z)/\partial X_i] X_i/E(Y_i) \quad (8)$$

Re-arranging given (4) gives:

$$[\partial E(Y_i)/\partial X_i]X_i/E(Y_i) = [\partial E(Y_i^*)/\partial X_i]X_i/E(Y_i^*) + [\partial F(z)/\partial X_i]X_i/F(z) \quad (9)$$

(Adesina and Baidu-Forson, 1995, 3).

Following Norris and Batie’s (1987) explanation of (9) the total elasticity consists of two effects: (1) the elasticity of the conditional expected willingness to pay for those who have a positive willingness to pay amount and (2) the change in the elasticity of the probability of making a positive bid. The two elasticities sum to equal the total elasticity or the percent change in the dependent variable given a one percent change in the independent variable (Norris and Batie, 1987).

Tobin (1958) and Amemiya (1973) show that consistent estimates of $\beta$ and $\sigma$ can be obtained using maximum likelihood techniques, where plim (b) = $\beta$ and plim (s) = $\sigma$. The idea of maximum likelihood estimation is to choose, as estimates of $\beta$, the values of $\beta$ that maximize the probability of obtaining the sample that is actually observed. Computer software uses numerical optimization methods to find values of $\beta$ that maximize the likelihood function or more generally the natural logarithm of the likelihood function. Greene (2000, 911) states that the log-likelihood for the tobit is:

$$\text{LogL} = \sum_{y_i} \frac{-1}{2} \left[ \log(2\pi) + \log\sigma^2 + \left( \frac{y_i - \beta'x_i}{\sigma^2} \right)^2 \right] + \sum_{y_i=0} \log \left[ 1 - \Phi \left( \frac{\beta'x_i}{\sigma} \right) \right].$$
The two parts correspond to the regression for the non-limit observations and the relevant probabilities for the limit observations respectively (Greene, 2000). A computer package (Shazam) which uses the Newton’s method for estimation was used for all tobit regressions. The log-likelihood function is reported for all regressions.

From equation 6 it should be noted that the effect of a change in $X_i$ on $Y^*$ is not equal to $\beta_i$. Thus the estimated coefficients must be adjusted using the following formula:

$$\text{Adjusted coefficient} = (\sigma) \times (F(z)) \times (\text{normalised coefficients})$$  \hspace{1cm} (10)

(Halstead et al., 1991).

Both the normalised and the adjusted (regression) coefficients are reported.

### 9.3.2 Protest bids and willingness to pay

Halstead et al., (1991) warn that it is essential to cull all protest bids for the decomposition effects to hold. That is, the presence of protest bids may introduce selection biases into the analysis by biasing sample mean willingness to pay towards zero (Goodwin et al., 1993). Also, if a respondent is protesting about some aspect of the survey a change in the independent variables will not result in a change in willingness to pay. When all bids are included 44.4% of respondents gave a non-positive willingness to pay bid. Following the removal of protest bids 14.2% of the sample gave a zero willingness to pay bid. The remainder of the dependent variables are continuously distributed. The explanatory variables thus influence both the probability of paying or not paying and the willingness to pay amounts given.

As noted in Section 9.1.4 the options ‘the wetlands are worth nothing to me’ and ‘I couldn’t afford to pay’ were considered the only true zero bids and all other options were considered to be protest bids. All zero bids, which indicated a protest option, were removed from the data set for the regression analysis. Following the removal
of the protest bids the number of responses for each case and the mean willingness to pay is given in Table 9.12\textsuperscript{13}.

<table>
<thead>
<tr>
<th>Case</th>
<th>All Cases</th>
<th>Protest bids removed</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Number</td>
<td>Zero bids %</td>
</tr>
<tr>
<td>No information</td>
<td>217</td>
<td>43.8</td>
</tr>
<tr>
<td>Ecological information</td>
<td>196</td>
<td>42.4</td>
</tr>
<tr>
<td>Use value information</td>
<td>217</td>
<td>39.6</td>
</tr>
<tr>
<td>Non-use value information</td>
<td>217</td>
<td>51.6</td>
</tr>
<tr>
<td>All Cases</td>
<td>849</td>
<td>44.3</td>
</tr>
</tbody>
</table>

The percentage of zero bids ranged from 39.6% for Case C to 51.6% for Case D while mean willingness to pay was highest for Case C and lowest for Case D. This may indicate that respondents were less adverse to paying to protect the wetlands when they consider the possible personal uses of the Bay compared to considering the existence of the endangered species that are dependent on the wetlands. The remaining number of zero bids for the responses ‘the wetlands are worth nothing to me’ and ‘I couldn’t afford to pay’ for each case are presented in Table 9.13. There were also 18 zero bids for which no protest response was received. These zero bids have also been included in the analysis as there was no means of determining if they were protest bids or true zero bids. These bids are presented under the category of other zero bids in Table 9.13.

\textsuperscript{13} Analysis of variance with all zero bids removed and with protest bids removed indicated that there was no significant difference between the differing cases (F=1.46, p>0.05 and F=2.079 and p>0.05 respectively).
Table 9.13  Remaining zero bids – all cases

<table>
<thead>
<tr>
<th>Survey</th>
<th>Wetlands worth nothing</th>
<th>Couldn’t afford to pay</th>
<th>Other zero bids</th>
<th>Total Number</th>
<th>Percent of total remaining responses</th>
</tr>
</thead>
<tbody>
<tr>
<td>Case A</td>
<td>1</td>
<td>15</td>
<td>3</td>
<td>19</td>
<td>13.5</td>
</tr>
<tr>
<td>Case B</td>
<td>0</td>
<td>11</td>
<td>3</td>
<td>14</td>
<td>11.0</td>
</tr>
<tr>
<td>Case C</td>
<td>2</td>
<td>10</td>
<td>5</td>
<td>17</td>
<td>11.6</td>
</tr>
<tr>
<td>Case D</td>
<td>1</td>
<td>20</td>
<td>7</td>
<td>28</td>
<td>21.1</td>
</tr>
<tr>
<td>All Cases</td>
<td>4</td>
<td>56</td>
<td>18</td>
<td>78</td>
<td>14.2</td>
</tr>
</tbody>
</table>

After the removal of the protest bids the Case D (non-use information) again had the highest number of zero bids and Case B recorded the lowest number. The distribution of bids for all cases following the removal of protest bids is presented in Table 9.14.

Table 9.14  Frequency of willingness to pay for wetland protection. All cases.

<table>
<thead>
<tr>
<th>Valid $</th>
<th>Frequency</th>
<th>Percent</th>
<th>Valid Percent</th>
<th>Cumulative Percent</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>78</td>
<td>14.2</td>
<td>14.2</td>
<td>14.2</td>
</tr>
<tr>
<td>1</td>
<td>1</td>
<td>0.2</td>
<td>0.2</td>
<td>14.4</td>
</tr>
<tr>
<td>2</td>
<td>11</td>
<td>2.0</td>
<td>2.0</td>
<td>16.8</td>
</tr>
<tr>
<td>5</td>
<td>28</td>
<td>5.1</td>
<td>5.1</td>
<td>21.9</td>
</tr>
<tr>
<td>10</td>
<td>125</td>
<td>22.8</td>
<td>22.8</td>
<td>44.7</td>
</tr>
<tr>
<td>15</td>
<td>5</td>
<td>0.9</td>
<td>0.9</td>
<td>45.6</td>
</tr>
<tr>
<td>20</td>
<td>182</td>
<td>33.2</td>
<td>33.2</td>
<td>78.8</td>
</tr>
<tr>
<td>25</td>
<td>14</td>
<td>2.6</td>
<td>2.6</td>
<td>81.4</td>
</tr>
<tr>
<td>30</td>
<td>17</td>
<td>3.1</td>
<td>3.1</td>
<td>84.5</td>
</tr>
<tr>
<td>35</td>
<td>2</td>
<td>0.4</td>
<td>0.4</td>
<td>84.9</td>
</tr>
<tr>
<td>40</td>
<td>7</td>
<td>1.3</td>
<td>1.3</td>
<td>86.1</td>
</tr>
<tr>
<td>50</td>
<td>47</td>
<td>8.6</td>
<td>8.6</td>
<td>94.7</td>
</tr>
<tr>
<td>70</td>
<td>1</td>
<td>0.2</td>
<td>0.2</td>
<td>94.9</td>
</tr>
<tr>
<td>100</td>
<td>22</td>
<td>4.0</td>
<td>4.0</td>
<td>98.9</td>
</tr>
<tr>
<td>150</td>
<td>1</td>
<td>0.2</td>
<td>0.2</td>
<td>99.1</td>
</tr>
<tr>
<td>200</td>
<td>3</td>
<td>0.5</td>
<td>0.5</td>
<td>99.6</td>
</tr>
<tr>
<td>250</td>
<td>1</td>
<td>0.2</td>
<td>0.2</td>
<td>99.8</td>
</tr>
<tr>
<td>500</td>
<td>1</td>
<td>0.2</td>
<td>0.2</td>
<td>100.0</td>
</tr>
<tr>
<td>TOTAL</td>
<td>548</td>
<td>100.0</td>
<td>100.0</td>
<td></td>
</tr>
</tbody>
</table>
It is noted that there is a clustering of bids around the $10, $20 and $50 values. This is not surprising as all respondents were presented with an initial dichotomous $20 bid and the other bids were presented sequentially. The distribution of bids demonstrates that 33.2% of respondents gave a $20 bid which may indicate that they anchored to this amount as it may have been viewed as the cost of providing the program. This may under or over represent the unknown true willingness to pay. However, as noted this amount was derived from the median bid from the open-ended pilot survey. As the purpose was not to determine a value for the wetlands per se but rather compare willingness to pay with respect to information this clustering of bids was not considered a problem as it applied equally to all respondents. Further, similar studies undertaken by Stevens et al., (1995) and Streever et al., (1998) which used a dichotomous choice technique also reported that willingness to pay was related to the initial price given. Thus, using a dichotomous choice format that Green et al., (1998) claim is statistically inefficient and relatively complex to analyse may not have avoided this problem.

Presenting respondents with an initial bid and then subsequent bids also increased the number of respondents who were willing to offer a positive amount (55.7%) in comparison with the open-ended pilot survey (23.3%). Thus, although starting point bias may have occurred through the use of the bidding technique, the more thorough preference searching required produced a positive outcome as evidenced by the higher number of non-zero bids. The remainder of the bids are derived from the final open-ended willingness to pay question.

9.3.3 Factors hypothesised to be relevant to willingness to pay

Factors that may influence willingness to pay bids include respondents attitudes towards wetland protection, their previous knowledge of the Bay as expressed through visits and activities undertaken and awareness of the Bay from media sources. Thus each of these factors needed to be included within the regression analysis.

As noted the term ‘health’ was used as a proxy for ecological value. Given, the interest in the ability of the contingent valuation technique to measure ecological
value, the variable health was included as an independent variable to test the importance of attitudes on willingness to pay. If attitudes influence willingness to pay then it would be expected that the greater the importance placed on maintaining the health of the Bay, the greater would be the willingness to pay for wetland protection.

To determine if knowledge contributed to willingness to pay for wetland protection it was considered that visits should be included as an explanatory variable. It was expected that those who visit the Bay would have a greater understanding and knowledge of the wetlands and that this may influence willingness to pay positively. The variable visit was included as a binary variable. Further testing on the importance of knowledge gained from visiting was tested for those who had and not visited Moreton Bay.

Respondents may also have gained information about the wetlands from the media as detailed in Section 9.1.2. The institutional mechanism for improving water quality was the Waterways Management Plan. As noted in Chapter 6, this strategy has been promoted through the Healthy Waterways campaign. As at least half of the respondents in all surveys had indicated that they heard of the Healthy Waterways campaign the information derived from the program may have contributed to willingness to pay. If the campaign has been successful in making respondents aware of problems and possible solution to water quality issues in the Bay it would be expected that scenario presented was considered to be true (which it was) which might in turn increase willingness to pay. Thus, HW Camp, the variable relating to the Healthy Waterways campaign was included in the model as a binary variable to test further for the effect of previous information.

Previous contingent valuation studies have related the socio-economic characteristics of the respondents to willingness to pay bids (Hanley and Craig, 1991). Information was collected on respondent’s sex, age, education and income. Sex was included as a binary variable. The age of the respondents was calculated at the mean of each age category and included as a continuous variable. The education categories were divided into those who had education up to a secondary level and those who had had post-secondary education. Education was thus also a binary variable. Income was
calculated at the mean of the income categories and included as a continuous variable. A description of the variables included in the model is given in Table 9.15.

Table 9.15 Variables included in model to determine factors influencing willingness to pay for wetland protection.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Dependent Variable</strong></td>
<td>Individual’s willingness to pay for wetland protection</td>
</tr>
<tr>
<td>WTP $</td>
<td></td>
</tr>
<tr>
<td><strong>Independent Variables</strong></td>
<td></td>
</tr>
<tr>
<td>Health</td>
<td>The importance of protecting wetlands to maintain health – Scale 1 = not at all important, 5 = extremely important</td>
</tr>
<tr>
<td>Visit</td>
<td>1 if individual visited wetlands, 0 otherwise</td>
</tr>
<tr>
<td>HW Camp</td>
<td>1 if individual heard of <em>Health Waterways</em> campaign, 0 otherwise</td>
</tr>
<tr>
<td>Age</td>
<td>Age of respondents. Mid-points of the 7 age categories</td>
</tr>
<tr>
<td>Sex</td>
<td>1 if individual a male, 0 if individual a female</td>
</tr>
<tr>
<td>Education</td>
<td>1 if individual has post-secondary education, 0 otherwise</td>
</tr>
<tr>
<td>Income</td>
<td>Income of respondents. Mid-points of the 8 income categories</td>
</tr>
</tbody>
</table>

The descriptive statistics for the independent variables included in the model are given in Table 9.16. The table demonstrates that on average respondents felt that it was very important (4 on the scale) to protect the wetlands in order to help maintain the health of the Bay. Also as noted, 76.3% of the sample had visited the Bay with 85% of respondents who received Case C (use information) visiting the Bay at least once in the past twelve months. The *Healthy Waterways* campaign had provided information about the Bay to at least half of all respondents in all cases. The sex variable indicates that there was a greater proportion of men than women included in the sample but as demonstrated in Table 9.6 this was representative of population from which the sample was drawn. The education category demonstrates that only 35% of the sample had received post secondary education. On average respondents were in the 45-56 age bracket and had average annual incomes below $28 000.
Table 9.16 Descriptive Statistics of individual characteristics used in demand models

<table>
<thead>
<tr>
<th>Independent Variables</th>
<th>All Cases Mean (SE)</th>
<th>Case A Mean (SE)</th>
<th>Case B Mean (SE)</th>
<th>Case C Mean (SE)</th>
<th>Case D Mean (SE)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Health</td>
<td>4.45 (4.31E-02)</td>
<td>4.46 (8.75E-02)</td>
<td>4.41 (8.69E-02)</td>
<td>4.56 (7.55E-02)</td>
<td>4.38 (9.52E-02)</td>
</tr>
<tr>
<td>Visit</td>
<td>0.80 (1.70E-02)</td>
<td>0.77 (3.58E-02)</td>
<td>0.83 (3.37E-02)</td>
<td>0.85 (2.95E-02)</td>
<td>0.77 (3.68E-02)</td>
</tr>
<tr>
<td>HW Camp</td>
<td>0.52 (2.14E-02)</td>
<td>0.51 (4.22E-02)</td>
<td>0.54 (4.44E-02)</td>
<td>0.50 (4.14E-02)</td>
<td>0.52 (4.35E-02)</td>
</tr>
<tr>
<td>Sex</td>
<td>0.41 (2.11E-02)</td>
<td>0.45 (4.20E-02)</td>
<td>0.43 (4.41E-02)</td>
<td>0.37 (4.00E-02)</td>
<td>0.41 (4.27E-02)</td>
</tr>
<tr>
<td>Education</td>
<td>0.32 (2.00E-02)</td>
<td>0.32 (3.94E-02)</td>
<td>0.35 (4.24E-02)</td>
<td>0.31 (3.84E-02)</td>
<td>0.32 (4.07E-02)</td>
</tr>
<tr>
<td>Income</td>
<td>$25972.41 (971.18)</td>
<td>$27425.24 (1980.73)</td>
<td>$26318.11 (2061.10)</td>
<td>$24296.05 (1806.26)</td>
<td>$25954.89 (1947.05)</td>
</tr>
<tr>
<td>Age</td>
<td>47.08 (0.75)</td>
<td>47.37 (1.345)</td>
<td>46.14 (1.6318)</td>
<td>47.42 (1.4547)</td>
<td>47.29 (1.6050)</td>
</tr>
</tbody>
</table>

9.4 RESULTS OF REGRESSION ANALYSIS

An initial test of the model was undertaken using both ordinary least squares (OLS) and Tobit regression. Only the last open-ended bid given in the survey is used in this analysis. To demonstrate the effects of removing the protest bids, the model was also run using ordinary least squares with all bids included. The results for the Tobit model with protest bids removed, the OLS with protest bids removed and OLS for all bids are given in Table 9.17.
Table 9.17  Results – All Cases – OLS and Tobit regression

<table>
<thead>
<tr>
<th>Variable</th>
<th>Least Squares – all bids</th>
<th>Least Squares – Protest deleted</th>
<th>Tobit – Protest deleted</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Estimated Coefficient</td>
<td>T-ratio</td>
<td>Estimated Coefficient</td>
</tr>
<tr>
<td>Health</td>
<td>2.634</td>
<td>2.872***</td>
<td>3.5257</td>
</tr>
<tr>
<td>Visit</td>
<td>5.561</td>
<td>2.368**</td>
<td>8.2862</td>
</tr>
<tr>
<td>HW Camp</td>
<td>2.486</td>
<td>1.266</td>
<td>-0.9568</td>
</tr>
<tr>
<td>Sex</td>
<td>1.117</td>
<td>0.552</td>
<td>5.74</td>
</tr>
<tr>
<td>Age</td>
<td>1.166E-02</td>
<td>0.204</td>
<td>0.99793E-02</td>
</tr>
<tr>
<td>Income</td>
<td>2.953E-04</td>
<td>6.411***</td>
<td>0.35318E-03</td>
</tr>
</tbody>
</table>

**OLS All bids:  R^2 = 0.119  Adjusted R^2 = 0.111**

**OLS Protest bids deleted:  R^2 = 0.1592  Adjusted R^2 = 0.1483**

Tobit:
- Predicted probability of Y>limit given average X(I) = 0.7066
- Observed frequency of Y> limit is = 0.8577
- At mean values of all X(I), E(Y) = 25.0133
- Log-likelihood function = -2382.8857
- Square Correlation between observed and expected values = 0.17243

**significant at 0.05, *** significant at 0.01 level

1 The Tobit coefficients have been adjusted according to (10).

The significant variables for all regressions are health, visit, education and income. The ordinary least squares with protest bids deleted also indicated that sex was significant (p < 0.05). The R^2 value improved with the deletion of the protest bids.

In the Tobit analysis the square correlation between observed and expected values also improved. Stone (1992) notes that low regression values are typical of surveys of the general population compared to homogenous groups such as on-site users. It could be argued that this is a result of individualised utility functions, which determine values for public goods.

Similar results have been found in other contingent value studies that have assessed information effects. For example, Edwards-Jones et al., (1995) found an R^2 of 0.121, Hoevenagel and van der Linden (1993) produced an R^2 of 0.22, Whitehead and Blomquist (1991) found R^2’s between 0.12 and 0.14 using ordinary least squares.
Stevens et al., (1991) report a square correlation of 0.19 using Tobit regression. Thus, the square correlation value of 0.17 is typical of other general public contingent valuation surveys that have tested for information effects. The value does exceed the standard of 15% which Mitchell and Carson (1989) consider the minimum for a reliable contingent valuation survey using ordinary least squares.

Tobit regressions were then run for each of the four cases using the variables health, visit, HW Camp, age, sex, education and income. The results for each case are presented in Tables 9.18 – 9.21 respectively.

Table 9.18  Tobit regression results Case A.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Normalized Coefficient</th>
<th>Asymptotic</th>
<th>Regression coefficient</th>
<th>Elasticities</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Standard error</td>
<td>T-ratio</td>
<td>Prob. +ive bid</td>
</tr>
<tr>
<td>Health</td>
<td>0.38487</td>
<td>0.10942</td>
<td>3.5172***</td>
<td>8.3700</td>
</tr>
<tr>
<td>Visit</td>
<td>0.31270</td>
<td>0.21841</td>
<td>1.431</td>
<td>6.8004</td>
</tr>
<tr>
<td>HW Camp</td>
<td>0.31589</td>
<td>0.17832</td>
<td>1.7174*</td>
<td>6.8698</td>
</tr>
<tr>
<td>Age</td>
<td>0.78122E-03</td>
<td>0.59941E-02</td>
<td>0.13033</td>
<td>0.1699E-01</td>
</tr>
<tr>
<td>Sex</td>
<td>-0.42641E-01</td>
<td>0.19397</td>
<td>-0.21984</td>
<td>-0.92734</td>
</tr>
<tr>
<td>Education</td>
<td>0.26383</td>
<td>0.19789</td>
<td>1.3333</td>
<td>5.7377</td>
</tr>
<tr>
<td>Income</td>
<td>0.13456E-04</td>
<td>0.4189E-05</td>
<td>3.2117***</td>
<td>0.29263E-03</td>
</tr>
<tr>
<td>Constant</td>
<td>-1.7540</td>
<td>0.54645</td>
<td>-3.2098***</td>
<td>-38.145</td>
</tr>
</tbody>
</table>

The predicted probability of \( Y > \)limit given average \( X(I) = 0.7981 \)

The observed frequency of \( Y > \) limit is = 0.8652

At mean values of all \( X(I) \), \( E(Y) = 20.6146 \)

Log-likelihood function = -561.18120

Mean-square error = 392.09196

Mean error = 1.2343830

Squared correlation between observed and expected values = 0.20733

* significant at 0.1 level, ** significant at 0.05 level, *** significant at 0.01 level

For Case A (no information) the variables health and income were significant (p<0.05) in explaining willingness to pay and the healthy waterways campaign was significant (p<0.1). A marginal increase in the importance of maintaining health increases willingness to pay by $8.37. A 10% increase in income has a less pronounced effect raising willingness to pay by $0.0029. The last two columns of
Table 9.18 present the elasticities of the decomposed model as discussed in (9). Following Norris and Batie (1987) the elasticities can be interpreted as follows using the health variable as an example. If the proportion of respondents who viewed health at a category higher than given increased by 10% then the overall willingness to pay would increase by 32.5% from that given. Of that total, 18% would be a result of an increase in the number of positive bids and 14.5% would be an increase in willingness to pay for those with a positive bid. A 1% increase in mean income would increase the probability of a positive bid by 0.38%. For those with a positive bid the expected increase in willingness to pay would be 0.30%. For Case A, the variable visit is not significant which suggests that knowledge gained from visiting the Bay did not influence willingness to pay amounts. It appears that for this case the most important factors influencing willingness to pay were respondents attitudes towards wetland protection and their income.
Table 9.19  Tobit regression results Case B

<table>
<thead>
<tr>
<th></th>
<th>Normalized Coefficient</th>
<th>Asymptotic Standard error</th>
<th>Asymptotic T-ratio</th>
<th>Regression coefficient</th>
<th>Elasticities</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Health</td>
<td>0.16068</td>
<td>0.11321</td>
<td>1.4192</td>
<td>5.0745</td>
<td>0.9801</td>
</tr>
<tr>
<td>Visit</td>
<td>0.36821</td>
<td>0.24999</td>
<td>1.4729</td>
<td>11.629</td>
<td>0.4211</td>
</tr>
<tr>
<td>HW Camp</td>
<td>-0.41604E-01</td>
<td>0.19109</td>
<td>-0.21771</td>
<td>-1.3139</td>
<td>-0.0308</td>
</tr>
<tr>
<td>Age</td>
<td>-0.75694E-02</td>
<td>0.52177E-02</td>
<td>-1.4507</td>
<td>-0.23906</td>
<td>-0.0483</td>
</tr>
<tr>
<td>Sex</td>
<td>0.34856</td>
<td>0.20907</td>
<td>1.6672*</td>
<td>11.008</td>
<td>0.2088</td>
</tr>
<tr>
<td>Education</td>
<td>0.41060</td>
<td>0.21245</td>
<td>1.9327*</td>
<td>12.968</td>
<td>0.1968</td>
</tr>
<tr>
<td>Income</td>
<td>0.65502E-05</td>
<td>0.47686E-05</td>
<td>1.3736</td>
<td>0.20687E-03</td>
<td>0.2385</td>
</tr>
<tr>
<td>Constant</td>
<td>-0.47178</td>
<td>0.53233</td>
<td>-0.88625</td>
<td>-14.9006</td>
<td>0.1600</td>
</tr>
</tbody>
</table>

The predicted probability of Y>limit given average X(I) = 0.7373
The observed frequency of Y> limit is = 0.8898
At mean values of all X(I), E(Y) = 25.0894
Log-likelihood function = -560.51376
Mean-square error = 870.19376
Mean error = 3.4024814
Squared correlation between observed and expected values = 0.13988
*significant at 0.1 level, ** significant at 0.05 level, *** significant at 0.01 level

In Case B (ecological information) sex and education were significant at the 0.1 level. Females were on average willing to offer an extra $11.00 than males to protect the wetlands. The elasticity of the probability of making a positive bid if female is 0.19%. Moving from secondary education or under to post secondary education increases willingness to pay by $12.97. The elasticities of the probability of making a positive bid moving from the lower to the higher education level is 0.2385% and the elasticities of an increase in willingness to pay (given a positive bid) rises by 0.16%. Neither attitudes, nor visits to the Bay had a significant impact as in the overall model. In an effort to improve the model’s performance a number of other independent variables were tested. However, it appears that the respondents’ sex (if female) and a post-secondary level of education were the most important factors in willingness to pay given ecological information. Education may have been significant as a higher level of understanding is required to unravel the complexity of the notion of ecological value.
Table 9.20  Tobit regression results Case C

<table>
<thead>
<tr>
<th></th>
<th>Normalized Coefficient</th>
<th>Asymptotic Standard error</th>
<th>Asymptotic T-ratio</th>
<th>Regression coefficient</th>
<th>Elasticities</th>
<th>Prob. +ive bid</th>
<th>Inc. WTP (given +ive bid)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Health</td>
<td>0.22190</td>
<td>0.10673</td>
<td>2.0791**</td>
<td>10.380</td>
<td>1.6750</td>
<td>1.0013</td>
<td></td>
</tr>
<tr>
<td>Visit</td>
<td>0.18635</td>
<td>0.25748</td>
<td>0.72377</td>
<td>8.7171</td>
<td>0.2624</td>
<td>0.1569</td>
<td></td>
</tr>
<tr>
<td>HW Camp</td>
<td>-0.44259E-01</td>
<td>0.17478</td>
<td>-0.25322</td>
<td>-2.0703</td>
<td>-0.0369</td>
<td>-0.0221</td>
<td></td>
</tr>
<tr>
<td>Age</td>
<td>0.52312E-02</td>
<td>0.52087E-02</td>
<td>1.0043</td>
<td>0.24470</td>
<td>0.4108</td>
<td>0.2456</td>
<td></td>
</tr>
<tr>
<td>Sex</td>
<td>0.27231</td>
<td>0.18433</td>
<td>1.4773</td>
<td>12.738</td>
<td>0.1687</td>
<td>0.1009</td>
<td></td>
</tr>
<tr>
<td>Education</td>
<td>0.50684</td>
<td>0.19702</td>
<td>2.5725***</td>
<td>23.709</td>
<td>0.2627</td>
<td>0.1570</td>
<td></td>
</tr>
<tr>
<td>Income</td>
<td>0.21400E-04</td>
<td>0.42411-05</td>
<td>5.0460***</td>
<td>0.10011E-02</td>
<td>0.8611</td>
<td>0.5148</td>
<td></td>
</tr>
<tr>
<td>Constant</td>
<td>-1.6742</td>
<td>0.55626</td>
<td>-3.0098***</td>
<td>-78.315</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

The predicted probability of Y> limit given average X(I) = 0.6921
The observed frequency of Y> limit is = 0.8844
At mean values of all X(I), E(Y) = 32.7007
Log-likelihood function = -695.20764
Mean-square error = 1903.5248
Mean error = 7.5118189
Squared correlation between observed and expected values = 0.29207
* significant at 0.1 level, ** significant at 0.05 level, *** significant at 0.01 level

The significant variables in Case C (use information) were health, education and income (p < 0.05). A marginal increase in importance for protecting wetlands increased willingness to pay by $10.36. If the proportion of respondents who viewed health at a category higher than given increased by 10% then the overall willingness to pay would increase by 26.76% from that given. Of that total, 16.75% would be a result of an increase in the number of positive bids and 10.01% would be an increase in willingness to pay for those with a positive bid. The education coefficient indicates that moving from secondary or lower education to post-secondary education increases willingness to pay by $23.70. Again, the influence of income, although significant (p < 0.01), is less pronounced with willingness to pay increasing by $0.009 as income is increased by 10%.
Table 9.21 Tobit regression results Case D

<table>
<thead>
<tr>
<th>Variable</th>
<th>Normalized Coefficient</th>
<th>Asymptotic Standard error</th>
<th>T-ratio</th>
<th>Regression coefficient</th>
<th>Elasticities Prob. +ive bid</th>
<th>Inc. WTP (given +ive bid)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Health</td>
<td>0.23416</td>
<td>0.97049E-01</td>
<td>2.4128**</td>
<td>4.8760</td>
<td>1.1452</td>
<td>0.9073</td>
</tr>
<tr>
<td>Visit</td>
<td>0.41414</td>
<td>0.23206</td>
<td>1.7846*</td>
<td>8.6237</td>
<td>0.3550</td>
<td>0.2812</td>
</tr>
<tr>
<td>HW Camp</td>
<td>0.16955</td>
<td>0.19722</td>
<td>0.85971</td>
<td>3.5306</td>
<td>0.0983</td>
<td>0.0779</td>
</tr>
<tr>
<td>Age</td>
<td>-0.60613E-02</td>
<td>0.52567E-02</td>
<td>-1.1531</td>
<td>-0.12622</td>
<td>-0.3204</td>
<td>-0.2538</td>
</tr>
<tr>
<td>Sex</td>
<td>0.33782</td>
<td>0.20391</td>
<td>1.6626*</td>
<td>7.0344</td>
<td>0.1533</td>
<td>0.1215</td>
</tr>
<tr>
<td>Education</td>
<td>0.73039</td>
<td>0.21415</td>
<td>3.4106***</td>
<td>15.209</td>
<td>0.2639</td>
<td>0.2091</td>
</tr>
<tr>
<td>Income</td>
<td>0.11051E-04</td>
<td>0.44905-05</td>
<td>2.4601**</td>
<td>0.23013E-03</td>
<td>0.3206</td>
<td>0.2540</td>
</tr>
<tr>
<td>Constant</td>
<td>-1.0719</td>
<td>0.52157</td>
<td>-2.0551**</td>
<td>-22.321</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

The predicted probability of Y>limit given average X(I) = 0.7679
The observed frequency of Y> limit is = 0.7895
At mean values of all X(I), E(Y) = 18.0570
Log-likelihood function = -489.47439
Mean-square error = 310.08924
Mean error = 0.83538
Squared correlation between observed and expected values = 0.27847
* significant at 0.1 level, ** significant at 0.05 level, *** significant at 0.01 level

The results for Case D were similar to Case C with health, education and income all being significant variables (p < 0.05). Visit and sex were also significant at the 0.1 level. A marginal increase in the importance of maintaining the health of the Bay increases willingness to pay by $4.88 and moving from the lower to higher education category increases willingness to pay by $15.21. For the income categories a 10% increase in income increases willingness to pay by only $0.002.

The results of the Tobit regression for each case demonstrate that respondent’s attitudes toward the protection of the wetlands in order to maintain health was a significant determinant of willingness to pay in Cases A, C and D. For Cases A, C and D a marginal increase in importance increases willingness to pay between $4.87 and $10.35. That is, for Cases A, C and D attitudes towards wetland protection is an important factor in willingness to pay. Education was significant in all models.
except Case A. If the education category of the respondent increases from secondary to post-secondary education the increase in willingness to pay is between $1.94 - $23.71. Income was significant for all cases except Case B. For Cases A, C and D an increase in income of 10% increases willingness to pay between $0.0023 and $0.01.

Unlike the overall model, the variable visit was only significant at the 0.1 level for Case D. Thus, for the Cases A, B and C no conjecture can be made about the influence of knowledge gained from visiting the Bay. The influence of information gained from the Healthy Waterways campaign did not have a significant effect on willingness to pay except for Case A at the 0.1 level. This may indicate that the information being provided in the campaign has not influenced the attitudes of respondents. The other forms of media (paper, TV and radio) which may have contributed to information were then tested in the individual models. However, none of the variables was found to be significant.

Case C and Case D had the same attitudinal variables explaining willingness to pay, however, the analysis of variance (with protest bids included) demonstrates that mean willingness to pay for the two cases was different. This leads to conjecture as to the influence of the information presented for the two cases. To test the effect on the model of the different types of information a Tobit analysis was constructed using dummy variables for each of the cases. The use of dummy variables indicate the difference in the intercept between the variable which is excluded (the reference case) and the other dummy variables used in the model.

For example, if one considers the difference between Case A (the reference case) and Case B so that Case A = 0 and Case B = 1. Using the model $Y = \beta_0 + \beta_1 \text{CaseB}$, then $E(Y|\text{Case A}=0) = \beta_0$ and $E(Y|\text{CaseB} = 1) = \beta_0 + \beta_1 \text{CaseB}$. The coefficient $\beta_1$ attached to dummy variable Case B is the differential intercept coefficient as it tells by how much the value of the intercept term differs from the intercept coefficient for the reference case, Case A. Table 9.22 presents the results when Case A was taken as the reference case and Cases B, C and D were each equal to 1.
Table 9.22  Results of Tobit regression including cases as dummy variables

<table>
<thead>
<tr>
<th></th>
<th>Normalized Coefficient</th>
<th>Asymptotic</th>
<th>Regression coefficient</th>
<th>Elastities</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Standard error</td>
<td>T-ratio</td>
<td></td>
</tr>
<tr>
<td>Case B</td>
<td>0.81642E-01</td>
<td>0.12588</td>
<td>0.64857</td>
<td>2.7832</td>
</tr>
<tr>
<td>Case C</td>
<td>0.26207</td>
<td>0.12176</td>
<td>2.1524**</td>
<td>8.9342</td>
</tr>
<tr>
<td>Case D</td>
<td>-0.77273E-01</td>
<td>0.12549</td>
<td>-0.61576</td>
<td>-2.6343</td>
</tr>
<tr>
<td>Health</td>
<td>0.21546</td>
<td>0.51848E-01</td>
<td>4.1555***</td>
<td>7.3450</td>
</tr>
<tr>
<td>Visit</td>
<td>0.33071</td>
<td>0.11728</td>
<td>2.8198***</td>
<td>11.274</td>
</tr>
<tr>
<td>HW Camp</td>
<td>0.20367E-01</td>
<td>0.90817E-01</td>
<td>0.22426</td>
<td>0.69430</td>
</tr>
<tr>
<td>Age</td>
<td>-0.19234E-02</td>
<td>0.26658E-02</td>
<td>-0.72152</td>
<td>-0.65569E-01</td>
</tr>
<tr>
<td>Sex</td>
<td>0.17884</td>
<td>0.95646E-01</td>
<td>1.8698*</td>
<td>6.0967</td>
</tr>
<tr>
<td>Education</td>
<td>0.38986</td>
<td>0.10030</td>
<td>3.8870***</td>
<td>13.290</td>
</tr>
<tr>
<td>Income</td>
<td>0.13236E-04</td>
<td>0.2164E-05</td>
<td>6.1159***</td>
<td>0.45123E-03</td>
</tr>
<tr>
<td>Constant</td>
<td>-1.2128</td>
<td>0.27561</td>
<td>-4.4003***</td>
<td>-41.343</td>
</tr>
</tbody>
</table>

The predicted probability of Y>limit given average X(I) = 0.7080
The observed frequency of Y> limit is = 0.8577
At mean values of all X(I), E(Y) = 24.9193
Log-likelihood function = -2378.7020
Mean-square error = 965.05414
Mean error = 3.8852445
Squared correlation between observed and expected values = 0.18762
* significant at 0.1 level, ** significant at 0.05 level, *** significant at 0.01 level

As for the overall model the independent variables of health, education and income were significant in explaining willingness to pay. As the Case C coefficient is significantly different from zero the hypothesis that the intercept for Case C is equal to zero can be rejected. Thus, there was a difference in the intercept between Case A and Case C. Therefore, it can be stated that information on use values (Case C) produced a higher willingness to pay (by $8.93) than when no information was provided. However, the hypothesis that the intercepts were not different from zero could not be rejected for Case B and Case D. This may suggest that the provision of use information is a determinant of willingness to pay while the provision of other types of information (relative to no information) are not significant in explaining changes in willingness to pay. Analysis of variance was undertaken on the data set

14 Case A used as reference
with the protest bids deleted. However, a significant difference between survey types was not demonstrated in this instance (p>0.05).

To test further the influence of information provided in the survey, regression analysis was run with Case C as the reference case to determine if there were any further differences between Cases C and Cases B and D. The results are presented in Table 9.23.

Table 9.23 Results of Tobit regression including cases as dummy variables15

<table>
<thead>
<tr>
<th></th>
<th>Normalized Coefficient</th>
<th>Asymptotic Coefficient</th>
<th>Regression coefficient</th>
<th>Elasticities</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Standard error</td>
<td>T-ratio</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Prob. +ive bid</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Inc. WTP (given +ive bid)</td>
</tr>
<tr>
<td>Case A</td>
<td>-0.26207</td>
<td>0.12176</td>
<td>-2.1524**</td>
<td>-8.9342</td>
</tr>
<tr>
<td>Case B</td>
<td>-0.18043</td>
<td>0.12444</td>
<td>-1.4500</td>
<td>-6.1509</td>
</tr>
<tr>
<td>Case D</td>
<td>-0.33935</td>
<td>0.12447</td>
<td>-2.7263***</td>
<td>-11.568</td>
</tr>
<tr>
<td>Health</td>
<td>0.21546</td>
<td>0.51848E-01</td>
<td>4.1555***</td>
<td>7.3450</td>
</tr>
<tr>
<td>Visit</td>
<td>0.33071</td>
<td>0.11728</td>
<td>2.8198***</td>
<td>11.274</td>
</tr>
<tr>
<td>HwCamp</td>
<td>0.20367E-01</td>
<td>0.90817E-01</td>
<td>0.22426</td>
<td>0.694309</td>
</tr>
<tr>
<td>Age</td>
<td>-0.19234E-02</td>
<td>0.26658E-02</td>
<td>-0.72152</td>
<td>-0.65569E-01</td>
</tr>
<tr>
<td>Sex</td>
<td>0.17884</td>
<td>0.95646E-01</td>
<td>1.8698*</td>
<td>6.0967</td>
</tr>
<tr>
<td>Education</td>
<td>0.38986</td>
<td>0.10030</td>
<td>3.8870***</td>
<td>13.290</td>
</tr>
<tr>
<td>Income</td>
<td>0.13236E-04</td>
<td>0.21642E-05</td>
<td>6.1159***</td>
<td>0.45123E-03</td>
</tr>
<tr>
<td>Constant</td>
<td>-0.95069</td>
<td>0.27692</td>
<td>-3.4331***</td>
<td>-32.409</td>
</tr>
</tbody>
</table>

The predicted probability of Y>limit given average X(I) = 0.7080
The observed frequency of Y> limit is = 0.8577
At mean values of all X(I), E(Y) = 24.9139
Log-likelihood function = -2378.7020
Mean-square error = 965.05414
Mean error = 3.8852445
Squared correlation between observed and expected values = 0.18762
* significant at 0.1 level, ** significant at 0.05 level, *** significant at 0.01 level

15 Case C as reference.
With Case C as the reference case there is a significant difference between the
intercepts for Case A and Case C as expected from the results presented in Table
9.22. There is also a significant difference between the intercepts for Case C and
Case D at the 0.01 level. Respondents are willing to offer $11.56 less when provided
with information in Case D than Case C as reflected by the negative coefficient. The
variables health, visit, education and income were all significant in the model.
Examination of Tables thus indicates that the type of information provided to
respondents did have a significant impact on their willingness to pay. There is a
difference between the provision of no information and use information. There is
also a difference between use and non-use information in a negative direction. The
model was also run using Case B and Case D as the reference level, however, there
were no further differences detected.

The regression analysis using the individual cases as dummy variables demonstrates
the information effect detected in the survey. It can be said there was an information
effect moving from no information to use information. That is, there was an
information effect as a result of extra information. The provision of extra
information did not however have an effect when information was provided about
ecological values or non-use values. There was also an information type effect as
can be seen in the difference between the provision of use and non-use information.
This information type effect did not hold for the case of ecological information.

Given that the information effect was most pronounced for use values the importance
of use on willingness to pay was further investigated. First, the sample was divided
into those who had visited the Bay (visitors) and those who had not (non-visitors).
Comparing the means of the two groups revealed that mean willing to pay to protect
the wetlands was higher amongst visitors (mean = $25.15 from 440 non-protest bids
at 95% confidence interval = $5.10 - $19.38) than amongst non-visitors (mean =
$12.91 from 108 non-protest bids at 95% confidence interval = $7.79 - $16.69). To
explore further the relationship between visits as an indication of knowledge and the
willingness to pay for the different cases Tobit regressions were run for those who
had visited and not visited. The results are presented in Tables 9.24 and 9.25
respectively.
Table 9.24 Results of Tobit regression for those respondents who had visited Moreton Bay

<table>
<thead>
<tr>
<th></th>
<th>Normalized Coefficient</th>
<th>Asymptotic Coefficient</th>
<th>Regression Coefficient</th>
<th>Elasticities</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Standard error</td>
<td>T-ratio</td>
<td>Prob. +ive bid</td>
</tr>
<tr>
<td>Case B</td>
<td>0.89568E-01</td>
<td>0.14033</td>
<td>0.63828</td>
<td>3.2535</td>
</tr>
<tr>
<td>Case C</td>
<td>0.27172</td>
<td>0.13420</td>
<td>2.0248***</td>
<td>9.8703</td>
</tr>
<tr>
<td>Case D</td>
<td>-0.58213E-01</td>
<td>0.14211</td>
<td>-0.40965</td>
<td>-2.1146</td>
</tr>
<tr>
<td>Health</td>
<td>0.26459</td>
<td>0.67953E-01</td>
<td>3.8937***</td>
<td>9.6112</td>
</tr>
<tr>
<td>HW Camp</td>
<td>0.26502E-01</td>
<td>0.10017</td>
<td>0.26457</td>
<td>0.96269</td>
</tr>
<tr>
<td>Age</td>
<td>-0.24643E-03</td>
<td>0.29904E-02</td>
<td>-0.82405E-01</td>
<td>-0.89514E-02</td>
</tr>
<tr>
<td>Sex</td>
<td>0.17399</td>
<td>0.10495</td>
<td>1.6578*</td>
<td>6.3200</td>
</tr>
<tr>
<td>Education</td>
<td>0.39262</td>
<td>0.11028</td>
<td>3.5602***</td>
<td>14.262</td>
</tr>
<tr>
<td>Income</td>
<td>0.13075E-04</td>
<td>0.23988E-05</td>
<td>5.4504***</td>
<td>0.47493E-03</td>
</tr>
<tr>
<td>Constant</td>
<td>-1.2410</td>
<td>0.35173</td>
<td>-3.5284***</td>
<td>-45.081</td>
</tr>
</tbody>
</table>

The predicted probability of Y>limit given average X(I) = 0.7275

The observed frequency of Y> limit is = 0.8909

At mean values of all X(I), E(Y) = 28.0575

Log-likelihood function = -1999.1832

Mean-square error = 1152.9278

Mean error = 4.4063239

Squared correlation between observed and expected values = 0.17577

* significant at 0.1 level, ** significant at 0.05 level, *** significant at 0.01 level

For those who visited the Bay the provision of information in Case C had a significant effect on willingness to pay (compared with no information) as reflected in the significant difference in the intercept. As for the overall model attitudes towards the health of the Bay, education and income were all significant variables in explaining willingness to pay. Regressions were also run for the situation where Case C was the reference level. It was again demonstrated that there was a significant difference in the intercepts between Case A and Case C at the 0.05 level. There was also a difference in the intercepts between Case C and Case D at the 0.05 level. Again, these results are similar to the overall model.
Table 9.25 Results of Tobit regression for those respondents who had not visited Moreton Bay

<table>
<thead>
<tr>
<th></th>
<th>Normalized Coefficient</th>
<th>Asymptotic coefficient</th>
<th>Regression coefficient</th>
<th>Elasticities</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Standard error</td>
<td>T-ratio</td>
<td></td>
</tr>
<tr>
<td>Case B</td>
<td>0.36798E-01</td>
<td>0.28779</td>
<td>0.12787</td>
<td>0.59208</td>
</tr>
<tr>
<td>Case C</td>
<td>0.99588E-01</td>
<td>0.29529</td>
<td>0.33726</td>
<td>1.6024</td>
</tr>
<tr>
<td>Case D</td>
<td>-0.72631E-01</td>
<td>0.27289</td>
<td>-0.26616</td>
<td>-1.1686</td>
</tr>
<tr>
<td>Health</td>
<td>0.15554</td>
<td>0.82013E-01</td>
<td>1.8965</td>
<td>2.5026</td>
</tr>
<tr>
<td>HW Camp</td>
<td>0.21339</td>
<td>0.22853</td>
<td>0.93375</td>
<td>3.4335</td>
</tr>
<tr>
<td>Age</td>
<td>-0.14129E-01</td>
<td>0.59942E-02</td>
<td>-2.3571**</td>
<td>-0.22733</td>
</tr>
<tr>
<td>Sex</td>
<td>0.26471</td>
<td>0.23412</td>
<td>1.1307</td>
<td>4.2593</td>
</tr>
<tr>
<td>Education</td>
<td>0.54909</td>
<td>0.25087</td>
<td>2.1887**</td>
<td>8.8349</td>
</tr>
<tr>
<td>Income</td>
<td>0.13551E-04</td>
<td>0.51692E-05</td>
<td>2.6214***</td>
<td>0.21803E-03</td>
</tr>
<tr>
<td>Constant</td>
<td>-0.62723E-01</td>
<td>0.46345</td>
<td>-0.13534</td>
<td>-1.0092</td>
</tr>
</tbody>
</table>

The predicted probability of Y>limit given average X(I) = 0.7172
The observed frequency of Y> limit is = 0.7222
At mean values of all X(I), E(Y) = 12.0727
Log-likelihood function = -348.52306
Mean-square error = 160.97705
Mean error = 0.28486734
Squared correlation between observed and expected values = 0.24846
* significant at 0.1 level, ** significant at 0.05 level, *** significant at 0.01 level

For those who had not visited the Bay there was no difference in willingness to pay (as reflected in the intercept term) between those who had been provided with no information (the reference case) and those who had been provided with other types of information. The model was also tested using Case C as the reference level but again there was no significant difference. The results indicate that for those who had visited Moreton Bay the provision of information about use values produced a significant impact on willingness to pay. This may indicate that for those who already have made use of the Bay the information relating to uses was of greater personal relevance and hence produced a stronger response as suggested by Ajzen et al., (1996, 52). Respondent’s attitudes towards protection of wetlands to maintain the health of the Bay were also significant for those who had made use of the Bay. Although, the term health was used as a proxy for the ecological values of the Bay in
the question asked, it may be that respondents wish to maintain the health of the Bay to provide the option to continue to use it in the future. That is, the significance of this variable may be related to option value. As noted in Chapter 2, option value will be positive for risk adverse individuals where future supply is uncertain. Given that the survey instrument indicated that supply of future uses may be limited due to poor water quality, respondents may have been willing to pay a premium to ensure that the supply of benefits the wetlands provide continues into the future. Education and income were significant determinants of willingness to pay for both groups.

9.5 DISCUSSION

This contingent valuation study has attempted to assess the effects of providing a similar amount but differing type of information about a public good to respondents. The contingent valuation survey was undertaken to answer two main questions. The first question was: does the mean willingness to pay for ecological value exceed mean willingness to pay for other components of economic value? The benchmarks for acceptance or rejection of this proposition would require a significant difference between the provision of ecological information and other types of information. If it is accepted that ecological attributes are the basis for the provision of instrumental human benefits from the ecosystem under question then the willingness to pay for these values should exceed those of the secondary economic outputs. That is, the connection between ecosystem values and economic values would need to be constructed by respondents and expressed through willingness to pay bids. If the proposition was to be rejected then it would need to be shown that willingness to pay for ecological attributes was significantly lower than for the secondary economic outputs as expressed through use and non-use values. As neither of these situations can be demonstrated through the differences in willingness to pay amounts it is not possible to state definitely that the contingent valuation method can capture ecological value.

On reflection this is perhaps not a surprising result. As demonstrated in Chapter 2 and Chapter 3, and as suggested by Brouwer et al., (1997), natural scientists are still grappling with difficulties in disentangling the attributes and functions of wetlands. If the attributes of wetlands that provide instrumental human benefits were well
understood and accepted by wetland scientists there would not be the need to value these attributes using subjective human preferences. To expect a sample of the general population (which may have possessed prior information due to their proximity to the resource) to make the connection between ecological attributes and economic value may be overestimating the cognitive capacities of the public. Rather, as Brouwer et al., (1997) suggest a respondent’s view of wetland functions and attributes would depend upon their own experiences and does not necessarily correspond with the scientific attributes of ecological value. This suggestion corresponds with the results of this study that those respondents who had experience of the wetlands (visitors) provided higher willingness to pay than those with no experience (non-visitors).

If it is accepted that information is required to link ecological and economic values then information would need to be provided that explicitly described this link. This produces the classic claim that more research is needed on the subject before it can be stated that respondents to a survey are capable of assimilating information on and valuing ecological attributes. However, this again raises the possibility of a circular argument because if the ecological information were available that could detail the links between the two sets of values then the subjective preference of the public would not necessarily be required to make the most appropriate resource allocation decisions.

The second main question to be answered is: does the provision of a similar amount and quality but different type of information produce a differing mean willingness to pay bid? As noted in Chapter 5 studies by Bergstrom et al., (1990), Hanley and Munro (1992), and Hoevenagel and van der Linden (1993) among others, demonstrated that giving extra information increased willingness to pay bids up to a point where information overload was conjectured to occur. The analysis in this study has found a similar result for between the cases where respondents were presented with no extra information and use information. However, there was no significant difference in mean willingness to pay between the cases where respondents were presented with no information and non-use information, or no information and ecological information. In these latter cases, the provision of extra information did not result in a significantly higher willingness to pay bid.
The provision of a differing type of information did produce an information effect as assessed by a significant difference in mean willingness to pay for the case of use information against no information and non-use information. The result was also demonstrated using Tobit regression analysis where a significant difference was found between use information and non-use information. This result is an important contribution to the contingent valuation information literature as it demonstrates that it is not only extra information but the type of information presented to respondents that can alter their willingness to pay bids. However, there was no discernable information ‘type’ effect between no information, ecological information and non-use information.

To test the effect of previous knowledge the groups were divided into those who had visited and had not visited the Bay on the premise that visiting the Bay would lead to an increase in knowledge of the attributes of the wetlands. For those who visited the Bay, the results were similar for the overall case. However, for those who had not visited there was no effect on providing extra information or type of information. This may indicate that the information provided in Case C was relevant to those who already use the Bay but not for non-visitors. That is, it may not have been information about use, which influenced willingness to pay but previous use itself.

The regression analysis also indicates that respondents are willing to pay a significantly (p<0.05) lower amount for non-use values compared to use values. This result is similar to the findings of a meta-analysis of contingent valuation studies of wetlands undertaken by Brouwer et al., (1997). In their analysis they found that use values, such as flood control and water quality attributes, exert a stronger influence over willingness to pay than non-use elements. That is, on average willingness to pay was higher for use values than non-use values (Brouwer et al., 1997).

Although the provision of different information produced differing willingness to pay amounts it is still unknown exactly what values were being expressed by respondents. Respondents may have been motivated by their own intrinsic social and cultural values. Also, respondents who were given use information about the values
of the wetlands provided a higher willingness to pay response than those who were given information about the endangered species dependent on the wetlands. It then remains to be asked whether this indicates that the latter group confined their preference formation to only endangered species while the former group considered only the use benefits the wetlands may provide for them.

In the case of ecological information the willingness to pay response was not different from cases where other information was provided. Does this indicate that they considered both the underlying ecological value of the wetlands and the goods and services they provide? The point is that although different types of information were provided it is unknown how the respondent linked this information to the benefits that they believed the wetlands provided them with. As there was a high level of use and knowledge of the Bay it may be that respondents had formed their preferences prior to the survey and the provision of use information merely reinforced the value for these preferences. The problem may be similar to that of scoping or embedding which was outlined in Chapter 5.

Scoping or embedding will occur if respondents do not give significantly different responses to changed amounts of the item being valued (Rolfe, 1998). This can occur within a geographical context (the value of one wetland is not different from the value of a group of wetlands) or if they do not differentiate between sub-components of benefits. To overcome the possibility of geographical scope in this study, the respondents were selected from the adjacent region to ensure they were familiar with the wetlands. The use of a map to mark clearly the area in question was also provided to ensure the geographical scope was clear. However, some respondents may have based their willingness to pay bids on their previously conceived preferences for the Bay and not on the values outlined in the information set.

If stated preference techniques are to be used to assess the economic value of wetlands (or sub-components of value) it is essential that information be provided about that class of benefit. Further, when eliciting willingness to pay bids the elicitation question must explicitly state the class of benefit that is being assessed. Follow up questions can also be inserted after the willingness to pay question to
determine which class of benefits respondents based their bids on. The question is particularly important when the purpose of the assessment is to determine non-use values. If a willingness to pay bid is given that actually reflects the direct use value of the wetlands for the respondent and these direct use values have already been assessed through other methods double counting will result.

For example, the contingent valuation survey undertaken in this study provided both use and non-use information. If it was considered that values were formed only on the basis of the information given then the total economic value of the wetlands would be the willingness to pay for use values plus willingness to pay for non-use values. Until the motivation intentions of the respondents are fully documented it is unknown if this was truly the case and such assumptions should be avoided for fear of double counting.

The fact that respondents are not willing to pay a higher amount when presented with ecological information does not indicate that these values are not important but merely that there was not a significant difference in the willingness to pay for these values. The explanation offered for this result is that respondents consider ecological values important but not as important as direct and indirect use values with respect to willingness to pay. The conclusion that ecological and non-use values were considered important by respondents is drawn from the analysis of attitudinal questions related to the protection of wetlands. The attitudinal rankings of importance of protecting wetlands in terms of the value they provide was not the same as the rankings of values derived from the willingness to pay bids. Respondents clearly indicated through their attitudinal rankings that the health and non-use components of total economic value were considered more important than use value. This was confirmed by the use of a constant sum question that allowed respondents to directly record which elements of economic value were of highest value to them. The result however contrasted with the hierarchy of values produced by considering the mean willingness to pay for each survey version.

The explanation given for this discrepancy is that respondents used differing value systems for the two questions. From previous discussions in the contingent valuation literature (for example, Blamey et al., (1995) and Gowdy (1997)) it was proposed
that when asked for their attitudes towards wetland protection the respondent acted as a citizen and gave answers from a moral and political standpoint. When these same respondents were asked to pay for the protection of wetlands, they acted as consumers and those who were given information on the use values offered a higher willingness to pay. It was assumed that information on the direct and indirect use values had greater influence on respondents as it was more clearly related to their well being.

9.6 CONCLUSION

The purpose of this chapter was to present the analysis of the results of the contingent valuation study of Moreton Bay. The background information collected in the survey has been presented in terms of response rate and use and awareness of the Bay. Results indicate that respondents possessed sufficient knowledge from use of the Bay and awareness campaigns to form a value for the wetlands. The socio-economic characteristics were compared to the population from which the sample was drawn. Consideration was then given to zero bids that may reflect a protest to some aspect of the survey. The attitudes toward wetland protection was then analysed in terms of differences in attitudes depending on the value being assessed and difference in attitudes depending on the information provided for each case. These results indicate that the respondents consider the health and non-use values of the wetlands to be more important than the use values.

The mean willingness to pay for each case was then compared to assess ability of the technique to capture ecological value and if the type of information presented to respondents is able to alter willingness to pay bids. Regression analysis using the Tobit model was also undertaken to detect differences between cases and the effect of other factors such as attitude, use of the Bay, age, sex, education and income. With respect to the latter question, the results demonstrate that the type of information presented can alter willingness to pay bids. This is considered important as it reinforces the need for contingent valuation practitioners to ensure that information presented to respondents is accurate and not deliberately persuasive. The survey also demonstrated, when examining the difference between no information and use information that there was an effect on willingness to pay bids.
This accords with previous studies, which have examined information effects. The effect was not however evident in the case of ecological information or non-use information.

With respect to the former question, the results of the survey do not lead to a conclusive answer with respect to the ability of the technique to capture ecological value. Perhaps the information supplied was not adequate to convince respondents that ecological value is important. Given the complexity of wetland ecosystems and the concept of ecological value it is perhaps over ambitious to expect members of the general public to link the concept of ecological and economic value and provide a monetary estimation for ecological value. Thus, it is suggested that both ecologists and economists must engage in cooperative research to link the two sets of value. An approach where both ecologists and economists work toward an agreed purpose using an integrative framework is presented in Chapter 10.
CHAPTER 10

A METHODOLOGY FOR LINKING ECOLOGICAL AND ECONOMIC VALUES

The purpose of this chapter is to suggest an alternative approach for linking ecological and economic values. This approach is developed to overcome deficiencies that have been detected in other approaches presented in this thesis. That is, this alternative approach is suggested as an area of further research by building on the information that has already been provided. The difficulties with linking ecological and economic value that have been noted include that the required information to link ecosystem functions to economic services at the ecological-economic interface is not well developed. A particular difficulty in this area is the need to understand marginal changes in economic value which requires the need for multiple studies over time. Further, it was not possible to determine the ecological value of wetlands that provide for instrumental human benefits due to a lack of ecological information. The alternative approach of capturing ecological value through the contingent valuation method did not produce conclusive results. Nevertheless, the interdependence of ecological and economic systems dictates a need for linking ecological and economic studies.

The usual approach is for ecologists and economists to undertake research within their respective disciplines and then to present their findings independently with no necessary linkages between the two. It is rare to find individuals who are jointly trained in ecology and economics that would be able to overcome this barrier. For example, in this thesis ecological and economic values of wetlands have been presented based on research undertaken in the respective disciplines. Yet there was limited information within the literature to detail the links between the two as this has not been the purpose of either discipline. The limitation of serial approach is that it does not provide information on the likely impacts of a change in ecological condition on the socioeconomic system. Nor can it explain the likely impacts of the socioeconomic system on the environment.

In this chapter it is argued that a further layer of information is required that explicitly links the two systems. This information will be in addition to other relevant ecological
and economic data. The approach for linking the systems should avoid the previous serial tendency, noted above, where ecologists collected information that is then interpreted by economists. Norton (1998) argues that the approach must be iterative and multidirectional. Building on the arguments of Norton (1998) a methodology for an iterative and multidirectional approach is presented to illustrate how the ecological and economic values of wetlands could be linked.

### 10.1 OUTLINE OF APPROACH

The approach presented for linking ecological and economic values is focused on the information and research presented in previous chapter. A framework that explores the interaction of the ecological and economic systems and has been applied to wetlands in general and Moreton Bay in particular has been presented in Chapter 2 and Chapter 6. An attempt has been made to link the ecological and economic values by outlining the attributes of ecosystems that could be considered important from the perspective of providing for instrumental human use in Chapter 3. From this analysis, it was argued that an index of ecological value would include the attributes of productivity, dependency and biodiversity and organisation. This index of ecological value has been compared with other indices used to measure ecological health and integrity.

Shogren and Nowell (1992) identified a lack of a well defined theoretical value system within ecology. Thus the possibility of using economic techniques to measure ecological value has been investigated within this thesis. A number of criteria was introduced in Chapter 4 to assess if any of the techniques is capable of capturing ecological value. It was determined that the stated preference technique of contingent valuation could possibly measure willingness to pay for ecological values if the respondents to a survey were provided with sufficient information. A case study of the wetlands of Moreton Bay was undertaken. Respondents to the survey were willing to

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1. However, it has been noted investigations on the interface between the two systems have been limited. Nevertheless, the underlying ecological features of wetlands and the secondary economic benefits they provide have been identified.

2. This approach was taken as there was not a well-defined concept of ecological value within the literature.
pay for the protection of wetlands when given ecological information. However, in this study it is argued that respondents were not able to assess fully the underlying ecological values of the system. Although it is not possible to generalise from a single study, the results suggest that it requires both ecologists and economists to contribute to the linking of the two values. Therefore, the approach presented here has been developed to direct explicitly how ecological and economic values could be linked. The approach requires a willingness to engage in cooperative research projects to identify both ecological and economic values and indices that can link the two disciplines.

Within the proposed approach, the types of questions and issues that would need to be addressed include: Does the issue/problem/management objective include both an ecological and economic component so that there is a common element that can form the basis of an integrated research project? Given the issue under investigation, what is the appropriate temporal and spatial scale of assessment given that there may be differences in spatial boundaries and time taken to collect and assess data for the two groups of researchers? In addition, the rate at which changes in ecological functions impacts on economic values needs to be considered. The researchers also need to link conceptually the impacts that changes in ecological value will have on economic values, so that the appropriate features of the ecosystem and its economic values are assessed. Further, to communicate the linkages between the two areas of study requires indicators which are compatible to the spatial and temporal dimensions of the disciplines. That is, economists do not need to understand all the details of the ecological assessment but they must understand the implications of the assessment. If the assessment is to be multidirectional these issues may need to be reassessed as the project proceeds to take account of new information.

In order to address these issues and questions, five essential stages of the proposed approach have been identified. First, Folke (1991) argues that the interface between the

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3 It should be noted that there will be many ecologists and economists whose research interests would not make them willing participants in the approach suggested. Thus, although the terms ecologist and economist are employed to differentiate the researchers in the two disciplines the roles given within the proposed approach would not extend to all those who engage in ecological or economic studies.
 ecological and economic systems can only be understood through direct dialogue between the practitioners of each discipline. This can be achieved through directing research to a common purpose, whether it be a management objective or in reaction to a perceived threat to the ecosystem. Ecologists and economists must co-develop the stated aim of the investigation and the spatial and temporal scale at which investigations will take place. Second, ecologists and economists must develop the tools that allow a common approach to be taken. This may be realised through the use of indicators and terms that include both an evaluative component and a normative component (Norton, 1998). For example, if ecologists are to measure water quality, the benefits (or losses) resulting from the change in water quality must be clearly articulated for the approach to be truly integrative. Economists must also develop indices, which indicate how the components of the total economic value provided by the ecosystem change in light of ecosystem changes\(^4\). Values identified in economics might then lead to further ecological investigations so that the approach is truly multidirectional. The next two stages may then be undertaken within the differing disciplinary paradigms. Ecologists continue to provide information via assessment and monitoring on the change in ecosystem condition. However, the assessment needs to be extended from simply explaining changes to demonstrate how changes in structure, process and functions will affect the outputs of the system. Similarly, economists must attempt to value the changes in economic benefits that arise from changes in ecosystem condition. Finally, the process must be iterative and the study should be reviewed to determine if the expected changes to both ecosystem and benefits have eventuated. Each of these stages is outlined. Figure 10.1 illustrates the stages and iterative nature of the methodology that is developed within this Chapter.

\(^4\) It may never be possible to measure total economic value nevertheless all components of economic value that are able to measured should be included.
Figure 10.1 An iterative approach for linking ecological and economic values

Stage 1: Co-develop purpose and scale of assessment

Stage 2: Develop discourse between ecologists and economists and indicators or indices

Stage 3: Undertake ecological assessment–including changes to ecosystem outputs

Stage 4: Undertake economic assessment of changes in ecosystem outputs

Stage 5: Assess linkages between indicators and review research

The current ecosystem health monitoring program (EHMP) in Moreton Bay is used to provide an example of how the methodology could be applied and to illustrate how monitoring ecological changes in the environment could be linked to changes in economic benefits.

10.2 STAGE 1: CO-DEVELOP PURPOSE OF ASSESSMENT AND APPROPRIATE SPATIAL AND TEMPORAL SCALE.

This thesis has attempted to link theoretically the ecological and economic systems through the concept of value. However, there is no well-developed general theory of value in ecology. Ecologists do however study certain attributes of ecosystems at a local level. Linkages between the two systems could begin to be forged at this level. As information is amassed on the linkages at the specific local level it is assumed that a more general theory will evolve.
A driving force behind many environmental research projects is the perception by resource managers or the community that a problem exists due to ecological degradation. To deal with these perceived problems in Australia, legislation has mandated that management of the environment should be directed towards the goal of ecological sustainable development (Commonwealth of Australia, 1992). Therefore, for any issue there will be a defined management objective. Ecologists and economists must first come together to determine if a problem or specific issue exists and identify the connections between the ecological and economic components of the issue/problem. To continue assessment for pure understanding will not be adequate. If an integrated approach is to be achieved practitioners of both disciplines must agree that the purpose of the assessment is to maintain the ecological system so that the economic values of the system are also maintained. This follows from the assumption that the economic system and hence humanity is ultimately dependent on ecological systems (Norton, 1998). That is, the purpose of an ecological economic assessment should be ends driven. That end will include both an ecological and economic dimension.

A consensus between ecologists and economists will only be reached if both parties are willing to accept the codependency of the ecological and economic systems. If ecologists do not agree that the purpose of maintaining ecosystem health is provide for human welfare then an integrated approach will not be possible. To reach consensus on the management objective/problem will thus require that both groups of researchers see merit in linking ecological and economic values. Once it is accepted that there are important links between the ecological and economic dimensions of the issue then it should be possible to reach a consensus on the nature and implications of the problem/management objective.

For example, it may be apparent that wetlands are being lost due to a deterioration in water quality. First, it must be agreed that wetland loss is a problem. Second, ecologists and economists must state why it is a problem. The sort of questions that need to be considered would include: What are the ecological consequences of wetland loss? How will this impact on ecological structure, process and function? As a result of changes in
structure, process and function what will be the changes in output of the system? How will these changes in output result in changes in the instrumental human benefits provided by the wetland? These questions must be addressed through dialogue between the two disciplines, not in isolation. These questions straddle the ecological-economic interface and require a two-way transfer of information. Economists need ecologists to inform them of changes in output and ecologist must understand that outputs provide for human needs.

For example, the Ecological Health Monitoring Program (EHMP) in Moreton Bay is being undertaken to measure the ecosystem health of the Moreton Bay and five major river estuaries. The definition of ecological health stated by the EHMP is “key environmental processes operate to maintain stable and sustainable ecosystems, zones of anthropogenic impacts do not deteriorate and critical habitats remain intact” (Dennison and Abal, 1999, 208). Thus, the EHMP has identified certain attributes of the system they consider important for maintaining ecosystem health. The purpose of the EHMP is to assess the ecological outcomes of management programs, evaluate the effectiveness of sewage plant upgrades, stormwater controls and wastewater treatment and fulfill licensing requirements. That is, the program is aimed at determining the ecological benefits of economic investments. The program does not address the economic benefits that flow from the investment in protection of the ecosystems which is the interest of this thesis. This approach has been successful in determining the direction of local government spending on sewage and wastewater treatment.

The EHMP has been conducted in conjunction with the Waterways Management Plan, which has the vision (adopted by all major stakeholders in the study) that “Moreton Bay and its waterways will by 2020 be a healthy ecosystem supporting the livelihood and lifestyles of residents and visitors”. However, the monitoring program has concentrated almost exclusively on the health of the ecosystem with little consideration given to how this is linked to the livelihood and lifestyles of residents and visitors. If this assessment were to be undertaken within an ecological economic framework then the changes in ecosystem health being monitored would need to be explicitly linked to the desired
livelihood and lifestyles of residents and visitors. The EHMP has made understanding ecological change central to its mission but as Rapport et al., (1998) state it, to understand the significance of those changes depends critically on the implication for ecosystem services. To make this link between ecosystem health and ecosystem services requires ecologists and economists integrating their research across the ecological-economic interface to determine the outputs of the system and how they contribute to instrumental human well-being.

Having agreed that there is a problem to investigate or a management objective to meet an appropriate scale of assessment needs to be considered. As noted in Chapter 2, scale refers to the temporal and quantitative dimensions that are used to study objects and processes (Gibson et al., 2000, 218). The temporal and spatial extent of scale that is chosen will determine the boundaries of the assessment. However, there is no reason to assume that the boundaries of the extent of study will coincide for the two disciplines. For example, the spatial extent of a wetland may straddle two human boundaries such as a local government area or state. The economists may only be interested in analysing the benefits of the wetland for a single, local government area but it makes little sense for the ecologist to study only half a wetland. Nevertheless, the services delivered by the ecosystem under investigation may extend well beyond the area which is being investigated by ecologists. Thus, a further problem with choosing the appropriate extent of the spatial scale is that ecological services flowing from the spatial scale of a wetland will extend beyond that wetland. It has been emphasised throughout this thesis that the benefits accruing from wetlands extend from local subsistence needs to global biogeochemical cycling. Therefore, if economists limit their studies of changes in wetland services to only the ecological spatial scale they may underestimate the full value of the changes that are occurring.

The temporal extent of scale will also depend on the issue to be resolved and the necessary resolution of data. Changes in ecological condition can be measured from hours to days to years to decades to centuries. The more intensive the monitoring effort the greater will be the precision of the measurement of change. Nevertheless, some
changes will only be detected by considering long-term trends. For example, as shown in Chapter 6 there has been a long-term loss of seagrass in Moreton Bay (Abal et al., 1998). The measurement of change in economic value requires a resolution of data that is capable of detecting changes in economic value. As noted in Chapter 4, the economic value of an environmental resource is measured at the margin. That is, a value is determined by assessing the willingness to pay for some marginal change in the resource. The question then becomes what is the temporal scale at which change can be measured. At one extreme, the complete removal of some ecosystem feature (for example, all the wetlands of Moreton Bay) may produce a dramatic change in the services supplied. However, with persistent and cumulative degradation of the system, the changes in ecosystem function and hence services may be more difficult to discern and marginal changes may not be detected in an economic assessment. Thus, economists may not be able to monitor change at the same temporal scale as ecologists.

The issue of the appropriate temporal scale at which economists can measure change is also influenced by the resolution of available data. Information on willingness to pay may be collected through surveys. The measurement of change will then depend on the frequency of data collection, that is the difference measured between surveys. Alternatively, stated preference methods may include a change over a specific period within the scenario. For marketed goods, the availability of data on prices and its aggregation will also influence the time period over which change can be assessed. For example, if fish catch statistics are only available annually at the level of the state economy resolution will be poor.

The hierarchical level of the study will differ between the disciplines. Ecologists studying wetlands could choose a hierarchical level from a population of a single species to the entire ecosystem (Mitsch and Gosselink, 2000). Economists may consider the benefits of wetland to accrue anywhere from the level of the individual/household to the regional, national or international economy level. Thus, to integrate ecological and economic information requires the ability to operate at two levels simultaneously or choose a level of aggregation that will encompass both.
The EHMP in Moreton Bay provides a good example to suggest solutions for the appropriate choice of extent and resolution of scale for both ecologists and economists to study a particular problem (Dennsion and Abal, 1999). The spatial extent of the assessment includes most of Moreton Bay and the estuaries of the major rivers of the Moreton Bay catchment. This scale is based on the physical boundaries of ecological assessment. The extent of the economic spatial scale is based on the jurisdictional boundaries of the local council areas within the region and these differ from the physical boundaries. The inherent mismatch between spatial scales has been overcome by the collaborative efforts of the local councils to work in unison to develop solutions for the whole catchment and Bay. Nevertheless, the vision of the Waterways Management Plan is concerned only with range of benefit provided to residents and visitors of the area. However, the spatial extent of the assessment excludes the southern areas of the Bay. It may be expected that changes in ecosystem condition in the study area will also influence this area. Thus, if the economic scale of assessment did not include this region the change in benefits from changes in ecosystem health may not be fully captured. Further, as indicated in Chapter 6 the range of benefits from maintaining the health of Moreton Bay will extend indirectly to humans beyond the Moreton Bay region. Temporally, the extent of the ecological assessment is undertaken on a monthly basis at over 100 sites, as an annual survey plus more intensive sampling is undertaken at selected sites. However, ecosystem health changes that occur monthly or annually may not be great enough to be reflected in changes of economic value. For example, the annual report card that rates the various regions with the Moreton Bay catchment system from A+ to F has shown little variation between 1998 – 2001 (MBCWQMST, 1998; EHMP, 2001a). An initial scale of economic investigation may require a change of at least one full level (i.e. from A+ to B+) before undertaking an assessment of changes in economic value. Consideration would also need to be given to data limitations and lags between changes in function and output.

A more specific problem illustrates these difficulties. In Chapter 6, it was noted that there have been outbreaks of the toxic cyanobacteria, *Lyngbya majuscula* in Moreton
Bay. These outbreaks are considered one of the greatest risks to ecosystem health in the eastern side of the Bay (Grice et al., 2000). Also, anecdotal reports suggest that over successive summers, since 1992, blooms have occurred covering an area of approximately 7 km² – 10 km² in Deception Bay in the western side of the Bay (Dennison and Abal, 1999). The bloom in the summer of 1999/2000 on the eastern side of the Bay at Amity and Moreton Banks covered 30 km² (Grice et al., 2000). Thus, the area impacted by the blooms determines the spatial extent of ecological study of this phenomenon. Temporally, monitoring continues annually particularly over the summer season. During outbreaks, resolution of data is high as monitoring is conducted by both remote sensing on a daily basis and with less frequent field trips. However, to estimate changes in function and outputs of the wetlands resulting from the blooms may require a greater extent of assessment. For instance, there may be little impact on populations utilising the seagrass if similar areas are available nearby. Thus, the abundance of species in these areas should also be included in the assessment.

The economic impacts identified from the outbreaks include loss of habitat for turtles and dugongs which may lessen non-use values. As noted in Chapter 6, the outbreaks of *Lyngbya* are correlated with the loss of seagrass in Deception Bay and it is suggested that there has been an associated loss of fisheries harvest (Dennison and Abal, 1999). *Lyngbya* is also toxic to fish (MBCWQMST, 1998). Thus, it may be expected that there will be a loss of recreational and commercial fishing values resulting from the blooms. Further losses in recreational value may also occur as Dennison and Abal (1999) note that following blooms large wracks of decaying *Lyngbya* gather on beaches in the region causing a putrid odour and requiring removal by local authorities. The loss of seagrass and the subsequent maintenance of indirect use values for nutrient assimilation are unknown. The spatial extent of these losses of benefits extends over a far greater area than where the *Lyngbya* occurs. Thus, an economist must use a spatial extent of the regional economy. Loss of fish habitat areas may also influence off-shore fisheries but data are not available to confirm this. The temporal extent of economic studies will be determined by time over which changes occur and the resolution of the data. Although there may be some immediate loss of recreational amenity and fishing values when the blooms occur there may be a lag before other changes can be detected. For example, it
may be a number of years before any change in fish catches due to the smothering of the seagrass beds becomes apparent. This would depend on the life cycle of captured species, that is the number of years it takes juveniles who would have gained food and protection from the seagrass to reach maturity. Therefore, economic studies would need to monitor these changes over several years. The resolution of available economic data may also make it difficult to correlate the change in a particular value to the occurrence of the *Lyngbya* blooms. Data may only be collected annually or monthly at best and is likely to be aggregated for the whole of the Bay. Monitoring intensity would need to be increased to improve resolution and hence the ability to detect changes over a time frame similar to ecologists. For example, if recreational amenity alters due to *Lyngbya* blooms then surveys of recreational behaviour would need to be conducted during the event as annual data may obscure changes.

Thus, if the problem to be investigated was *Lyngbya* outbreaks, at the completion of stage one, ecologists and economists would agree upon a clear need and desired end for the assessment program. In addition, the temporal and spatial scale of assessment will be clearly articulated by both groups in terms of spatial boundaries of the study, frequency of data collection and the time horizon of the study. It would then be necessary to develop a discourse that allowed the information from the two assessments to be linked. This task needs to be undertaken in stage 2.

10.3 STAGE 2: DEVELOP A DISCOURSE THAT ALLOWS A COMMON APPROACH TO THE PROBLEM/MANAGEMENT OBJECTIVE.

A common discourse needs to be developed that allows the problem/management objective to be viewed from a common perspective. This discourse will include both the ecological dimension of the problem and the way this dimension is linked to socioeconomic indicators. However, the individual disciplines of economics and ecology have differing paradigms and hence differing assumptions, methods and semantics (Norton, 1998; Bingham *et al.*, 1995). Nevertheless, within each discipline a variety of sub-disciplines exist which may hold differing views with respect to the general representation of the two disciplines outlined here (Norgaard, 1989).
For example, Shogren and Nowell (1992) argue that the two disciplines differ in their approach to experimentation. Ecology is based on observation induced descriptions, which lead to competing theories without a hierarchy of axioms and laws. On the other hand, economics is based on the tradition of logical positivism using theory induced propositions which has led to the dominant paradigm of neoclassical optimization theory (Shogren and Nowell, 1992). Further, Shogren and Nowell (1992) suggest that the two disciplines differ with respect to the objective function of the model. The objective function describes the cause-effect or dose-response relationship. For ecology the aim is to describe how the function works which due to the complexity of ecosystems does not lend itself to axiomatic descriptions. Economists assume that the objective function is well defined and is based on theoretical axioms of preference or production. More recently, economists have begun to test the behavioural assumptions on which the theory rests (Shogren and Nowell, 1992). This experimental work includes laboratory testing and field based environmental valuation experiments such as contingent valuation studies.

Bockstael et al., (1995) note that economists study the allocation of scarce resources among humans while ecologists study the allocation of resources for all other agents in the biosphere. Bockstael et al., (1995) also argue that ecologists study physical flows of water, biomass, energy or spatial connections while economists study flows of information based on money and prices as the basis of change. Differences in terminology are also likely to impede a free flow of information between the two groups. Bingham et al., (1995) suggest that semantic difficulties are a barrier in the progress of the integration of the two disciplines with terms such as ‘benefit’, ‘value’ and ‘function’ having different meanings across the disciplines. The differences in terminology are related to the approaches of the two disciplines. For example, ecologists use a language that characterizes the world while economists try to use this information to understand its impacts on societal well being. As Norton (1998) states it, one language describes the world while the other evaluates it.
Given that the practitioners of the two disciplines take different approaches to experimentation, populations of interest and terminology, there is a need for some form of interdisciplinary glossary that is able to translate the work of one group for the other and vice versa. For example, ecologists would come to such a discussion armed with a range of ecological data that indicate changes in the environment. Similarly, economists will present information of the range of social and economic values that may be impacted on by the changes. The goal is to link these disparate sets of information in a manner that it is both clear and accurate to both groups.

One means of achieving this dialogue is through the development of indicator terms or an index that represents both the measurable characteristics of the ecosystem and also reflects impacts on social values. As Norton (1998) states it, the value of indicators is that they provide a clue to a matter of larger significance. Schiller et al., (2001) argues that communication between the two disciplines requires that ecological indicators are translated into an understandable language and to articulate the economic values that are associated with those indicators. Thus, ecologists should employ indicators that encapsulate information about ecosystem condition and changes. These indicators then need to be linked to an economic indicator that illustrates how economic values are likely to be affected by the condition or change. If a discourse can be developed that retains scientific accuracy and embodies an evaluative component without having to convey detailed ecological or economic information then communication will be enhanced between the two disciplines.

Terms such as ecological integrity and health have been developed to try to link the discourse of ecology with the normative discourse of evaluation (Norton, 1998). The indicators that are employed should be focussed on the agreed problem/management objective. However, in order to develop such indicators ecologists must be able to demonstrate how the characteristics they are measuring will lead to changes in the functioning and output of the system. Similarly, indicators used by economists must reflect changes in ecosystem outputs. To develop these indicators economists must work with the ecologists to understand the full suite of benefits provided by these
outputs and inform ecologists of the nature of economic values. Thus, the indicators should be developed in a collaborative fashion so that they reflect the interdependency between the ecological and economic systems.

The discourse chosen to link the ecological and economic systems in this thesis has been one of values. An index of ecological value was developed to include those attributes of ecosystems, which provide for instrumental human values. The purpose of developing the index was initially to be able to compare systems on the assumption that some ecosystems are more valuable than others to humans. The index could also be used to determine what values would be lost if a system was degraded or destroyed. Through the inclusion of specific ecological value, assessing proposed changes could incorporate the value of the system and not just the partial products, which are economically valuable. However, the index by necessity was developed from an economic perspective and derived from the attributes of conservation value. An identified problem with the index was the lack of information available to make it operational.

Other approaches that have been developed to link ecological and economic include indices of health and integrity. However, it would not be expected that full information is available for any index developed\(^5\). For example, Costanza’s (1992) index of ecological health requires the same information on productivity/vigour and organisation as the index of ecological value. Further, Costanza’s (1992) index requires that the resilience of the system is maintained. The maintenance of resilience is an explicit management objective and although important is a difficult concept to measure. Also, Rapport (1995) notes that there is a plethora of approaches for measuring ecological health. Thus, the attributes chosen for any index will reflect both the needs of the assessment and ability to measure the attributes. The most important aspect to be considered in developing any indices is that they reflect the underlying complexity of the ecosystem (Haskell et al., 1992).

\(^5\) The index of biological integrity which only uses information on one taxa would be an exception to this general conclusion.
However, if stage 1 of the proposed methodology has been completed successfully, the range of measurements likely to be undertaken and their possible links to changes in economic benefits should already be clear to both the ecologists and the economists. Thus, an indicator can be developed at the local level that can link the ecological condition (as proposed to be measured) with the economic benefits (as proposed to be included in the assessment). It is not necessary to amass information on all dimensions of natural variation. Rather the goal should be to trace, evaluate and communicate the condition of the system and the consequences on human activities. Bingham et al., (1995) argue that it is information that permits the implicit question to be answered or decisive information that is required. Further, the index or indicators developed should not be restricted to narrow indicators of commodity production or status of threatened species.

For example, the ecosystem health report card for Moreton Bay, developed by the EHMP communicates to the public the outcomes of scientific assessment and monitoring (MBCWQMST, 1998). The report card is then an indication of the state of health of the ecosystems within Moreton Bay. However, this notion of health must be extended to encompass the values held by people with respect to a healthy environment and the range of goods and services provided by the wetlands of Moreton Bay. Chapters 2 and 6 have outlined the range of goods and services both use and non-use that are possibly provided by the wetlands. As noted, it is not sufficient to lay these values on top of the scientific indicators in a serial fashion as it will not facilitate the flow of information (Norton, 1998). Ecologists working on assessment in the Bay need to work with economists to determine the possible linkages between the two disparate areas of information and agree on extending the health metaphor to include anthropogenic benefits.

For example, the EHMP uses ecosystem health indicators such as monthly water quality, capacity and frequency of algal blooms and seagrass distribution and depth range to produce its report card rating. Similar indicators that could be encapsulated within an index (similar to the report card) need to be developed by economists. This would entail
detailing the range of economics benefits currently provided by the wetlands and
developing indicators that demonstrate how these benefits have altered in response to a
change in ecosystem health. For example, Chapter 6 outlined the economic values of
the wetlands considered important. These included the use values of fishing
(recreational and commercial), recreational amenity and waste assimilation. The non-
use value included bequest values and the value of endangered species dependent on the
wetlands. An indicator could be developed that would include changes in each of these
values that result from ecological change.

Having developed an indicator, whether it be ecological value, health or integrity, it
should then be possible to develop a conceptual notion of the likely changes in economic
value as a result of ecosystem changes before monitoring and assessment takes place.
This may entail considering the expected impacts of a change in each ecosystem health
indicator in terms of ecosystem services and goods. For example, if seagrass
distribution and depth range is reduced it may be expected that this will affect fish stocks
and stability of the seabed. Economic indicators used to measure this change may
include changes in commercial and recreation fish catches. This is an important element
in the methodology as the process is expected to be iterative as required for adaptive
management. Learning by both ecologists and economists will take place by reviewing
the changes that were found and comparing them to those expected. Following
acceptance of indicators by both ecologists and economists it is the role of each to assess
and monitor the attributes included within the indicators.

10.4 STAGE 3: MONITORING OF ECOSYSTEMS TO INCLUDE
ASSESSMENT OF CHANGES IN OUTPUTS

Stage 3 and 4 of this approach to link ecological and economic values need to be
undertaken in parallel. Although the studies are conducted independently it does not
imply that they should be undertaken in isolation. It will be necessary for the flow of
information to continue between the two groups throughout monitoring to ensure the
objectives of assessment are being met. For example, quantitative ecological
information about an increase or decrease in an ecosystem service may be a pre-requisite
to determine which valuation studies to undertake but values may also determine (implicitly or explicitly) which services one chooses to measure (Bingham et al., 1995). However, as noted small changes in ecosystem condition may not be sufficient to drive changes in economic value that can be readily assessed. That is, if a multidirectional approach is to be taken information must reviewed throughout the study so that the science is informed by economic values as well as informing economists about the services provided by ecosystems (Norton, 1998). Thus, communication is necessary to allow unexpected findings to be more fully pursued and related to the indicators that have been developed in Stage 2.

Ecologists have a range of monitoring tools that can assess changes in water quality parameters, identify changes in critical habitat, nutrient processing and sediment nutrient fluxes (Dennison and Abal, 1999). These tools will be employed in Stage 3 to provide an assessment of anthropogenic and other influences on wetlands and changes in the status of the ecosystem. However, if these ecological indicators are to be linked to changes in instrumental human benefits the assessment needs to be extended. For example, if the ability of wetlands to process nutrients has been exceeded then the consequent changes on ecosystem function need to be documented. Further, changes in ecosystem functioning need to be related to changes in the expected outputs from the system. This is not a simple task and complete information on the interconnection between structure, process and function and all ecosystem outputs is unlikely to ever be fully known. Nevertheless, ecologists must in some way provide information to economists on at least the possible impacts of a change in ecosystem functioning, process and structure that is related to the benefits likely to arise from the system as agreed in Stage 2.

Models could be developed to facilitate an understanding of the change in output of the system arising from changes in ecosystem functioning. For example, Bockstael et al., (1995) have begun to integrate ecological and economic modelling and analysis in the Patuxent River drainage basin in Maryland, USA. The purpose of this project is to model the interaction of ecological processes and human activity. The model includes
an ecological and economic component. The ecological component termed the Patuxent Landscape Model simulates the succession of the ecological system using a landscape perspective. The economic component models the impacts of land use and agricultural decisions and the effects of these decisions on the environment, zoning, transport and agricultural policies. Norton (1998) criticises this approach due to its serial nature as the economic modelling is undertaken subsequent to the ecological descriptions. However, Bockstael et al., (1995) argue it is a parallel approach. Either way models such as the one being developed by Bockstael et al., (1995) when calibrated to real life situations will improve the information required for the approach suggested here.

In this thesis, the framework to illustrate the connection between ecological and economic values has been presented in Figure 2.1. It has also been noted that ecosystem functions may produce services interdependently and simultaneously. If the functioning of the ecosystem could be directly linked to independent outputs, such as fishery resources, the economic assessment of the value of wetlands would be more complete. A number of studies that have attempted to link wetlands to fisheries production have been reviewed in Chapter 2 and Chapter 4. However, the approach is not without pitfalls. The dynamic interaction of functions means that it may not be possible to link one function with one output. For example, fish may be dependent on the structure of wetlands to provide protection from predators, nutrient processes to provide food and the biogeochemical functions to maintain a suitable level of water quality. Similarly, these same structures, processes and functions may provide other outputs of economic value, such as habitat for endangered species that have existence value. To impute a value for each input and output would require disentangling these links to avoid the issue of double counting of values.

Chapter 2 reported a number of studies that have tried to link wetlands and fisheries were outlined. However, these studies have only reflected a small number and type of commercial fisheries (Rönnbäck, 1999). For example, the linkages between mangrove area and fisheries production in the Ruitenbeek (1994) study had to be assumed due to lack of ecological information. Thus, even these studies will underestimate the true
value of the wetlands as they do not include all fish and shellfish species dependent on the wetlands and exclude the value of subsistence and recreational fishing from the calculations. To overcome this deficiency so that economic value attributed to wetlands from fish production reflect their true value requires much greater information from ecologists. The information required from ecological studies include answers to questions such as – What are the species that are dependent on wetlands for some part of their lifecycle? Which of these species are important for commercial, recreational and subsistence fisheries inshore and offshore? What are the structures, processes and functions that are crucial for wetlands to provide for dependency? What is the correlation between wetland area and fisheries production? Is there some threshold for the size of wetlands for sustaining fishery production? This information is required before economists can build models which try to estimate the value of wetlands as an input into fishery production. Barbier (2000) notes that then economists must establish the catch and effort involved to estimate the cost of harvest which in turn will be influenced by the institutional arrangements for the fishery.

However, further problems may arise with this approach if only the partial direct use values are included in the analysis. This will limit the understanding of the full range of economic benefits provided by the system and may limit management to a sectorial approach based on only one resource. As noted in Chapter 3, Holling and Meffe (1996) argue that a focus on exploiting a single resource can lead to a collapse in fisheries and eutrophication of water bodies. Adaptive ecosystem management will be directed towards maximising the mix of goods and services provided by wetlands at all scales while understanding the trade-offs between services. Nevertheless, considering the environment as an input into the production of goods and services may provide information about some of the more direct use values of the wetlands. Where this link is made the ecological and economic value of the wetland can be partially determined using methods such as the production function approach.

Currently, the EHMP in Moreton Bay assesses ecosystem health by monitoring:
water quality, extent of sewage plumes, seagrass loss and recovery, toxic algal blooms, occurrence of nuisance algae, phytoplankton growth responses and turtle and dugong populations (MBCWQMST, 1998). These assessments then form the basis for determining changes in ecosystem health. However, it needs to be demonstrated how and why those changes in ecosystem health will affect economic benefits derived from the Bay. For example, what is the impact of changed water quality on the benefits ascribed to the wetlands of Moreton Bay? Will fishery production change? Will recreational opportunities be decreased? What is the impact on the biodiversity of the Bay? As noted above, it may not be possible to link each parameter to a component of economic value. The assessment should, however, include an indication of at least the likely direction and magnitude of those changes. If Stages 1 and 2 of the methodology have been successfully completed then the focus will be directed to the likely benefits already agreed upon. That is, changes in the ecosystem health card need to be clearly linked to changes in human benefits.

10.5 STAGE 4: EVALUATION OF BENEFITS/LOSSES RESULTING FROM CHANGED ENVIRONMENTAL CONDITIONS

Having determined the likely range of goods and services that will be provided by the ecosystem through the information flows developed in Stage 2 it is the role of economists to value the services provided. Chapters 2 and 4 have outlined possible approaches for measuring the economic value of the components of total economic value. Each method has its own advantages and disadvantages and some are activity or site dependent. For example, direct use values could be imputed using production function techniques, travel cost and hedonic pricing. The production function approach can be employed to measure changes in fishery production related to ecological changes within wetlands. If ecologists have also been able to identify the linkages between ecosystem function and output, this method can compute a direct value for this resource. Market prices can be used to assess changes in value of the resource, although care must be taken to ensure that changes in catch and effort and capital expenditures are also included within the calculations. The travel cost method is best suited to measuring the recreational benefits provided by wetlands. Hedonic pricing can be employed to
measure changes in housing prices with respect to their vicinity to wetlands. However, it has been noted that in many instances wetlands may provide disamenity (in the form of mosquitoes and smell) which may actually lower property values adjacent to wetlands.

Thus, to determine the initial economic values of the system under investigation and changes to those values will most likely entail undertaking a number of studies using a range of techniques to assess the range of ecosystem goods and services provided by the wetlands. Although ecological assessments such as the EHMP use a range of monitoring and assessment tools to establish the ecological condition of a system, multiple economic studies of an area are less common. For example, in Moreton Bay there have been very few economic studies to determine the value of the ecological goods and services provided by the area\(^6\). Thus, the tasks suggested in this stage are of greater complexity and magnitude than previous estimates of the economic values of the region. However, if economic studies were funded at the same rate as ecological studies this situation could be changed. For example, the EHMP study cost $2.29 million over three years and an expansion of the study is planned at the cost of $1.32 million per annum (EHMP, 2001b). The same level of funding could be provided to undertake the assessments suggested within this stage of the approach.

The types of economic studies that could be undertaken can be illustrated by considering the change in economic value resulting from a change in water quality. If water quality deteriorates due to excessive sediment loads and impacts on wetland structure and processing it may be expected that fish catches would decline\(^7\). From information provided by ecologists and government agencies on the changes in fish stocks a value for the change could be imputed by economists using catch and effort and market price data. In addition, recreational value may be reduced as water quality falls from a

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\(^6\) Studies undertaken in Moreton Bay include Driml and McBride (1982) who undertook a travel cost study of southern Moreton Bay and Morton (1990) who estimated the average value of fish catch from mangroves. There are no known contingent valuation studies on Moreton Bay apart from the one presented in this thesis.

\(^7\) For example, there is clear evidence of a link between the decline of commercial fish species in parallel with seagrass decline in Western Port, Victoria (Connolly \textit{et al.}, 1999).
primary contact level (swimming) to a secondary contact level (boating). The visual amenity may also be reduced from more turbid water and loss of floral components of the wetlands. These changes in these values could be measured by determining changes in travel costs to recreational sites. To estimate the value of all these benefits and attribute them to the biogeochemical functioning of the wetland would lead to double counting. However, the change in each value (in this case a loss) can be added as each is impacted upon by the change in wetland structure.

Indirect values are more difficult to evaluate fully, particularly those that occur off-site. One method for estimating the water quality improvements that are provided by wetlands is to consider the cost of replacing those services using human made equipment. In Chapter 6 it was suggested that the cost of recent upgrades to sewage treatment in the Moreton Bay is an indication of this value, which was previously provided as a public good by the wetlands (MBCWQMST, 1998). The wetlands also help protect shoreline structures from wind and flood damage (Mitsch and Gosselink, 1993). The value of this service could be assessed by considering the cost of damage, which might be caused in the absence of wetlands. Hence, the value is imputed as the damage costs avoided.

Non-use values by their nature cannot be measured using market information and depend on the use of stated preference techniques for their estimation. The contingent valuation technique has been used in this thesis to assess if the technique can capture ecological value and the effect of information type on willingness to pay. The results from the survey of wetlands of Moreton Bay provide insight into the types of values held by respondents. However, as noted in Chapter 8 the provision of information may not have been the only motivation used by respondents to form their willingness to pay bids. The values respondents hold for the wetland may have been formed previous to the provision of information in the survey. Evidence for this suggestion comes from the significant difference in willingness to pay between those who had and had not visited the Bay.
Overall, problems facing economists are related to determining a value for a variety of goods and services that are interdependent on wetland functioning. Further, it can be concluded that before changes in economic value can be estimated those receiving the benefit must perceive that there has been a change in benefits. For example, if recreational activity associated with wetlands does not change in response to deteriorating water quality it might be assumed the economic benefits provided by the wetlands have not changed sufficiently to be measured. Again, some of the complexity can be reduced by using indicators that convey the direction and magnitude of changes in economic benefits related to ecological changes. Economic indicators, such as changes in gross domestic product and employment which Norton (1995) points out are perhaps flawed, are already well accepted by the general public as conveying meaningful information. Given the difficulties in measuring environmental benefits and changes of those benefits an indicator needs to be developed that incorporates both those changes that can be measured and those that can only be qualitatively assessed, such as the global biogeochemical cycling of wetlands. However, an indicator will only be useful if it is based on accurate information. Thus, it must be ensured that if stated preference techniques are used to assess value that the willingness to pay bids provided are only based on the class of value under consideration by the researcher. Such an indicator must also only include those benefits that have been identified in Stage 1 and 2 as likely to arise from the ecosystem under consideration.

For example, if the indicator suggested in stage 2 is accepted, a measure needs to be derived that shows there has been a change in the economic benefits derived from the environment. As stated above, there will be difficulties in measuring the components of total economic value of environmental resources due to information shortages, limitations of techniques and the possibility of double counting. This does not mean that this notion should be discarded. Rather, changes in some of the benefits can be measured to indicate the change in full benefits. As for ecological assessment it is not possible to measure every variable in the system, but some key values must be selected that can be assessed over repeated periods. For example, the results of the contingent valuation study of Moreton Bay presented in Chapter 9 suggest that the most important
values held by respondents, in terms of willingness to pay are the use values of the wetlands which include fishing and recreational activities and indirect use values of water quality improvements. Thus, these values could form the focus of economic investigations to assess changes in the value of fish catches, recreational expenditures and the costs of wastewater treatment.

Also, it is expected that changes in one category of benefit will be reflected in others. If commercial fishery values are lost due to loss in the structure of seagrass, the habitat for endangered species will also be lost. For example, it has been noted in Chapter 6 that commercial species of prawns, crabs and fish are dependent on wetlands for part of their life cycle. Similarly, green turtles and dugongs also feed on the seagrasses. Therefore, it could be expected some non-use values such as the existence values associated with dugongs, will be lost in conjunction with direct use values. If an initial baseline study has been undertaken to determine the magnitude of non-use values, a scaling factor from the loss of fishery value may be applied to show the commensurate loss. Although it may never be possible to determine all the economic values produced by an ecosystem, it should be possible to develop an index that reflects the major components of value for the population under consideration. It is then necessary to try to use available information and repeatable studies to demonstrate how each component of value has changed.

In Moreton Bay, a change in fishery values can be assessed through data collected by the fishery agencies, which include catch and effort in their calculations. From a recreational perspective, indicators of change in benefits may able to be derived from the number of boat registrations in the area, boat ramp counts or data collected on visitation rates. Changes in indirect use values such as water quality improvements may be indicated by changes in investment in stormwater and sewage treatment undertaken to maintain water quality. As indicated above, non-use values may only be assessed through the use of stated preference methods. However, such studies will only provide

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8 If the law of diminishing marginal returns holds for non-use values, a reduction in the number of endangered species may actually make non-use value rise.
an assessment of the change for the ‘good’ under investigation in the particular study. Further changes to the ‘good’ would require further studies\textsuperscript{9}. These methods can be expensive and time-consuming to undertake. However, it is suggested that if the provision of these values is dependent on certain functions, such as dependent habitat, a loss in this habitat can be related to changes in non-use value. As with all indices the most complex task it how to weigh and integrate the various components of values. In this respect, an initial estimation of as many value components as possible is essential to provide baseline data and the contribution of each component to total economic value. Once again, as stated in Chapter 8, the caveat for using stated preference methods to determine economic value is that it must be clear which components of value are being assessed and to whom the benefits will flow.

10.6 STAGE 5: REVIEW AND REASSESSMENT OF INDICATORS AND PROBLEMS/MANAGEMENT OBJECTIVES

Following the assessment of both ecological and economic changes in an ecosystem ecologists and economists must again engage in the exchange of information with reference to purpose of the investigation and the co-development of indicators. The questions that need to be addressed at this stage will include: Was the scale of assessment appropriate for both disciplines? Were the changes in ecosystem condition expected? How were these changes able to be related to change in ecosystem outputs? Were the changes in economic value of the expected direction and magnitude? Are there any lags between changes in ecosystem condition and economic benefits? Does the current rate of change in the ecosystem suggest that the current level of benefits will be maintained? Have all the economic benefits identified for the ecosystem been included in the ecological assessment tasks? What further monitoring, assessment and valuation is necessary to answer the questions raised? How does the assessment provide communicable information that may help solve the problem/management objective?

It is unlikely that each of these questions will be able to be resolved. Rather the

\textsuperscript{9} Choice modelling may be the best stated preference method for this purpose as it allows a value function to be estimated and changes in ecological function can be inserted into the value function.
methodology should be iterative. The purpose of the assessment may need to be reassessed in the light of new information, threats or management objectives. Indicators may need to be refined to reflect new information or clarify issues in communication between the two disciplines. The range of ecological monitoring tools may need to be extended in the face of new stresses on the system. Alternatively, the intensity of monitoring may need to be altered in the face of seasonal or other trends that have been detected. Alternative, economic valuation techniques may need to be employed to explore in further detail changes in the class of values which now appear to be more important or threatened due to greater information about linkages within the system.

10.7 CONCLUSION

The approach suggested to link ecological and economic values is a multi-directional one. There is not a simple linear link between each ecosystem function, output and good and service. However, barriers to a truly multi-directional approach will continue until further progress is made by ecologists in the understanding of the benefits provided by the outputs of the system for humans and impacts of humans on the system. Similarly, economists must increase their understanding of the ways in which humans are dependent upon the environment and the consequence of human impacts upon the environment. However, these are not tasks that can be achieved solely within the individual disciplines. As noted information must flow across the intangible ecological economic interface so that both disciplines can contribute to the understanding of the functions, outputs and values of ecosystem that provide for instrumental human benefits. The methodology developed helps to develop links between the ecological and economic system by ensuring that ecologists and economist share a common perspective to the issue at hand and communicate their findings to each other and decision makers in a form that is understandable and accurate. These are the most critical steps in developing linkages between the two systems. Ecological information cannot be collected and overlaid with economic information is the hope that the two mesh. Thus, even if full information was available for both disciplines without a common perspective it will not link ecological and economic value. This poses a range of challenging opportunities for further research.
CHAPTER 11
CONCLUSIONS

This final chapter outlines the major findings of the thesis. The major question addressed in the thesis has been how ecological and economic values can be linked. This question can be answered by considering the answers to the ancillary questions that have been addressed throughout the thesis. The approach taken to link ecological and economic values is also critically assessed and alternative approaches proposed. Finally, future research needs for linking ecology and economics are suggested.

11.1 MEASURING ECOLOGICAL VALUE FROM AN ECOLOGICAL PERSPECTIVE

The aim of this thesis is to investigate the linkages between the ecological values and economic values of environmental goods, using wetlands as an example. A framework was introduced which demonstrated how ecological and economic values could be linked through the ecological-economic interface. This framework was then applied to coastal wetland ecosystems. This framework demonstrated that the underlying ecological values of wetlands provided for secondary economic benefits. However, approaches to link wetland functions to economic services at the ecological-economic interface have been limited. A major impediment for applying direct linkage models is the lack of relevant ecological information. For example, it is known that some fish are dependent on wetlands for part of their life cycle, yet it is unknown what portion of the value of fish can be attributed to the wetlands. However, it was argued that the interaction of the structures, process and functions of an ecosystem produces a set of attributes that could then be linked to economic value if they could be defined.

In order to determine the attributes that constitute ecological value it was necessary to define value from the perspective of the discipline of ecology. However, there was no clear consensus in the literature on the definition and measurement of ecological value. This deficiency has been addressed by asking the question of how ecological value can
be defined and assessed. The approach taken was to use an assigned system of value so that ecological value could be considered from the perspective of those attributes that provided instrumental human benefits.

From this perspective a definition of ecological value was proposed that considered that the ecological value of a system can be determined by its productivity, ability to provide habitat for dependent species and the diversity and organisation of species it supports. That is, these are the attributes of an ecosystem which ultimately provide for human needs. This definition provides an advance in the field of ecology as a single notion of value has been lacking to date. From the definition of ecological value proposed an index of ecological value was constructed which included considerations of the appropriate temporal and spatial scale. However, the ecological knowledge required to understand the interconnections between the attributes of ecological value was lacking. This limited the ability of operationalising the index, as the functional form of the index could not be identified. Further, although ecologists measure and monitor various aspects of ecosystems, information has not been gathered on the attributes of ecological value.

Thus, the research was able to provide an important contribution to the ecological-economics debate by defining and proposing a measurement of ecological value that provided economic benefits. The conclusions that can be drawn from these investigations are that the links between humans and nature need to be expanded within ecology so that the information collected is relevant to the measurement of ecological value. Similarly, economists must be aware of all the goods and services provided by ecosystems when assessing their economic value. This situation will only occur if ecologists and economists can work together on identified problems. That is, it is not sufficient for each discipline to approach a problem only from their own perspective if the information is be truly integrated for decision purposes. To overcome the problem of the lack of relevant ecological information to measure ecological value in this study, the possibility of measuring ecological value from an economic perspective was investigated.
11.2 MEASURING ECOLOGICAL VALUE FROM AN ECONOMIC PERSPECTIVE

Before attempting to measure ecological value from an economic perspective the question of the ability of economic techniques to capture this value had to be addressed. It was considered that all non-market techniques are theoretically able to capture this value. Therefore, four criteria were introduced to determine the techniques that were the most appropriate for the task. This approach for assessing the ability of economic techniques has not previously been extensively undertaken and therefore the conclusions drawn provide an advance in the valuation literature. The major interest in environmental valuation is the ability to measure the value of changes in a resource. In order to measure ecological value a change in the value of the good must reflect changes in ecological value. Revealed preference models were limited in the ability to capture this change although repeated studies may overcome the problem. Another overriding issue was the ability of economic techniques to measure use and non-use economic values, as all values are dependent upon ecological value. Stated preference techniques, which do not rely on linkages to markets, have been shown to have the ability to measure all components of total economic value. It was thus determined that stated preference techniques could theoretically measure ecological value if respondents to a survey were informed about these values. Having determined that it was possible to measure ecological value from an economic perspective the issue of providing information within a contingent valuation survey was then considered.

The amount and type of information to provide to respondents is an active area in environmental and resource economics. Previous studies have assessed the impact of information with respect to services provided by wetlands and ecological goods among other things. However, there are no exogenous criteria about the amount and type of information to provide. The review of the literature noted that the provision of extra information impacted on willingness to pay yet no studies were found which investigated the impacts of different types of information. Therefore, as an adjunct to
the main question of the ability of the technique to measure ecological value the
question of assessing the impact of differing types of information was also addressed.

In order to assess these two questions a contingent valuation survey was undertaken for
the wetlands of Moreton Bay, Australia. Moreton Bay was chosen as the case study area
as the importance of the wetlands has been identified internationally as a Ramsar site
and at the national and state level. However, there have been no previous studies
undertaken to assess the full range of economic values provided by the Bay. Nor, have
any studies been undertaken in eastern Australia on the economic value of any coastal
wetland. The study thus provides an important contribution in understanding the value
of sub-tropical coastal wetlands in a developed nation. Although, full information was
not available on all the attributes of ecological value it was possible to develop an
information set from the scientific literature that illustrated the ecological values of the
wetlands. Information on the economic values of the Bay was also limited but it was
possible to identify the major components of economic value that are provided by the
wetlands of the Bay.

Analysis of the results indicated that the provision of information did have an effect on
willingness to pay. There was an extra information effect between no information and
use value information in accord with previous studies. The study was also able to
demonstrate that there was an information type effect between use and non-use values.
This is an important contribution to the contingent valuation information literature.

As respondents were willing to pay for wetland protection in each of the four cases it
suggests that each type of information prompted a positive response to pay for the
benefits listed. However, the provision of use information produced the highest mean
willingness to pay. In this study this may have resulted from the high number of
respondents who use the Bay for a range of activities. If the survey had been provided
to areas further afield from the Bay (and thus included more non-users) the payment for
non-use values may have been greater. Use benefits are the most tangible aspects of
total economic value and thus likely to be more relevant to respondents. However, it has
been argued throughout the thesis that to account only for the direct use benefits of wetlands may lead to an undervaluation of the resource. This leads to the conclusion that information on all the economic benefits provided by an ecosystem should be provided in contingent valuation surveys so that the full benefits of the system are captured in the assessment. To overcome the possibility of information overload the information presented will need to be summarised and simplified for respondents. The evidence from this study suggests that respondents were able to assimilate the information presented which was derived from academic sources. Further work will be required to determine at what point the information load becomes too great for respondents to grasp.

The contingent valuation study was not able to demonstrate that respondents understand that the primary, ecological value of wetlands provide secondary, economic benefits. Although respondents were willing to pay for wetland protection when provided with ecological information there was no difference in the amount offered compared to the other cases. The information provided appears to have been interpreted to represent the underlying existence value of the system rather than the underlying ecological value as was intended. As there was no difference in willingness to pay from the other cases it cannot be concluded that contingent valuation is able to fully assess the total underlying ecosystem values. To do so would require that the link between the two sets of value were fully explained to respondents before they formed their willingness to pay bids. However, if these links are not fully understood from academic research then the researcher may be introducing inaccurate information into the survey. This does mean that respondents should not be alerted to ecosystem values. Rather it suggests that both ecological and economic analysis of the issue at hand is required as an input into the decision making process.

An approach for undertaking an integrated study of problem/issue that had both an ecological and economic component has been developed. The approach requires by necessity that practitioners of both disciplines are willing to engage in common dialogue. This in turn entails agreeing that a problem or management objective exists
and defining a scale of study that can include the full spatial and temporal extent of the ecological and economic values. To enhance communication of complex scientific and economic information it is suggested that indicators be developed which are appropriate to the task. Monitoring of ecological and economic change will then continue independently but the information found should be shared between disciplines to ensure that the project objectives continue to be met. Within the final stage of the approach, the results should demonstrate how a change in ecological value will be reflected in a change in economic value. However, the approach should be iterative to incorporate new information and resolve possible problems with communication, scale and monitoring effort that may help in providing better answers to the issue/management objective under consideration. An application of this approach that would allow the current ecological health monitoring program in Moreton Bay to be linked to the changes in economic value of the Bay has been provided.

The conclusions that have been drawn from the research to answer the ancillary questions support the answer to the main thesis question. Put simply, ecological and economic values can only be linked through relevant information from ecologists and economists. From an economic perspective, ecological information is required about the functions and attributes of systems that provide instrumental human benefits. That is, ecologists need to identify those aspects of ecosystems which ultimately result in the goods and services on which humanity depends. The challenge for economists would then be to find methods that can fully account for the societal benefits provided by the ecosystems. The present economic approach of measuring the value of systems based on human expressions of relative exchange values cannot be assumed to account fully for these values. This was demonstrated in this study of the willingness to pay for wetland protection. When provided with information about ecological value the response was not different from when information was given about other private benefits.

An alternative approach is for ecologists to identify the attributes of a system that are considered valuable from an ecological perspective. That is, ecologists need to engage
in the discourse of values and the making of judgements. An ecological perspective may be able to include further types of value such as intrinsic values. To avoid making these judgements on the basis of insufficient knowledge ignores the fact that decisions to alter ecosystems and wetlands in particular are occurring constantly. For example, in Moreton Bay excess sediments and nutrients flow into the wetlands as a result of current management practices within the catchment. Ecologists are beginning to assess the impacts of these actions on the wetland ecosystems of the Bay, but these assessments should also be linked to the changes in economic benefits that are likely to occur.

11.3 CRITICAL ASSESSMENT OF APPROACH TAKEN AND POSSIBLE ALTERNATIVE APPROACHES

The approach taken to link the ecological and economic systems has been to identify the values that can be attributed to each system and how they may be related. Value is a well defined concept in economics and at the beginning of the research project it was expected that ecology would have a similarly defined concept of value. As has been shown this is not the case. This deficiency has been addressed by using the economic concept of value to determine ecological value. However, given the difficulty in defining the attributes of ecological value and the lack of accumulated information on the measurement and interaction of these attributes the approach has been limited. This is not to say that the approach is without merit but that further research is required before an index of ecological value can become operational.

The economic approach of contingent valuation for valuing underlying ecosystem values in this thesis has also been limited. Economic values are based on expressions of human preferences. There is no reason to assume that these will correspond with the full societal benefits provided by ecosystems. The results of the contingent valuation survey where both use values and ecological value are given equal preference is testimony to this fact. Economic methods need to be developed that can account for the total economic benefits of a system while avoiding double counting.
An alternative approach for considering the linkages between values is to consider directly the relationship between function and ecological services. That is, scientists must determine the relationships between the properties of wetlands and their outputs. Wetland scientists have begun this task through the development of a functional analysis of wetlands. For example, it is suggested by Maltby (1998) that these assessments can provide social scientists with predictions on how wetlands actually work and recognise the way in which functions, services and attributes may be modified by human actions.

This approach has been endorsed at an international level with the launch of the Millennium Ecosystem Assessment (MA) project in June, 2001\(^1\). The MA project is being undertaken by the United Nations in conjunction with scientific groups, governments, foundations and other international agencies. The aim of the project is to undertake an integrated ecosystem assessment to analyse the capacity of ecosystems to provide goods and services important for human development. The assessment thus includes ecological and economic components. The rationale for the assessment is that wise decisions can only be made by looking at the entire array of goods and services provided by ecosystems and the interlinkages among them. Previous sectorial approaches are considered inadequate because they focus on one resource without considering the trade offs with other goods and services provided by the ecosystem. The MA project is being run over four years and will draw on the work of over 1,500 scientists. It is hoped that this project will provide the knowledge that will allow the values of ecosystems to be more fully linked with economic goods and services. By taking a more holistic approach to the entire range of goods and services provided by ecosystems the trade-offs between benefits should become more transparent to decision makers.

To achieve the aims of the MA (or similar) project, further research is required to determine the most appropriate methods for estimating the economic value of goods and services and other economic costs and benefits. Research is also required to understand non-economic costs and benefits such as resilience of human communities and food

\(^1\) http://www.ma-secretariat.org
security. Further, a methodology needs to be developed that can assess multiple scales of ecological processes and multiple scales of decision-making. The mismatch of ecological and economic time and spatial scales in the past has been identified as a cause of ill-informed management decisions. For wetlands, the questions to be addressed are not just the scale and loss of resources but also defining major wetland functions that have economic relevance. For example, wetland functions related to the provision of drinking water supply and reduction in floods and storm damage need to be assessed within the global water cycle, and within regional, national and local water regimes. Similarly, assessment is required on the functions of wetlands that provide the required habitat for a host of different species at different stages of their life cycle that in turn provide economic benefits.

11.4 CONCLUDING REMARK

This thesis has examined the relationship between the underlying ecological values of wetlands and the economic values they produce. It is clear that the economic values attributed to wetlands are dependent on the underlying ecological values. However, it is still unknown how changes in the underlying values will affect the goods and services produced. It is hoped that the research agenda of the sort set out by Millennium Ecosystem Assessment Project will inspire collaboration between ecologists and economists. Through an understanding of the full gamut of services provided by wetlands at local, regional and global scale decision makers will able to make the required trade offs to sustain these ecosystems in an integrated fashion.
APPENDIX A

CRITERIA USED FOR THE IDENTIFICATION OF MARINE AND WETLAND PROTECTED AREAS.

The purpose of this appendix is to present in tabular form the sources of information used to derive Table 3.1 – Criteria used for the identification of Marine and Wetland Protected Areas.

Table A1 Criteria for the selection of marine protected areas and wetlands of national or international importance

<table>
<thead>
<tr>
<th>CRITERIA</th>
<th>MARINE PROTECTED AREAS</th>
<th>WETLANDS</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>NRSMPA</td>
<td>IUCN</td>
</tr>
<tr>
<td>Representativeness/ Biogeographic importance</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Naturalness</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Dependency - Important for vulnerable life stages - nursery, juvenile, feeding, breeding, and resting areas</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Life support, essential ecological processes - or important ecological/hydrological role for wetland</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Diversity</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Rare and endangered species</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Uniqueness</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Productivity</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Vulnerability</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Integrity</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>International or national importance</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Supports 1% or more of the populations of any native plant or animal taxa/waterfowl</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Regularly supports 20 000 waterbirds</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Supports substantial numbers of individuals from groups of waterbirds</td>
<td>✓</td>
<td>✓</td>
</tr>
</tbody>
</table>
Sources and Key:


APPENDIX B
PILOT SURVEY INSTRUMENT

The purpose of this appendix is to reproduce the survey instrument presented to respondents for the pilot survey. Each page is reproduced in the same format as given to respondents. However, the survey instrument was double sided. The complete survey provided is for Case A where no extra information was presented. The information that was presented for each of the other cases is then given. Within the survey instrument this extra information was included after Question 6 and before the section “Threats to the Wetlands of Moreton Bay”.

Figure B1  Front cover of survey

A SURVEY OF YOUR OPINIONS

THE WETLANDS OF MORETON BAY

AUSTRALIAN SCHOOL OF ENVIRONMENTAL STUDIES
GRIFFITH UNIVERSITY NATHAN Q 4111
The Wetlands of Moreton Bay

The wetlands of Moreton Bay include areas of mangroves and seagrasses found along the foreshores and islands, and in shallow waters. These wetlands have been identified as being internationally important, especially for wader birds. These areas are highlighted on the map of Moreton Bay on the next page.

1. Do you agree or disagree with the statement that the wetlands of Moreton Bay should be protected? Please place a cross in the box next to the answer of your choice.

- [ ] Strongly disagree
- [ ] Slightly disagree
- [ ] Neither agree or disagree
- [ ] Slightly agree
- [ ] Strongly agree
WETLANDS OF MORETON BAY

Map supplied by QPWS
2. This question lists some of the reasons why some people believe it is important to protect the wetlands of Moreton Bay. Please circle the number that shows how important (if at all) each of these reasons is to you for the protection of wetlands in Moreton Bay.

<table>
<thead>
<tr>
<th>Statement</th>
<th>Not at all Important</th>
<th>Not Very Important</th>
<th>Neither Important or Unimportant</th>
<th>Very Important</th>
<th>Extremely Important</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wetlands help maintain the health of Moreton Bay</td>
<td>1</td>
<td>2</td>
<td>3</td>
<td>4</td>
<td>5</td>
</tr>
<tr>
<td>Wetlands provide opportunities for recreational fishing, bird watching and pleasant scenery</td>
<td>1</td>
<td>2</td>
<td>3</td>
<td>4</td>
<td>5</td>
</tr>
<tr>
<td>Wetlands help to clean the water and trap sediments and so improve water quality</td>
<td>1</td>
<td>2</td>
<td>3</td>
<td>4</td>
<td>5</td>
</tr>
<tr>
<td>Preserving wetlands will allow future generations of people to also enjoy them</td>
<td>1</td>
<td>2</td>
<td>3</td>
<td>4</td>
<td>5</td>
</tr>
<tr>
<td>The plants and animals that depend on the wetlands have a right to exist</td>
<td>1</td>
<td>2</td>
<td>3</td>
<td>4</td>
<td>5</td>
</tr>
</tbody>
</table>
3. Below are five statements that give reasons why some people believe it is important to protect the wetlands of Moreton Bay. Please allocate 100 points among these statements to reflect the relative importance you attach to each statement. The more points a statement receives, the more important that statement is to you. If you think the statement is not at all important give it zero points. If one statement is twice as important as some other statement, it should receive twice as many points. Read all five parts first, then answer each part. Together your five answers should total 100.

- Wetlands help maintain the health of Moreton Bay
- Wetlands provide opportunities for recreational fishing, bird watching and pleasant scenery
- Wetlands help to clean the water and trap sediments and so improve water quality
- Preserving wetlands will allow future generations of people to also enjoy them
- The plants and animals that depend on the wetlands have a right to exist

TOTAL 100
4. **Please indicate if you have undertaken any of the following activities in Moreton Bay (including foreshores and Bay islands) during the past twelve months.** *Place a cross in the box next to the answer that applies to you (you may cross more than one).*

- Visited any part of Moreton bay
- Been fishing on Moreton bay
- Been bird watching on Moreton bay
- Engaged in water based activities (eg. boating, diving or swimming)
- Other activities

5. **Please indicate how many times during the past twelve months you have undertaken the following activities (if any) in Moreton Bay (including foreshores and Bay islands)**

<table>
<thead>
<tr>
<th>Activity</th>
<th>Number of Times</th>
</tr>
</thead>
<tbody>
<tr>
<td>Visited any part of Moreton Bay</td>
<td>_______________</td>
</tr>
<tr>
<td>Been fishing on Moreton Bay</td>
<td>_______________</td>
</tr>
<tr>
<td>Been bird watching on Moreton Bay</td>
<td>_______________</td>
</tr>
<tr>
<td>Engaged in water based activities (eg. boating, diving or swimming)</td>
<td>_______________</td>
</tr>
<tr>
<td>Other activities</td>
<td>_______________</td>
</tr>
</tbody>
</table>

6. **During the past twelve months have you become aware of Moreton Bay (including foreshores and Bay islands) from any of the following sources?** *Place a cross in the box next to the answer that applies to you (you may cross more than one).*

- Read newspaper articles about Moreton Bay
- Heard radio programs about Moreton Bay
- Seen TV shows about Moreton Bay
- Heard of the *Health Waterways Campaign*
Threats to the Wetlands of Moreton Bay

The areas of marine vegetation in Moreton Bay are under threat due to the effects of low quality water coming from the rivers into the Bay. This poor quality water effects the mangroves and seagrasses which means that they can no longer function properly and in some cases results in their complete loss. To stop the loss of the wetlands the Local councils and State government have prepared a Waterways Management Plan. Some actions identified in the plan are being undertaken by Local and State governments. Other activities have not yet been undertaken due to a lack of funding.
7. A non-government Conservation Trust fund could be set up that specifically undertakes actions as identified by the Waterways Management Plan to improve water quality in Moreton Bay. If such a fund was set up to specifically undertake actions that would improve water quality and hence protect the wetlands of Moreton Bay, and keeping in mind your household budget and other areas of environmental spending, what would be the maximum amount you be willing to make as a once off contribution? Please write the amount in dollars in the space below.

$_________ If you were unable or unwilling to give a positive answer to this question, please go to question 7a. Otherwise, go to question 8.

7a Why were you unwilling to place a positive amount at Question 7?
Please place a cross in the box next to any of the reasons listed that apply to you (you may cross more than one).

☐ I did not have enough information to place a $ value on protecting the wetlands
☐ I did not want to place a dollar value on protecting wetlands
☐ I disagree with paying money into a fund
☐ The wetlands are worth nothing to me therefore I don’t want to pay
☐ The government should pay to protect the wetlands using taxes already paid
☐ I couldn’t afford to pay anything
☐ Other (please specify)__________________________

NOW GO TO QUESTION 8
The next few questions ask you some questions about yourself. This allows us to check that this survey represents the views of the larger population. Please remember that all answers are confidential and will only be used to compare the answers different people give.

8. **What sex are you?** *Please place a cross in the box next to your answer.*

   - Male
   - Female

9. **What age group do you belong to?** *Please place a cross in the box that applies to you.*

   - 15-24
   - 25-34
   - 35-44
   - 45-54
   - 55-64
   - 65-74
   - 75 and over
10. **What level of education have you obtained or are obtaining?** Please place a cross in the box that applies to you.

- Never attended school
- Left school between the ages of 14-19
- Basic vocational qualification
- Skilled vocational qualification
- Associate diploma
- Undergraduate diploma
- Post-bachelor degree
- Graduate diploma
- Higher degree
- Other – please specify ………………………………..

11. **From the list below, please indicate which group represents your usual gross total weekly or annual income (before tax)?** Please place a cross in the box next to the range that best describes your income.

<table>
<thead>
<tr>
<th>Weekly Income</th>
<th>Annual Income</th>
</tr>
</thead>
<tbody>
<tr>
<td>Less than $79 a week</td>
<td>Under $4159 a year</td>
</tr>
<tr>
<td>$80 - $159 a week</td>
<td>$4160 - $8319 a year</td>
</tr>
<tr>
<td>$160 - $299 a week</td>
<td>$8320 - $15599 a year</td>
</tr>
<tr>
<td>$300 - $499 a week</td>
<td>$15600 – $25999 a year</td>
</tr>
<tr>
<td>$500 - $699 a week</td>
<td>$26000 - $36399 a year</td>
</tr>
<tr>
<td>$700 - $999 a week</td>
<td>$36400 - $51999 a year</td>
</tr>
<tr>
<td>$1000 - $1499 a week</td>
<td>$52000 - $77999 a year</td>
</tr>
<tr>
<td>$1500 a week or greater</td>
<td>$78000 a year or greater</td>
</tr>
</tbody>
</table>

| Don’t Know             |                        |

12. **What is your postcode?** ________
13. Are you a member of any environmental group? (for example, the Wildlife Preservation Society, the Australian Conservation Foundation, the Australian Marine Conservation Society or a local environmental group) Please place a cross in the box next to your answer.

☐ Yes
☐ No

THANK YOU VERY MUCH

Thank you for taking the time and effort to complete this questionnaire. Your views are important. Please place the questionnaire in the stamped envelope provided and post as soon as possible. If you would like a copy of the study results please write your name and address on a separate piece of paper and enclose it with your questionnaire.

Do you have any other comments that might be important? If so, please feel free to use the back of this booklet to write any thoughts or comments.
For Cases B, C and D- the following information was inserted after Question 6 and before “Threats to the Wetlands of Moreton Bay”.

**INFORMATION INCLUDED FOR CASE B – ECOLOGICAL VALUE**

<table>
<thead>
<tr>
<th>The Values of the Wetlands in Moreton Bay</th>
</tr>
</thead>
</table>

The large numbers of plants in the wetlands capture energy from the sun and convert this energy to a form that can be used by animals. This productivity provides the basis for all further biological activity. The wetlands of Moreton Bay also provide areas for animals to feed, rest and reproduce. Many animals, including fish, return to particular areas for breeding and so local populations of these animals are dependent on the existence of the wetlands. Migratory wader birds also use the wetlands to feed and rest before their return journey to the Northern Hemisphere. Due to the fact that the wetlands are productive and provide places for animals to feed, breed and rest, they contain a wide range of different plants and animals. The interaction between the variety of plants and animals produces food and helps assimilate wastes. This in turn helps to maintain the health and integrity of the complex wetland system.

**INFORMATION INCLUDED FOR CASE C – USE INFORMATION**

<table>
<thead>
<tr>
<th>The Values of the Wetlands in Moreton Bay</th>
</tr>
</thead>
</table>

The areas of Moreton Bay sheltered by wetlands provide safe places for recreational fishing and boating. In fact, over three-quarters of fish caught by the 300,000 fishers who use Moreton Bay each year depend on the wetlands in some way. The sheltered waters also provide for a range of water related recreational activities such as sailing, motor boating, water skiing and diving. Others enjoy visiting the foreshores to view the wildlife such as the 50,000 migratory wader birds that visit the Bay each year. The visual character of the wetlands also adds to the pleasant scenery provided by the Bay. Also, as wetlands purify the water, it means that people can swim safely and eat the seafood they catch. However, in some parts of the Bay wetlands have been degraded and destroyed and they can no longer provide this service.
The wetlands provide a place to live for a number of threatened and endangered species such as turtles and dugongs. Sea turtles are air breathing marine reptiles. All the turtles in Moreton Bay are directly or indirectly dependent on the wetlands. For instance, green turtles feed directly on the seagrasses, which form part of the wetlands. Turtles may take between 20 – 50 years to reach sexual maturity. Dugongs (also known as sea cows) are harmless marine mammals. They only feed on plants such as seagrasses. Dugongs have a low reproduction rate, only producing a single calf every 3-7 years. As both turtle and dugong populations only increase very slowly, loss of adults can result in chronic declines in their population. Both these animals are threatened by habitat destruction, such as the loss of wetlands in Moreton Bay. These protected animals may not provide direct benefits to humans but their continued existence relies on the habitat provided by the wetlands in the Bay.
APPENDIX C

TEXT OF LETTERS AND POSTCARD

The purpose of this appendix is to present the full text of the letters and a postcard sent to respondents of a contingent valuation survey for the protection of wetlands in Moreton Bay. An initial covering letter was sent. This was followed a week later with a postcard. For those respondents who had not responded to the first two contacts a follow up letter (and additional survey instrument) was then sent. The wording for the letters and postcard were the same for both the pilot and main study. The initial letter is presented followed by the postcard and then the follow up letter. It should be noted that the official Griffith University letterhead was used for the letters. Also, as noted the author personally signed all letters and postcards.

C1 COVERING LETTER – INITIAL CONTACT

Dear

I am a student at Griffith University studying Moreton Bay. As part of my research, I am conducting a survey about people’s attitudes to the wetlands of Moreton Bay.

You are one of a small group of people in south-east Queensland who have been selected at random (from the electoral roll) to give your opinion about the protection of the wetlands of Moreton Bay. In order that the results of this survey represent the thinking of the people of south-east Queensland it is important that each questionnaire be completed and returned. Of course, your participation in the survey is entirely voluntary. Your views are considered very important and the success of the research depends on you filling out the questionnaire.

The information provided by you will be kept strictly confidential and will be used only for the overall statistical results. The questionnaire has an identification number for mailing purposes only. This is so we can check your name off the mailing list when your questionnaire is returned. Your name will never be placed on the questionnaire. I would be most happy to answer any questions that you might have. Please write or call. My telephone number is 07 3348 6672.

The results of this research will be made available to those agencies working to improve the environmental condition of Moreton Bay. You may receive a summary of the results by placing a piece of paper with your name and address on it in the envelope with the survey. Please do not put this information on the questionnaire itself.
The questionnaire should only take about 10 minutes to complete. When you have completed the questionnaire, please return it in the enclosed return addressed and stamped envelope. I thank you in anticipation of your co-operation and look forward to receiving your completed questionnaire as soon as possible.

Yours sincerely,

C2 POSTCARD AND WORDING – ALL RESPONDENTS

Figure C1 Front of postcard sent to all survey respondents.

On the reverse side of the postcard the following was written:

Last week a questionnaire seeking your opinions about the protection of wetlands in Moreton Bay was mailed to you. Your name was selected at random from the electoral rolls for south-east Queensland.

If you have already completed and returned the questionnaire to me, please accept my sincere thanks. If not, please do so today. Because it has been sent to only a small, but representative, sample of south-east Queensland residents it is extremely important that yours also be included in the study if the results are to accurately represent the opinions of south-east Queensland residents.

If by some chance you did not receive the questionnaire, or it has been misplaced, please call me on (07) 3348 6672 and I will get another one in the mail to you today.

Yours sincerely,

Beth Clouston.
C3  COVERING LETTER – FOLLOW UP CONTACT

Dear

About three weeks ago I sent you a survey about the protection of the wetlands of Moreton Bay. To the best of my knowledge, the survey has not yet been returned.

The comments of people who have already returned the survey indicate a wide range of views on the protection of the wetlands of Moreton Bay. Many have indicated that they use Moreton Bay in a variety of ways. I think that the results will be useful in developing strategies to improve water quality in the Bay.

I am writing to you again to stress the importance that your questionnaire has for helping to get accurate results. Although I sent questionnaires to a number of residents of south-east Queensland, it is only by hearing from nearly everyone in the sample that I can be sure that the results are truly representative.

I would like to remind you that the questionnaire has an identification number written on the back cover so that I can check your name off the mailing list when it is returned. The list of names is then destroyed so that it can never be connected to the results in any way. Protecting the confidentiality of people’s answers is very important to me, and to the university.

I hope that you will fill out and return the questionnaire soon, but if for any reason you prefer not to answer it, please let me know by returning a note or the blank questionnaire in the enclosed stamped envelope.

Yours sincerely,

Beth Clouston.

P.S. If you have any questions please feel free to phone me on (07) 3348 6672.
APPENDIX D
MAIN SURVEY INSTRUMENT

The purpose of this appendix is to reproduce the survey instrument presented to respondents for the main survey. Each page is reproduced in the same format as given to respondents. However, the survey instrument was double sided. The complete survey provided is for Case A where no extra information was presented. The information that was presented for each of the other cases is then given. Within the survey instrument this extra information was included after Question 7 and before the section “Threats to the Wetlands of Moreton Bay”.

Figure D1  Front cover of survey

A SURVEY OF YOUR OPINIONS

THE WETLANDS OF MORETON BAY

AUSTRALIAN SCHOOL OF ENVIRONMENTAL STUDIES
GRIFFITH UNIVERSITY NATHAN Q 4111
The Wetlands of Moreton Bay

The wetlands of Moreton Bay include areas of mangroves and seagrasses found along the foreshores and islands, and in shallow waters. These wetlands have been identified as being internationally important, especially for wader birds. These areas are highlighted on the map of Moreton Bay on the next page.

Your views on the protection of wetlands in Moreton Bay.

1. Do you agree or disagree with the statement that the wetlands of Moreton Bay should be protected? Please place a cross in the box next to the answer of your choice.

- Strongly disagree
- Slightly disagree
- Neither agree or disagree
- Slightly agree
- Strongly agree
WETLANDS OF MORETON BAY

Data Source: Hyland et al 1989;
Hyland & Butler 1988
Map supplied by QPWS
2. This question lists some of the reasons why some people believe it is important to protect the wetlands of Moreton Bay. Please circle the number that shows how important (if at all) each of these reasons is to you for the protection of wetlands in Moreton Bay.

<table>
<thead>
<tr>
<th>Statement</th>
<th>Not at all Important</th>
<th>Not Very Important</th>
<th>Neither Important or Unimportant</th>
<th>Very Important</th>
<th>Extremely Important</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wetlands help maintain the health of Moreton Bay</td>
<td>1</td>
<td>2</td>
<td>3</td>
<td>4</td>
<td>5</td>
</tr>
<tr>
<td>Wetlands provide opportunities for recreational fishing, bird watching and pleasant scenery</td>
<td>1</td>
<td>2</td>
<td>3</td>
<td>4</td>
<td>5</td>
</tr>
<tr>
<td>Wetlands help to clean the water and trap sediments and so improve water quality</td>
<td>1</td>
<td>2</td>
<td>3</td>
<td>4</td>
<td>5</td>
</tr>
<tr>
<td>Preserving wetlands will allow future generations of people to also enjoy them</td>
<td>1</td>
<td>2</td>
<td>3</td>
<td>4</td>
<td>5</td>
</tr>
<tr>
<td>The plants and animals that depend on the wetlands have a right to exist</td>
<td>1</td>
<td>2</td>
<td>3</td>
<td>4</td>
<td>5</td>
</tr>
</tbody>
</table>
3. Below are five statements that give reasons why some people believe it is important to protect the wetlands of Moreton Bay. Please allocate 100 points among these statements to reflect the relative importance you attach to each statement. The more points a statement receives, the more important that statement is to you. If you think the statement is not at all important give it zero points. If one statement is twice as important as some other statement, it should receive twice as many points. Read all five parts first, then answer each part. Together your five answers should total 100.

Wetlands help maintain the health of Moreton Bay

Wetlands provide opportunities for recreational fishing, bird watching and pleasant scenery

Wetlands help to clean the water and trap sediments and so improve water quality

Preserving wetlands will allow future generations of people to also enjoy them

The plants and animals that depend on the wetlands have a right to exist

TOTAL 100
4. Please indicate if you have undertaken any of the following activities in Moreton Bay (including foreshores and Bay islands) during the past twelve months. Place a cross in the box next to the answer that applies to you (you may cross more than one).

- Visited any part of Moreton bay
- Been fishing on Moreton bay
- Been bird watching on Moreton bay
- Engaged in water based activities (eg. boating, diving or swimming)
- Other activities

5. Please indicate how many times during the past twelve months you have undertaken the following activities (if any) in Moreton Bay (including foreshores and Bay islands)

<table>
<thead>
<tr>
<th>Activity</th>
<th>Number of Times</th>
</tr>
</thead>
<tbody>
<tr>
<td>Visited any part of Moreton Bay</td>
<td>________________</td>
</tr>
<tr>
<td>Been fishing on Moreton Bay</td>
<td>________________</td>
</tr>
<tr>
<td>Been bird watching on Moreton Bay</td>
<td>________________</td>
</tr>
<tr>
<td>Engaged in water based activities (eg. boating, diving or swimming)</td>
<td>________________</td>
</tr>
<tr>
<td>Other activities</td>
<td>________________</td>
</tr>
</tbody>
</table>
6. Do you think of yourself as living “at the Bay”? Please place a cross in the box next to your answer

☐ Yes
☐ No

7. During the past twelve months have you become aware of Moreton Bay (including foreshores and Bay islands) from any of the following sources? Place a cross in the box next to the answer that applies to you (you may cross more than one).

☐ Read newspaper articles about Moreton Bay
☐ Heard radio programs about Moreton Bay
☐ Seen TV shows about Moreton Bay
☐ Heard of the Healthy Waterways Campaign
Threats to the Wetlands of Moreton Bay

The areas of marine vegetation in Moreton Bay are under threat due to the effects of low quality water coming from the rivers into the Bay. This poor quality water effects the mangroves and seagrasses which means that they can no longer function properly and in some cases results in their complete loss. To stop the loss of the wetlands the Local councils and State government have prepared a Waterways Management Plan. Some actions identified in the plan are being undertaken by Local and State governments. Other activities have not yet been undertaken due to a lack of funding.
8. A non-government Conservation Trust fund could be set up that specifically undertakes actions as identified by the Waterways Management Plan to improve water quality in Moreton Bay. If such a fund was set up to specifically undertake actions that would improve water quality and hence protect the wetlands of Moreton Bay, and keeping in mind your household budget and other areas of environmental spending, would you be willing to make a once off contribution of $20? Please place a cross in the box next to the answer of your choice.

☐ 1 Yes Go to Question 9
☐ 2 NO GO TO QUESTION 10

9. Would you be willing to pay $50? Please place a cross in the box next to the answer of your choice

☐ 1 Yes Go to Question 11
☐ 2 No Go to Question 11

10. Would you be willing to pay $10? Please place a cross in the box next to the answer of your choice

☐ 1 Yes Go to question 11
☐ 2 NO GO TO QUESTION 11

11. What is the maximum amount you would be prepared to pay?

$____________(Write the highest dollar amount you would be prepared to pay)

If you were unwilling to pay any amount please go to Question 12, otherwise go to Question 13.
12. Why were you unwilling to offer to pay any amount? Please place a cross in the box next to any of the reasons listed that apply to you (you may cross more than one).

- I did not have enough information to place a $ value on protecting the wetlands
- I did not want to place a dollar value on protecting wetlands
- I disagree with paying money into a fund
- The wetlands are worth nothing to me therefore I don’t want to pay
- The government should transfer spending from other areas and pay to protect the wetlands
- I couldn’t afford to pay anything
- Other (please specify)_______________________

Information about yourself

The next few questions ask you some questions about yourself. This information is important as it allows us to check that this survey represents the views of the larger population. Please remember that all answers are confidential and will only be used to compare the answers different people give.

13. What sex are you? Please place a cross in the box next to your answer.

- Male
- Female

14. What age group do you belong to? Please place a cross in the box that applies to you.

- 15-24
- 25-34
- 35-44
- 45-54
- 55-64
- 65-74
- 75 and over
15. What is the highest level of education you have obtained or are obtaining?  
*Please place a cross in the box that applies to you.*

- Completed primary school only
- Completed Junior/Intermediate/Form4/Year10
- Completed secondary school/Form6/Year12
- Trade/Technical Certificate
- Diploma or Associate Diploma
- Bachelor Degree
- Post-graduate Degree
- Other – please specify……………………….

16. From the list below, please indicate which group represents your usual gross total weekly or annual income (before tax)?  *Please place a cross in the box next to the range that best describes your income.*

<table>
<thead>
<tr>
<th>Weekly Income</th>
<th>Annual Income</th>
</tr>
</thead>
<tbody>
<tr>
<td>Less than $79 a week</td>
<td>OR Under $4159 a year</td>
</tr>
<tr>
<td>$80 - $159 a week</td>
<td>OR $ 4160 - $ 8319 a year</td>
</tr>
<tr>
<td>$160 - $299 a week</td>
<td>OR $ 8320 - $15599 a year</td>
</tr>
<tr>
<td>$300 - $499 a week</td>
<td>OR $15600 – $25999 a year</td>
</tr>
<tr>
<td>$500 - $699 a week</td>
<td>OR $26000 - $36399 a year</td>
</tr>
<tr>
<td>$700 - $999 a week</td>
<td>OR $36400 - $51999 a year</td>
</tr>
<tr>
<td>$1000 - $1499 a week</td>
<td>OR $52000 - $77999 a year</td>
</tr>
<tr>
<td>$1500 a week or greater</td>
<td>OR $78000 a year or greater</td>
</tr>
</tbody>
</table>
- Don’t Know

17. What is your postcode? __________

18. Are you a member of any environmental group? (for example, the Wildlife Preservation Society, the Australian Conservation Foundation, the Australian Marine Conservation Society or a local environmental group)  *Please place a cross in the box next to your answer.*

- Yes
- No
On the back of the final page the following was written.

**THANK YOU VERY MUCH**

*Thank you for taking the time and effort to complete this questionnaire. Your views are important. Please place the questionnaire in the stamped envelope provided and post as soon as possible. If you would like a copy of the study results please write your name and address on a separate piece of paper and enclose it with your questionnaire.*
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**INFORMATION INCLUDED FOR CASE B – ECOLOGICAL VALUE**

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**INFORMATION INCLUDED FOR CASE C – USE INFORMATION**

The areas of Moreton Bay sheltered by wetlands provide safe places for recreational fishing and boating. In fact, over three-quarters of fish caught by the 300,000 fishers who use Moreton Bay each year depend on the wetlands in some way. The sheltered waters also provide for a range of water related recreational activities such as sailing, motor boating, water skiing and diving. Others enjoy visiting the foreshores to view the wildlife such as the 50,000 migratory wader birds that visit the Bay each year. The visual character of the wetlands also adds to the pleasant scenery provided by the Bay. Also, as wetlands purify the water, it means that people can swim safely and eat the seafood they catch. However, in some parts of the Bay wetlands have been degraded and destroyed and they can no longer provide this service.
The wetlands provide a place to live for a number of threatened and endangered species such as turtles and dugongs. Sea turtles are air breathing marine reptiles. All the turtles in Moreton Bay are directly or indirectly dependent on the wetlands. For instance, green turtles feed directly on the seagrasses, which form part of the wetlands. Turtles may take between 20 – 50 years to reach sexual maturity. Dugongs (also known as sea cows) are harmless marine mammals. They only feed on plants such as seagrasses. Dugongs have a low reproduction rate, only producing a single calf every 3-7 years. As both turtle and dugong populations only increase very slowly, loss of adults can result in chronic declines in their population. Both these animals are threatened by habitat destruction, such as the loss of wetlands in Moreton Bay. These protected animals may not provide direct benefits to humans but their continued existence relies on the habitat provided by the wetlands in the Bay.
BIBLIOGRAPHY


