

**Paper Title: Economics for Natural Resources Management: Bio-economic Modelling, Policy Threshold Analysis and Transaction Costs<sup>#</sup>**

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## **Abstract**

Environmental policy has assumed a high profile in Australia with policies being recently developed to target aspects of land degradation, forest management and climate change, among others. These policies are often based on relatively little information about the likely benefits to be generated, costs borne by stakeholders, or, the relative transaction costs of policy options to stakeholders and government. In this paper we address these issues via two case studies of wetland policy. A bio-economic model of wetlands on private lands informs the scale of the policy response. The concept of policy threshold analysis is then developed as a filter to take into account transaction costs in an information poor environment. The transaction costs incurred include policy development, implementation, and, resource reallocation. The degree of irreversibility and the notion of environmental impact thresholds also affect the choice and timing of alternative policy options. Timing of policy is therefore a function of quasi-option values – the value of postponing a decision to obtain more information.

**Keywords:** policy, risk, transaction costs, wetland policy

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## 1 Introduction

Recent environmental policies in Australia have targeted land and water degradation, water management, forest management, and climate change. The Australian Commonwealth government land degradation and climate change special programs alone have allocated over A\$4 billion dollars towards environmental management.<sup>1</sup> Relatively little economic analysis of the likely benefits and costs of action has been undertaken to underpin the design and implementation of most of these policies despite the scale of expenditure and the claims about the levels of benefits in press releases (Bennett 2001). For example, the National Action Plan for Salinity and Water Quality requires economic evaluation of actions identified under regional plans but is not based on a similar analysis of the overall policy. Therefore, the question arises as to how an economic analysis of the available policy goals and options should proceed. What tools are available to help decide on an appropriate mix and level of policies to address environmental issues?

In this paper we apply the findings from an application of one tool, bio-economic modelling, to assist in the development and demonstration of a second tool, policy threshold analysis. Bio-economic modelling can be used to help develop goals and target incentives in the policy development process. Specifically, bio-economic modelling provides information on the scale, nature and distribution of the potential benefits and costs of changes to environmental management. Inclusion of this information in the policy development process provides for a more rigorous assessment of public investments and improved policy targeting.

Achieving any potential benefits from changing wetland management as estimated within the bio-economic model requires a policy delivery mechanism to facilitate the changes to environmental management that are desired. However, policy mechanisms are not cost free. Rather, resources are consumed in overcoming the transaction costs in policy development and implementation, as well as by any resource reallocation processes. Hence, many policy options may not deliver a net benefit to the community once their transaction costs are taken into account. Policy threshold analysis is developed as a filter of potential policies to select 'best bet' options in the presence of uncertainty about transaction costs. It involves a comparison of the likely range of market and government transaction costs against the net benefits of the additional environmental outputs so produced by the implementation of a particular policy.<sup>2</sup>

The traditional approach to policy, involving development, implementation, review and refinement may be complicated by elements of risk, uncertainty, or ignorance, about the information input to either the bio-economic model or the policy threshold

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<sup>1</sup> For example, these programs include the devolved grants through National Heritage Trust (A\$2.5b), the National Action Plan for Salinity and Water Quality (A\$0.7b), water reallocation via purchase and compensation through the Living Murray Initiative (A\$0.6b), and, energy efficiency and green energy subsidy programs administered via the Australian Greenhouse Office (A\$0.9b).

<sup>2</sup> Note that transaction costs are estimated 'net' of any reduction in transaction costs that they may induce in markets or government activities, especially where policies are specifically directed towards reducing transaction costs (for example, policies targeted towards reducing 'red tape').

analysis.<sup>3</sup> For example, there is often uncertainty about how a particular species or ecosystem will react to a change in management. In particular, information is often lacking about the degree of irreversibility and the possibility of environmental impact thresholds inherent in the production of the environmental outputs desired and the impacts of alternative management actions. Sensitivity analysis can be used to assess potential range of outcomes caused by risk and uncertainty about the information included in bio-economic modelling. In this environment policy timing is, in part, a function of quasi-option values because the presence of thresholds and irreversibilities may alter the timing and strength of the desirable policy response.<sup>4</sup>

The motivation for this paper has been set out in this introduction. The remainder of the paper is divided into four sections. The conceptual framework is set out in the next section. A brief description of the concepts of bio-economic modelling is initially provided. The focus then switches to the linkages between the bio-economic model and policy identification and development. The impact of market and government inefficiencies inherent in alternative policy options (the transaction costs of alternative policies) are explored via the development of policy threshold analysis as a filter of ‘best bet’ policy options in an environment of incomplete and uncertain information about these costs. An assessment of the impact of the more extreme risk and uncertainty outcomes introduced by potential irreversibilities or environmental impact thresholds on the conceptual framework completes the section.

The bio-economic modelling and policy threshold concepts are demonstrated using two wetland case studies: the Upper South East (USE) of South Australia (SA); and, the Murrumbidgee River Floodplain (MRF) in New South Wales (NSW). The findings from the bio-economic model for each region are reported in Section 3 along with the main conclusions from the sensitivity analysis. These results are then used to identify a set of potential policies in each case study area. The potential wetland policy options and existing wetland policies are subjected to a policy threshold analysis to identify the ‘best bet’ policies in the two case study regions. The policy implications of the conclusions from the case study applications are discussed in the fourth section along with the implications of the irreversibilities identified in the sensitivity analysis. A summary of the paper’s main findings and their implications for economic research concludes the paper.

## **2 Conceptual framework**

### **2.1 Bio-economic modelling**

Policy makers are faced with uncertainty about the scale of management change that is desirable and about the relative effectiveness of alternative policy options and packages when considering the design and adoption of environmental policy. Bio-economic modelling reduces this uncertainty via a quantitative assessment of the change in the net benefits to the community that result from changes to environmental

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<sup>3</sup> A situation involving risk is defined as one in which the probability of different outcomes is known. Uncertainty is characterised by unknown probabilities. Ignorance is where both the outcome and probabilities are unknown.

<sup>4</sup> Quasi –option values are defined as the welfare gain that comes from delaying a decision to use a natural resource when there are uncertainties about the payoff of alternative uses of the resource, and when at least one of the uses is irreversible (Grafton, Pendleton and Nelson 2001).

management.<sup>5</sup> In doing this, the process reduces the uncertainty about the scale of management change that is appropriate. It should be noted that the bio-economic modelling in this paper is presented primarily as an input to the policy threshold analysis and therefore only a relatively cursory overview is provided.

The concept of cost-benefit analysis underlies the structure of the bio-economic model. That is, bio-economic modelling is explicitly an examination of economic efficiency. Despite the focus on economic efficiency, bio-economic modelling can supply important information about the interplay between economic efficiency and other goals. For example, the process of bio-economic modelling generates information about the distributional impacts of alternative policy options that is useful for judging equity considerations. Furthermore, it explicitly involves the incorporation of non-market environmental benefits and costs into the modelling of net benefits.

Bio-economic modelling is a three-stage process:

1. Biophysical modelling – estimating changes in the biophysical status from changes to environmental management;
2. Economic modelling – estimating community values associated with alternative biophysical states; and,
3. Consolidation into a bio-economic model – estimating changes in community costs and benefits as a result of changes in the biophysical states.

Biophysical modelling is the compilation of the biological information underlying each element of the cost-benefit analysis. Hence, biophysical modelling has three main components:

1. The identification of the biological factors that drive peoples' values;
2. The estimation of the outcomes, in terms of changes to biological factors, under different landscape scale management strategies; and,
3. The estimation of the time and path of the biological factors for each of the potential outcomes of different landscape scale management strategies.

In practice, it is difficult to distinguish step two from step three. This is because all ecosystems are in a continual state of change and flux. Hence, outcomes will continue to change over time, with and without changes to management. Potential physical changes to environmental management practices (such as excluding introduced grazing animals from areas of remnant vegetation or wetlands) are also defined during the second and third steps of the biophysical modelling.

Both the biophysical and economic modelling steps are based on the concept of the margin. Each management strategy defined as part of the biophysical analysis involves a change to a relatively small proportion of total landuse. This relatively small proportion is referred to as the 'margin'.<sup>6</sup> Despite the relatively small area that

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<sup>5</sup> There are number of different types of bio-economic models, the model that is the focus in this paper is focused on estimating the net values generated by alternative resource management strategies.

<sup>6</sup> Definable impacts may occur beyond the area that has changed landuse, that is, beyond boundaries of changed landuse. These are 'externalities' of changes in land management and are also included in the analysis. The difficulty of defining appropriate limits to the analysis of changes introduces an element of uncertainty that should be taken into account in the policy development process.

changes use, it is posited to impact significantly on the costs and benefits of the system being examined.

Whereas biophysical modelling is the compilation and analysis of the biological factors that underlie peoples' values, economic modelling is the compilation and analysis of the economic information required for a cost-benefit analysis. Economic modelling is based on the theory of economic surpluses. An economic surplus occurs where either the producer or consumer receives a net benefit or cost as a result of changing management. The net benefit or cost estimated within the bio-economic model includes both monetary and non-monetary values, thus complicating the economic modelling process. While monetary values are relatively easily estimated within the market place, non-monetary values are more difficult to estimate. A variety of non-market valuation techniques were used to arrive at the estimates used in the case study applications.

Finally, bio-economic modelling involves the integration of the biophysical and economic modelling components. Specifically, bio-economic modelling facilitates the comparison of alternative biological states in terms of the net benefit that they would generate to society. Comparison is via the aggregation of economic costs and benefits for each of the alternative management strategies developed within the biophysical modelling phase. Cost-effective policies can then be targeted towards obtaining the highest net benefit outcome as estimated within the bio-economic model.

## **2.2 Policy identification**

A bio-economic model of the type described in Section 2.1 can thus assist in setting appropriate environmental outcome goals for policy development. The next logical step is identification of a range of policy options, and selection from these of cost effective policies to achieve the environmental goals. The components of bio-economic modelling provide significant input into identifying appropriate policy tools. However, considerable uncertainty remains about the relative cost-effectiveness of alternative policy options due to incomplete knowledge about market and government transaction costs. This uncertainty needs to be systematically addressed in order to select cost-effective policy options taking into account transaction costs. One approach to dealing with such uncertainty is the concept of policy threshold analysis.

In a bio-economic model, the beneficiaries and those who have costs imposed on them as a result of management changes are identified along with the relative scale and nature of these costs and benefits. This information can be used to target policy development towards affected parties and towards key constraints to management change (for example, costs of changing management) or opportunities (for example benefits from new or increase environmental outputs). The identification of these policy options broadly follows a search procedure developed by Weimer and Vining (1992), whereby the key constraints and opportunities are examined to identify whether any market or government failures are likely to exist. Specifically, theories of market and government failure are applied to the institutions that govern the key constraints and opportunities identified within the bio-economic model to identify policy opportunities (see Whitten and Bennett 2004 for a more complete discussion). The use of the bio-economic model to facilitate policy generation rather than to

simply evaluate pre-determined policy options (that may or may not target key costs and benefits) is an important strength of the approach.

### 2.3 Policy threshold analysis

The bio-economic model reported in this paper only includes estimates of the direct costs or benefits to wetland owners and the wider community of management changes. The model does not take into account the additional transaction costs that may be imposed (or in some cases reduced) by policies directed towards wetland management change. For example, if a policy is instituted to reduce the costs of fencing wetlands, only the direct cost of fencing is included in the bio-economic model, not the costs of redistribution or of applying for such assistance.

There are a number of indirect costs associated with the development and adoption of any policy as shown in Table 1 that must be considered before policies can be concluded to offer net benefits over the status quo. These indirect costs are not included within the bio-economic model for two reasons. First, there are a number of policies that may achieve any particular desired management change, each of which has different costs. For example, the costs to implement a tax rebate will differ from those associated with setting up a materials grant program or a subsidy payment program. Second, it would be prohibitive to estimate the transaction costs associated with a large number of policies, each of which may have several levels. Estimation costs are exacerbated by the lack of agreed methodologies for transaction cost estimation. Therefore, the cost-effectiveness of the policy options identified is subject to uncertainty.

**Table 1: Summary review of policy transaction costs**

Cost	Definition
<b>Government policy transaction costs</b>	
Design information	Defining the scale and scope of the problem to be addressed and gathering and analysing information about a potential policy instrument.
Enactment	Costs of legislation including deadweight costs of lobbying.
Implementation	Regulatory design and development of agency structures and protocols required for administering the policy instrument.
Administration	Ongoing costs including negotiation, contracting and government transfer costs.
Detection	Monitoring compliance and effectiveness together with evaluation of the policy instrument.
Prosecution	Enforcement of compliance through legal systems.
Risk	Includes risk of adverse outcomes due to failure in the policy instrument implementation process or due to unintended consequences.
<b>Market policy transaction costs</b>	
Direct costs	Direct fees or charges that are netted off government transaction costs.
Additional information	Cost of additional information beyond that required for direct compliance with the policy instrument.
Contracting	Costs of negotiation and contracting.
Detection and protection	Monitoring and enforcement costs of claiming, codifying or defending property rights or prosecution.
<b>Other</b>	
Dynamic attributes	Incentives for innovation in management to improve environmental outcomes, reduce costs and increase flexibility in the face of exogenous change.

**Sources:** McCann and Easter (1999, 2000); Thompson (1999); Russell and Powell (1999); Colby (1990); Falconer, Dupraz and Whitby (2001); and, Gangadharan (2000)

Categorising the nature of the intervention contemplated under alternative policy options can also inform the types of transaction costs that are likely to be present. Bromley (1997) divides policies into facilitative, inducive and coercive categories. Facilitative policies seek to improve the functioning of markets, inducive policies change the incentives in markets via taxes or subsidies (both monetary and non-monetary) but retain the benefits of markets, and, coercive policies compel wetland owners to change management via use of the government's police powers. A range of generic policies is divided into these classes in Table 2. For example, prohibition of wetland uses is a coercive strategy.

**Table 2: Range of potential policy instruments**

<b>Category</b>	<b>Instrument</b>
<b>Coerce</b>	Government production Prohibition of resource uses or specific actions Specified management actions or specified best management practices Duty of care performance standard
<b>Induce</b>	Output taxes and charges (applied to damaging outputs) Input taxes and charges (applied to damaging inputs) Input subsidies (lump sums and on-going) Liability law provisions
<b>Facilitate</b>	Create, clarify or allocate property rights (where linked to a fee these become inducive) Provision of information (including technical advice, measurement advice or requirements and certification schemes) Voluntary agreements

**Source:** Primarily adapted from Russell and Powell (1999). Additional input from Weimer and Vining (1992) and National Center for Environmental Economics (2001).

Facilitative policies seek to reduce (but cannot eliminate) market transaction costs. They are likely to rank highly in terms of cost-effectiveness but are less likely to achieve the policy target in isolation due to remaining market transaction costs (for example resulting from the public good nature of many wetland outputs). Inducive policies reduce market transaction costs but at the cost of introducing at least some government transaction costs, and, coercive policies may remove market inefficiencies but are subject to higher degree of government transaction costs. Inducive policies may generate the desired outcomes to the community, but due to the greater degree of government transaction costs that they introduce, may not be cost effective.

The suggested means of incorporating uncertainty about the costs of market and government inefficiencies into policy development is via *policy threshold analysis*. Policy threshold analysis is a comparison of the likely range of market and government transaction costs against the net benefits of the additional environmental outputs so produced. A threshold policy analysis would be conducted in a similar fashion to threshold value analysis in benefit-cost analysis. A threshold value analysis compares how large a benefit or cost would need to be to alter the conclusions of the analysis and assesses the likelihood of the threshold value being achieved. Threshold policy analysis asks how large the transaction costs involved in adopting a specific policy would need to be in order for that option to either not be cost effective, or be less cost effective than an alternative policy option. The

threshold cost is defined by the net benefit to society of achieving the desired environmental management change. The likelihood that the transaction costs of the policy option exceed the threshold is then judged, and the conclusions about the relative ranking of policy options or packages developed accordingly. Judgement is based on qualitative and quantitative information regarding the costs of any redistribution through tax mechanisms, the likely costs of any necessary program management, and, the likely transaction costs in the market place as listed in Table 1. Therefore, threshold policy analysis provides a consistent methodology for judging relative policy effectiveness even though it does not necessarily involve the quantitative estimation of alternative policies' transaction costs.

### **2.3.1 An example of how threshold policy analysis works**

Assume a policy maker is required to develop policies for conserving areas of native vegetation on private property and is deciding between purchasing remnant vegetation and incorporating them into reserves or paying current landowners to manage remnants towards specified outcomes. The policy maker knows:

- the costs of purchasing the remnants;
- the costs of their ongoing management; and,
- the likely average costs of private management from the bio-economic modelling process.

The policy maker will need to make judgements about factors for which their knowledge is incomplete, such as the costs of redistributing funds under each policy and the ongoing costs of monitoring if landowners are paid to change management. Performing a threshold policy analysis requires the policy maker to qualitatively assess the relative scales of the transaction costs. The relative costs of each policy are then compared against those of other policy options and the relative benefits generated in order to assess the implications for policy selection and implementation.

## **2.4 Impact of environmental thresholds or irreversibilities**

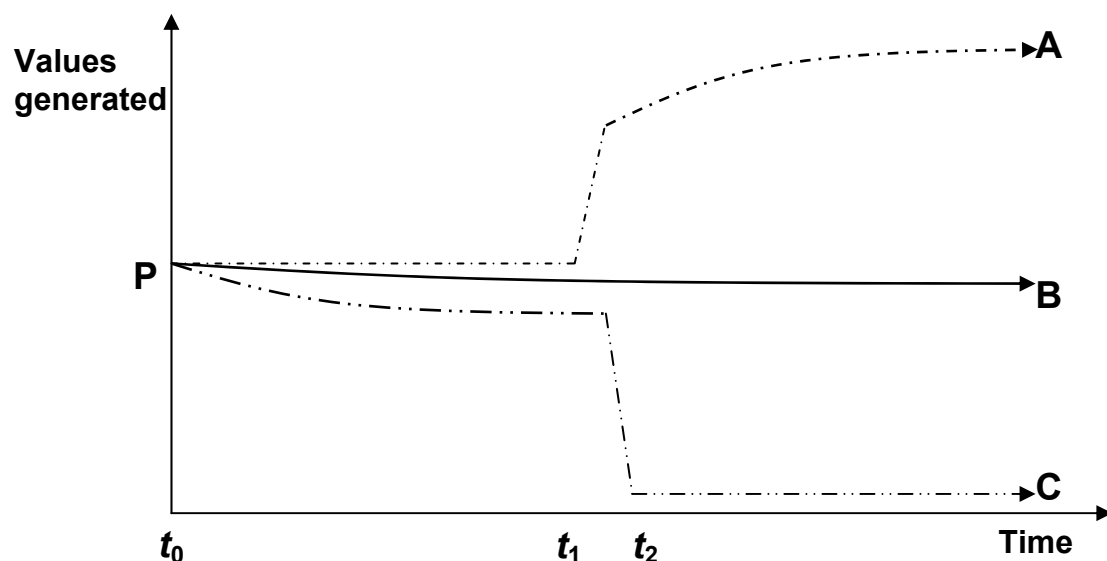
The management strategy estimated to generate the highest net benefit is not necessarily an optimal strategy because it remains subject to uncertainty. Uncertainty arises from incomplete knowledge about biophysical relationships, economic values and individuals' preferences. Uncertainty relates to both the outcomes of changing management and continuing business as usual. In some instances, the behaviour of the system being modelled may exhibit extreme forms of uncertainty as a result of aspects of irreversibility or environmental impact thresholds. Irreversibility occurs where a management strategy leads to irrecoverable loss of an attribute generating values (at least within a rational cost or time frame). For example, a management policy that leads to species extinction exhibits irreversibility. Environmental threshold impacts occur where a small change in management leads to, or prevents, a large change in the environmental outputs that are valued. For example, loss of wetlands beyond a certain point may lead to a dramatic fall in the population of native fish in a river system.

Irreversibility and environmental impact thresholds represent discontinuities in the bio-economic model. Their potential impact is shown in Figure 1. Pursuing management strategy 'A' may achieve a positive environmental impact threshold being reached at time  $t_1$ . Similarly, management strategy 'C' may lead to an

irreversible decline in values at time  $t_2$ . Strategy 'B' exhibits no such thresholds or irreversibilities.

The impact of these uncertainties on the values generated by alternative outcomes can be assessed via a sensitivity analysis of key parameters. The output from the sensitivity analysis can be used to identify the range of potential net benefit outcomes and which parameters have the greatest leverage on net benefit estimates. The distribution of outcomes within the range assessed by the sensitivity analysis usually remains in doubt because the probability associated with each outcome is uncertain. Sensitivity analysis thus provides information about the relative value of additional information about different parameters – a point returned to later in this paper.

**Figure 1: Irreversibility or environmental impact thresholds in a bio-economic model**



The impact of discontinuities in the bio-economic model complicates selection of appropriate policy options. A traditional approach to policy suggests that policies should be reviewed and adjusted over time where goals are not achieved. However, the penalty for failing to reach policy goals may be much larger in the presence of significant discontinuities in the bio-economic model. For example, in Figure 1 the penalty for failing to adopt 'A' may be 'C' rather than 'B' if irreversibility exists. Furthermore, the costs of acting now (at  $t_0$ ) to prevent the irreversible decline from occurring may be much lower than acting at sometime in the future, but before  $t_2$  is reached. Thus, the quasi-option value of postponing a decision in order to collect and process more information will also be lower than where irreversibility is not present. In this environment the policy maker may attempt to reduce risk by adopting additional policies or expanding policy options. That is, more and larger policies should be adopted sooner rather than postponing a decision and collecting more information (that may reduce the policy costs).

The adoption of additional policies or policy expansion in order to reduce the risk associated with failing to achieve policy goals imposes additional costs thus reducing the net benefit outcome. Hence, additional policies should only be considered where a net benefit would remain. Considering policies beyond this point is self-defeating

because the optimal strategy (that would generate the maximum net benefit) is to do nothing and allow the irreversible decline to occur.

A related problem may occur where the distribution of benefits from changing management exhibits a similar discontinuity. As an example, consider the case of potential species extinction where the endangered species is only present on a relatively small proportion of sites. A policy may successfully achieve the goal of changing a proportion of land management in the target area. However, if the policy was not successful in changing management where the endangered species was present (leading to extinction) the policy may fail to generate a net benefit to society. Hence, ignorance about individual values may cause policy failure even where there is little uncertainty about aggregate values.

### 3 Case study applications

#### 3.1 Case study areas

The policy discussion in this paper is illustrated by reference to two case study areas, the Upper South East (USE) region of SA and the Murrumbidgee River Floodplain (MRF) in NSW. The location of the case study areas is shown in Figure 2.

*Figure 2: Location of case study areas*



In the USE sixty-three thousand hectares of healthy wetlands, or less than seven percent of the original wetland area, remain in the region. The reduction in wetland area was motivated by the private values obtained from conversion pasture for agricultural production. Part of the remaining area of healthy wetlands is threatened by the impacts of dryland salinity. The reduction in wetland area has significantly reduced the private and social values generated by natural wetlands in the region. These values include drought refuge for waterbirds from southeastern Australia, bird-breeding events, landscape appearance, recreation, and, duck hunting. The issue is

whether the net benefit of current land use is less than that which other management practices could deliver.

In the MRF, relatively few wetlands have been drained, but many wetlands on the floodplain have been droughted while those closely linked with the river have been over-flooded as floodwater is stored and released for irrigated crop and pasture production. Wetlands in the MRF have also been degraded by logging, grazing, and to a lesser extent, irrigation drainage management practices. In the MRF, private values are generated from irrigation, grazing and timber production and divided between wetland owners (benefits resulting from grazing, logging and some irrigation) and irrigators downstream. The social values of wetlands have been lowered via reduced bird and fish breeding and reductions in water quality and wetland health. The issue is whether alternative wetland management could deliver a higher net benefit than present management strategies.

In both the USE and the MRF the community may wish to consider institutions and incentives through policy changes that would alter land and water management practices and may lead to increased net benefits to society as a whole.

## **3.2 Case study bio-economic models**

### **3.2.1 A bio-economic model of USE wetlands**

A literature review was initially undertaken to establish the nature of the values generated by the USE wetlands and the current wetland management strategies in the region (Whitten and Bennett 2004). Supplementary information was gathered via a survey of wetland owners and managers in the case study area (Whitten and Bennett 2000). The biological factors that drive these values were identified via the literature review and in consultation with scientists with expertise either in the region or in the types of physical relationships in the USE. The key biological factors driving values can be summarised as the area (and type) of healthy wetlands, and the area (and geographical relationship to wetlands and other remnant vegetation areas) of healthy remnant vegetation. Therefore, improved quality and increased quantity of wetlands and remnants and their spatial relationships to each other may increase the values generated by wetlands.

Three management options were developed to address the key biophysical factors:

1. Rehabilitate degraded wetland areas – termed ‘*wetland retention*’;
2. Re-create additional wetlands and linkages between wetlands to increase connectivity for wetland specific flora and fauna – termed ‘*pro-wetlands*’; and,
3. Rehabilitate additional remnant vegetation and re-create linkages between remnants and wetlands – termed ‘*wetlands and remnants*’.

These alternative management strategies are compared to what would occur without changes to management – termed the ‘business as usual’ (BAU) case. This structure was pursued because it makes little sense in the context of increasing wetland protection outputs to consider re-creating wetlands or rehabilitating and linking remnants without first rehabilitating degraded wetland areas. Additional strategies were rejected on the basis that they would not have a significant impact on the biological factors that drive wetland values, or their impacts were not sufficiently differentiated from one or more of the above set. A summary of the biophysical

modelling results is shown in Table 3. For example, changing wetland management to achieve the ‘wetlands and remnants’ option would involve reducing agricultural productivity by 257,000 dry sheep equivalents<sup>7</sup> (dse) but increase the area of healthy wetlands by 284 km<sup>2</sup>.

**Table 3: USE BAU and alternative resource allocation biophysical outcomes**

Descriptive attributes	Unit	BAU	Wetland retention	Pro-Wetlands	Wetlands and remnants
<b>Aggregate totals for USE</b>					
Agricultural pasture	km <sup>2</sup>	5,435	5,435	5,309	5,138
Healthy wetland	km <sup>2</sup>	442	569	695	726
Degraded wetland	km <sup>2</sup>	190	63	63	32
Healthy remnants	km <sup>2</sup>	513	513	513	1026
Degraded remnants	km <sup>2</sup>	342	342	342	0
Grazing productivity	dse	3,333,000	3,316,000	3,253,000	3,075,000
Waterbirds hunted	No.	6,300	9,300	11,100	11,600
Hunting trips	No.	1,100	1,700	2,000	2,100
Total tourist numbers	No.	6,350	18,250	41,500	41,500
<b>Marginal change from BAU</b>					
Agricultural pasture	km <sup>2</sup>	n.a.	0	-126	-297
Healthy wetland	km <sup>2</sup>	n.a.	127	253	284
Degraded wetland	km <sup>2</sup>	n.a.	-127	-127	-158
Healthy remnants	km <sup>2</sup>	n.a.	0	0	513
Degraded remnants	km <sup>2</sup>	n.a.	0	0	-342
Grazing productivity	dse	n.a.	-16,000	109,000	257,000
Waterbirds hunted	No.	n.a.	3,000	4,800	5,300
Additional hunting trips	No.	n.a.	600	900	1,000
Additional tourist numbers	No.	n.a.	11,900	26,150	35,150
Improved conservation status of species	No.	n.a.	15	24	35
Hydrological management works	km <sup>2</sup>	n.a.	63	190	205
Revegetation management works	km <sup>2</sup>	n.a.	0	98	269
Fencing required	km	n.a.	442	947	2289
Ongoing management – wetlands	km <sup>2</sup>	n.a.	126	253	284
Ongoing management – remnant veg.	km <sup>2</sup>	n.a.	0	0	513

**Note:** Conservation status of species is flora and vertebrate fauna species only.

The outcome of the economic modelling and consolidation into bio-economic modelling is shown in Table 4. For example, the cost of the lost agricultural production of the ‘wetlands and remnants’ option to wetland owners is \$13.4m while the environmental benefits amount to \$25.3m. However, when the additional costs of wetland management are considered (\$27.2m) the option would generate a net loss to the community of \$15.2m. Only ‘wetland retention’ generates a net benefit, or a potential Pareto improvement, to the community of just over \$5.5m.

An important aspect of bio-economic modelling is sensitivity analysis of the outcomes to a change in the key parameter values. The last four rows of Table 4 show part of the sensitivity analysis performed within the bio-economic model. The conclusions from the bio-economic model are sensitive to the extrapolation of the non-market benefits generated by wetlands. A less conservative extrapolation of the wetland non-market value generates a net benefit from all strategies. The results of the model are also sensitive to the number of endangered species that benefit. For example, if only half as many endangered species benefit then no management change

<sup>7</sup> Dry sheep equivalents are the standard measure of pasture productivity in much of Australia and are based on the amount of feed required to maintain a 50kg wether (castrated male sheep).

would generate a net benefit. Thus the value of more information about the likely benefits to endangered species may be high but postponing policy implementation decisions may lead to local extinction and the potential for higher costs if species reintroduction is considered.

**Table 4: NPV from adoption of alternative USE management strategies**

Cost or benefit	Wetland retention	Pro-wetlands	Wetlands and remnants
<b>Changes to agricultural activities</b>			
Pasture establishment and management costs saved	\$ 0	\$2,462,000	\$ 4,963,000
Cost of lost agricultural productivity	-\$1,166,000	-\$5,672,000	-\$18,332,000
<i>Sub-total</i>	<i>-\$1,166,000</i>	<i>-\$3,210,000</i>	<i>-\$13,368,000</i>
<b>Management costs of wetlands and remnants</b>			
Capital costs of wetland rehabilitation	\$ 253,000	\$ 4,624,000	\$ 4,621,000
Capital costs of native vegetation rehabilitation <sup>a</sup>	\$ 0	\$ 0	\$ 6,761,000
Capital costs of fencing	\$1,137,000	\$ 2,436,000	\$ 5,883,000
Ongoing management costs	\$1,614,000	\$ 3,231,000	\$ 9,894,000
<i>Sub-total</i>	<i>-\$3,004,000</i>	<i>-\$10,290,000</i>	<i>-\$27,159,000</i>
<b>Environmental values generated – consumers’ surpluses</b>			
Duck hunting	\$ 85,000	\$ 220,000	\$ 238,000
Tourism	\$ 531,000	\$ 972,000	\$ 1,492,000
Non-use values	\$8,029,000	\$8,120,000	\$21,217,000
<i>Sub-total</i>	<i>\$8,645,000</i>	<i>\$9,312,000</i>	<i>\$22,947,000</i>
<b>Environmental values generated – producers’ surpluses</b>			
Duck hunting	\$ 26,000	\$ 66,000	\$ 72,000
Tourism	\$ 726,000	\$1,778,000	\$ 2,290,000
Other wetland owner use values		<i>Not estimated</i>	
<i>Sub-total</i>	<i>\$ 752,000</i>	<i>\$ 1,844,000</i>	<i>\$ 2,364,000</i>
<i>Total environmental values</i>	<i>\$9,397,000</i>	<i>\$11,156,000</i>	<i>\$25,311,000</i>
<b>Total changes valued</b>	<b>\$5,227,000</b>	<b>-\$ 2,343,000</b>	<b>-\$15,217,000</b>
<b>Sensitivity of results</b>			
Less conservative non-use value estimates <sup>b</sup>	\$16,816,000	\$16,472,000	\$48,833,000
Total net benefits (less conservative non-use)	\$14,015,000	\$ 6,009,000	\$12,399,000
More conservative endangered species benefits <sup>c</sup>	\$1,171,000	\$ 348,000	\$11,158,000
Total net benefits (more conservative species)	-\$1,631,000	-\$10,116,000	-\$25,275,000

<sup>a</sup> Native vegetation rehabilitation includes wetland and other revegetation.

<sup>b</sup> Conservative non-monetary benefit estimates assume non-respondents to a choice modelling survey hold zero values and only extrapolate accordingly to the survey response rate and only to SA residents. Less conservative assumptions extend the values to the same proportion of Victorian residents at 50% of the values held by SA residents (see Whitten and Bennett 2004).

<sup>c</sup> More conservative endangered species benefits assume only 50% of the estimated benefit to endangered species.

**Note:** Values are net present values of benefit and cost streams over 30 years using a 7% discount rate.

### 3.2.2 A bio-economic model of MRF wetlands

A similar approach, based on an initial literature survey and then a survey of wetland owners and managers, was followed in the MRF (Whitten and Bennett 2004). The key biological factors driving values can be summarised as the area (and type) of healthy wetlands (including size and health of buffer vegetation) and both upstream and site water management. MRF resource reallocations are designed to:

- Facilitate a more natural flood-pulse and reduce the impacts of dams on wetlands;
- Restore interaction between floodplain wetlands and the Murrumbidgee River;

- Alleviate pressures on native vegetation caused by grazing, timber harvesting and weed competition and enhance natural succession towards the desired state and output mix; and,
- Alleviate pressures on native fauna from feral animals and competition by domestic livestock.

Four alternative resource allocations were developed from these goals:

1. Improved hydrological management of water (termed '*water management*');
2. Improved grazing management practices in wetlands and buffer areas (termed '*grazing management*');
3. Improved management of timber harvesting practices in wetlands (termed '*timber management*'); and,
4. Combining the three different options into a single strategy creates a fourth option (termed '*combined strategies*').

The first three options are considered separately and the fourth is an additive combination of the individual resource reallocations. The outcomes of the 'combined strategies' option are not a simple summation of the components because of synergistic responses to resource allocations. A fifth option of continuing BAU provides a comparison point. The summary results from the biophysical modelling of these options are presented in Table 5. For example, compared to BAU, changing grazing management in the MRF is estimated to reduce agricultural production by 15,000 dse, but increase the area of healthy wetlands by 67 km<sup>2</sup>, the populations of wetland and woodland birds by 50 percent, and, the population of native fish by 25 percent.

Economic modelling of the changes to values that would result from the estimated biophysical changes shown in Table 5 are summarised within the bio-economic modelling results presented in Table 6. The costs of production losses and changes to wetland management outweigh the non-monetary benefits generated for all but one option, the 'grazing management' option. For example, the management costs of wetlands, including acquiring and managing water, under the 'hydrological management' strategy (\$13.6m) outweigh the environmental values generated (\$9.2m). However, these results are also sensitive to the degree of extrapolation of the non-monetary environmental values. For example, a less conservative extrapolation of non-use values generates a net benefit from changing management under all strategies. The MRF results are not as sensitive to ecological information as in the USE and no sensitivity tests of the biophysical information are presented in Table 5.

**Table 5: Alternative MRF wetland management strategies and estimated outcomes**

Management strategy / outcome	BAU	Water management	Grazing management	Timber management	Combined strategies
<b>Wetland hydrology</b>	(km <sup>2</sup> )	(km <sup>2</sup> )	(km <sup>2</sup> )	(km <sup>2</sup> )	(km <sup>2</sup> )
Too wet (km <sup>2</sup> )	15	11	15	15	11
Too dry (km <sup>2</sup> )	469	313	469	469	313
Unknown (km <sup>2</sup> )	2	2	2	2	2
Near natural (km <sup>2</sup> )	0	160	0	0	160
<b>Grazing management</b>					
Set stocking rate (km <sup>2</sup> )	217	217	134	217	134
Rotational/crash grazed (km <sup>2</sup> )	239	239	216	239	216
No grazing (km <sup>2</sup> )	61	61	167	61	167
<i>Grazing productivity (dse)</i>	55,000	55,000	40,000	55,000	40,000
<i>Marginal change (dse)</i>	n.a.	0	-15,000	0	-15,000
<b>Timber management</b>					
No logging (km <sup>2</sup> )	206	206	206	294	294
Fallen timber only (km <sup>2</sup> )	33	33	33	27	27
Sustainable management (km <sup>2</sup> )	143	143	143	82	82
Unsustainable yield (km <sup>2</sup> )	41	41	41	20	20
<i>Timber productivity</i>					
Saw logs (m <sup>3</sup> )	20,600	20,600	20,600	11,600	11,600
Residue (m <sup>3</sup> )	41,200	41,200	41,200	23,100	23,100
Firewood (m <sup>3</sup> )	2,000	2,000	2,000	1,600	1,600
<i>Marginal change</i>					
Saw logs (m <sup>3</sup> )	n.a.	0	0	9,000	9,000
Residue (m <sup>3</sup> )	n.a.	0	0	18,100	18,100
Firewood (m <sup>3</sup> )	n.a.	0	0	400	400
<b>Ecological outcomes</b>					
Total healthy wetlands (km <sup>2</sup> )	23	50	90	23	135
Wetland and woodland birds (% pre 1800 population)	40	60	60	50	70
Native fish population (% pre 1800 population)	20	30	25	25	40
Marginal wetland change (km <sup>2</sup> )	n.a.	27	67	0	112
Marginal bird population (% change)	n.a.	50	50	25	75
Marginal fish population (% change)	n.a.	50	25	25	100

**Table 6: Aggregate cost-benefit analysis of MRF management strategies**

Cost or benefit	Water management	Grazing management	Timber management	Combined strategies
<b>Changes to agricultural activities</b>				
Lost agricultural production	\$ 0	-\$3,137,000	\$ 0	-\$ 3,137,000
Cost of providing watering points	\$ 0	-\$ 198,000	\$ 0	-\$ 198,000
Lost timber production	\$ 0	\$ 0	-\$4,678,000	-\$ 4,678,000
<i>Sub-total</i>	\$ 0	-\$3,335,000	-\$4,678,000	-\$ 8,013,000
<b>Management costs of wetlands</b>				
Capital costs of water acquisition	-\$18,161,000	\$ 0	\$ 0	-\$18,161,000
Capital costs of wetland rehabilitation	-\$ 1,151,000	\$ 0	\$ 0	-\$ 1,151,000
Capital costs of fencing	\$ 0	-\$1,261,000	\$ 0	-\$ 1,261,000
Capital costs of wetland revegetation	\$ 0	-\$ 209,000	\$ 0	-\$ 209,000
Ongoing costs of wetland management	-\$ 566,000	-\$1,187,000	\$ 0	-\$ 2,072,000
Income from future water sales	\$ 6,246,000	\$ 0	\$ 0	\$ 6,246,000
<i>Sub-total</i>	-\$13,633,000	-\$2,657,000	\$ 0	-\$16,609,000
<b>Environmental values generated – consumers’ surpluses</b>				
Recreation	\$ 742,000	\$ 1,842,000	\$ 0	\$ 3,078,000
Non-use values	\$ 8,459,000	\$ 9,212,000	\$3,016,000	\$11,832,000
<i>Sub-total</i>	\$ 9,201,000	\$11,053,000	\$3,016,000	\$14,911,000
Wetland owner use values		<i>not estimated</i>		
<b>Total changes valued</b>	<b>-\$ 4,432,000</b>	<b>\$ 5,061,000</b>	<b>-\$1,661,000</b>	<b>-\$ 9,711,000</b>
Non-monetary benefits (less conservative estimates)*	\$26,602,000	\$29,302,000	\$7,093,000	\$38,696,000
<b>Total net benefits (less conservative)</b>	<b>\$13,711,000</b>	<b>\$25,151,000</b>	<b>\$2,416,000</b>	<b>\$17,153,000</b>

**Note:** Values are net present values of benefit and cost streams over 30 years using a 7% discount rate.

\* Conservative non-monetary benefit estimates assume non-respondents to a choice modelling survey hold zero values and only extrapolate according to the survey response rate and only to residents of the Murrumbidgee catchment (including the ACT). Less conservative assumptions extend the response rate to NSW residents at 25 percent of the values held by survey respondents (however, the underlying assumptions remain conservative). For more information see Whitten and Bennett (2004).

### 3.3 Case study policy identification

Policies suitable for consideration in the USE and MRF must generate, in part or whole, the potential Pareto improvements in the bio-economic models in a cost-effective manner. Effective policies will leverage change in wetland management behaviour. Possible points of policy leverage were identified by examining the costs and opportunities within the bio-economic model in combination with an examination of whether market or government failures exist. Policy can then be designed to address these specific targets. Consider the costs to wetland owners of changing their management. Costs are comprised of the upfront costs of changing management (such as new fences, earthworks and revegetation) and ongoing costs (such as pest and weed control). The initial threshold costs of changing wetland management may be larger than landholder benefits but the ongoing costs may be less than ongoing benefits. Therefore, a policy subsidising the upfront costs of changing management may be sufficient to leverage a significant change to wetland management.

The array of potentially cost-effective policies developed for the two case study areas is shown in Table 7. These policies are applicable at alternative jurisdictional levels of government in Australia<sup>8</sup> and are subject to different levels of on-ground effectiveness and policy transaction costs. Note that the list of policy instruments in Table 7 is not complete (there may be others we did not find) and that they may not necessarily be cost-effective. That is, the costs of implementation may outweigh the benefits so generated. Informing the answer to this question is the focus of the policy threshold analysis.

### **3.4 Case study policy threshold analysis**

Policy threshold analysis involves assessing the likely scale of net benefits that would be generated via adoption of alternative policies relative to their transaction costs (due to market or government inefficiencies). Comparing the transaction costs for each policy option requires a scale to facilitate systematic evaluation and comparison. A relative scale of costs was defined for each transaction cost criterion noted in Table 1 and is shown in Table 8. The likely change to wetland protection outputs that would result from each policy instrument when appropriately implemented must also be assessed. For some incentives, structural restrictions are imposed by the level at which the incentive is applied and the possible scale of the incentive. For example, most local government incentives are limited to the geographic area of the jurisdiction. Note that many of the measures in Table 8 are qualitative due to the lack of objective data available, however this approach is consistent with the information constraints.

In addition to the actual transaction costs of the policy itself there may also be impacts on the transaction costs associated with continuing current wetland uses. For example, enacting a more restrictive zoning policy may require an application process before wetland owners can graze wetlands. Assessing the scale and scope of changes to these BAU transaction costs follows a similar methodology to that for other policy attributes. The main difference is that these impacts may be positive or negative.

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<sup>8</sup> Three levels of government potentially influence wetland management in Australia, namely local, state and federal. Local governments have primary responsibility for land-use planning, state governments have the primary legislative basis to natural resource property rights such as water, native vegetation and indigenous animals while the Federal government has the primary taxation powers. In some states catchment management authorities also have some planning powers.

**Table 7: Potential new and existing policy instruments in the USE and MRF**

Policy instrument	Description	Case study	
		MRF	USE
<b><i>New policy instruments</i></b>			
Accreditation or licensing schemes	License potentially damaging activities subject to preventative management strategies or activities being employed..	✓	✓
Assistance in legal cases	Assistance in using legal system to prevent or control external impacts on wetlands.	✓	✓
Broaden tax incentives – management	Options include: 1) tighten eligibility for deduction of damaging capital works; 2) broaden deductions to non-business lands, interest payments and potentially labour; 3) grant primary producer status to conservation land managers (extends option 2 to GST input refunds and diesel fuel rebate); 4) extend heritage structures 20% tax rebate to conservation.	✓	✓
Broaden tax incentives – NPOs	NPO options include: 1) tax deductibility of the bargain sale gap prices for NPOs; 2) capital gains tax exemptions for sales to NPOs; and, 3) higher tax incentives for high conservation value assets.	✓	✓
Conservation covenants	Voluntarily entered, but permanently legally binding restriction on landholder property rights. At present only government can hold and enforce covenants.	✓	✓
Conservation wetland sales	Relaxation of local government restrictions on land parcel size sales (and related) could reduce the transaction costs of wetland protection and lead to improved management.	✓	✓
Devolved management of environmental water	Devolution of environmental water rights and responsibilities to an independent trust could improve management efficiency and reduce potential for capture or political influence.	✓	
Education/ extension	Targets wetland owners who are not interested in change. May motivate self-interest in maintaining private wetland values response to improved understanding of threats and consequences of non-action.	✓	✓
Extend rate rebates / concessions	Rate rebates are restricted to NPWS covenants. Extension to land meeting minimum requirements would reduce costs. Could also extend to other levies (e.g. Pastures Protection boards in NSW).	✓	✓
Farm management planning	Farm management planning (including wetlands): 1) encourages landowners to consider wetlands as an integrated aspect of whole farm management; and, 2) reduces transaction costs of other incentives.	✓	✓
Flexible environmental water allocation	Manage environmental water allocations to include broader ex-stream or floodplain flows.	✓	
GIS database access	Access to government information to assist in program targeting and management. May reduce costs of private sector.	✓	✓

Policy instrument	Description	Case study	
		MRF	USE
Land for wildlife (wetlands)	Targets already interested wetland owners and: <ul style="list-style-type: none"> <li>• recognises their management via a voluntary, non-binding agreement; and,</li> <li>• provide information to improve wetland management.</li> </ul>	✓	✓
Markets for wetland preservation	Competitive bidding for harvest and non-harvest rights (including harvest deferral) in State Forests NSW wetlands.	✓	
New enterprise training	Access to local government business development programs may help develop innovative wetland enterprises or products.	✓	✓
NPO fee waivers	Reduce stamp duty and other fees levied on NGOs for property transfers and conservation management applications.	✓	✓
Non-profit organisation support	NPOs may offer efficient wetland management structures. Government could provide or subsidise capacity building in Australia. (In the US some large non profit organisations undertake this role).	✓	✓
Operationalise 'duty of care'	Specify minimum wetland management performance standard.	✓	✓
Ranching for wildlife	Managing native fauna in their natural environment in order to harvest a proportion of the fauna for sale.	✓	✓
Real estate tools for NPOs (training)	<ul style="list-style-type: none"> <li>• Option to purchase within a specified time and price;</li> <li>• Rights of first refusal at the offer price.</li> <li>• Options on conservation covenant purchase prior to sale.</li> </ul>	✓	✓
Removal of perverse tax incentives	Remove tax deduction or rebates for construction of any drought water storage in a wetland (which is often the easiest and most convenient site for such storages).	✓	✓
Remove perverse water subsidies	Fully include all costs of water supply.	✓	
Repair / investment funds	Subsidises upfront costs of wetland rehabilitation and changing management.	✓	✓
Resource based zoning	Require landholders to maintain wetland resource bundle such as water and land (similar to native vegetation clearance laws).	✓	✓
Revolving funds	Signals the conservation value wetlands to buyer. Forms include: <ol style="list-style-type: none"> <li>1) urgent land purchases on-sold to government or others;</li> <li>2) short-term bridging funding to allow NPO fundraising; and,</li> <li>3) adding a conservation covenant and on-selling.</li> </ol>	✓	✓
'Safe harbour' schemes	Legal guarantee that improved wetland management outcomes will not become subject to existing legislation. May lead to mitigation banking or tradable credit schemes. Name borrowed from US programs – see Environmental Defense (2000).	✓	✓
Signalling measures	Recognises management of 'high value' wetlands (such as Ramsar, national or state significant). May trigger wetland owners to seek support or assistance with management.	✓	✓
Simplify wetland rehabilitation/ re-	Development applications are often required for wetland rehabilitation works – particularly in designated areas	✓	✓

Policy instrument	Description	Case study	
		MRF	USE
creation	such as floodplains. Simplification would reduce costs.		
Stewardship payments	Subsidise ongoing wetland management towards agreed goals.	✓	✓
Water covenants	Partial property rights for water could facilitate privately owned environmental water. For example, water for filling wetlands.	✓	
Wetlands tourism infrastructure	Development of tourism support infrastructure including scenic drives, brochures, training and market access.		✓
Wetlands tourism promotion	Ensure local tourism promotion incorporates wetland information including national and international importance.		✓
Zoning regulations	Review zoning regulations to ensure appropriate flexibility in management.	✓	✓
<b><i>Existing policy instruments</i></b>			
Management advice	State government agencies offer free advice to wetland owners on ways of managing their wetlands.	✓	✓
Landcare tax incentives	Tax rebates or deductions of eligible expenses including property planning and revegetation. Must be in an approved farm plan.	✓	✓
Local council rate rebate	Reduced local government rates subject to minimum conditions.		Some local governments
Materials grants	Fencing materials supplied for livestock management in wetlands.	✓	✓
Other materials grants	As above but supplied by Greening Australia – a non-government conservation organisation.	✓	
NHT Grants	Grants to groups for works to address environmental concerns.	✓	✓
USE specific grants	A subsidy for conservation works in USE wetlands. Subject to minimum conditions local drainage and salinity taxes reduced.	n.a.	✓

**Table 8: Definition of measures for policy transaction cost criteria**

Policy criteria	Measure of criteria		
	Low	Medium	High
<b>Government policy transaction costs</b>			
<i>Design information</i>	Already exists	Existing information can be adapted / little new information collected	Significant new information must be collected
<i>Enactment</i>	No new legislation required, little or no new regulations required and no significant lobbying	Templates for legislation and regulations available in other jurisdictions and/or minor lobbying	New legislation and/or regulation required and/or significant lobbying
<i>Implementation</i>	Implementation through existing structures and processes – few new resources required	Some new structures and processes required but within existing organisations.	New organisations required or significant enlargement of existing organisations
<i>Administration</i>	Homogeneous treatment of landholders – no site visits required Few transfers through government	Heterogeneous treatment of landholders or single site visit Small transfers through government	Multiple site visits or heterogeneous treatment and site visit required Significant transfers through government
<i>Detection</i>	Effective within existing procedures (low marginal cost to these), no onsite monitoring	Site visit required or development of new monitoring procedures	Ongoing site visits required and/or costly measurement strategies
<i>Prosecution</i>	No prosecution option	Existing mechanisms leading to a well defined, rarely-contested set of penalties (few legal suits)	Prosecution requires action through law courts or penalties are regularly contested
<i>Risk</i>	Removes proven threats to healthy existing wetlands using proven policy instrument	Proven policy instrument applied to a new location or issue, treatment of hypothesised threats	Unproven policy instrument or new potential management solutions
<b>Market policy transaction costs</b>			
<i>Direct costs</i>	No direct fees or charges	Nominal fees or charges	Full cost recovery of administration and monitoring costs
<i>Additional information</i>	No additional information required	Provision of information collected for other uses anyway or which is cheap and easily collected	Supply of previously uncollected information requiring new methods or processes
<i>Contracting</i>	No written agreements and no more than one meeting	Non-legally binding written agreement or addition to existing requirements or few meetings	New legally binding agreement (requiring legal input) or multiple meetings
<i>Detection and protection</i>	No change to property rights or requires greater 'duty of care' or similar to retain existing rights	Property rights only retained/obtained by simple application	Property rights require defence or challenge through complicated process requiring new information

Policy criteria	Measure of criteria		
	Low	Medium	High
<b>Other</b>			
<i>Dynamic attributes</i>	Mandated actions – no opportunity to innovate	Mandated actions but significant site adjustment or broad specification – some innovation	No mandated action – only outcome is specified
<b>Overall effectiveness in achieving wetland protection</b>			
<i>Effectiveness</i>	Unlikely to significantly impact on aggregate outcomes in isolation but may be important signal in policy instrument mix	Important supporting role to other policies and will achieve some management change in isolation	Significant driver of changing wetland management – key element of policy instrument mix

**Note:** Sources for the rationale behind most criteria measures are based on literature noted under Table 1 and are not repeated here. Other sources include the National Center for Environmental Economics (2001) and Wiersma (1991) (dynamic attributes), Stavins (1995) (administration, contracting and other areas), and, Polinsky and Shavell (1992) and Kaplow (1992) (prosecution costs).

Applying the criteria in Table 8 to the policy instruments in Table 7 generates the summary qualitative evaluation of alternative policy options presented in Table 9.<sup>9</sup> The results in Table 9 are a summary of the complete transaction cost assessment matrix shown in Appendix 1. For example, applying the criteria in Table 8 to a new policy instrument introducing stewardship payments to wetland owners yields the following assessments:

- design and information costs are medium – some new information about individual wetland management will be required to target stewardship payments;
- enactment costs are medium – templates are available for similar programs and could be adapted (for example, the Bushtender program in Victoria). There may be some lobbying relating to payment amounts and targets;
- implementation costs are medium – some new structures will be needed within an existing organisation to deliver the instrument;
- administrative costs are high – generated by heterogeneous treatment of landholders through differential stewardship payments that require more detailed evaluation and a site visit to assess eligibility;
- detection costs are high – ongoing site visits are required or the development of new monitoring strategies;
- prosecution costs are medium – prosecutions are considered unlikely due to the costs of prosecution but could be costly if undertaken;
- risk is medium – stewardship payment mechanisms are proven overseas but have not been applied to wetlands in Australia;
- direct costs are low – no direct fees or charges apply;
- additional information costs are high – collection or assembly of new information is required to apply for stewardship payments and possibly to show ongoing compliance;
- contracting costs are high – stewardship payments are likely to require a legally binding agreement with consequent negotiation and private administration costs;
- private detection/protection costs are low – a stewardship payment changes property rights or duty of care responsibilities by agreement;
- dynamic impacts are medium – eligible management actions are broadly specified with some opportunity for innovation;
- BAU transaction costs are unchanged; and,
- wetland protection effectiveness is high – stewardship payments are defined to be greater than monetary losses less non-monetary benefits to wetland owners. Hence, significant changes to wetland management and consequent increases in wetland protection outputs are anticipated.

Overall it is unclear whether the policy transaction costs of a stewardship payment scheme are likely to outweigh the benefits from additional wetland conservation.

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<sup>9</sup> The summary results in Table 9 are the result of applying weightings to each of the transaction cost criteria in Table 8. The policy transaction cost data is shown in Appendix 1 (Table A1) together with the weightings (Table A2). Sensitivity analysis of the weightings showed little variation in the conclusions from a wide change in weighting.

**Table 9: Wetland policy instrument cost-effectiveness**

<b>Policy instrument</b>	<b>Type</b>	<b>Jurisdiction</b>	<b>Overall transaction costs</b>	<b>Biophysical wetland protection effectiveness</b>	<b>Cost-effectiveness rating</b>
<b><i>Potential policies common to USE and MRF</i></b>					
Conservation covenants	Facilitate	State	Low	High	Likely
Land for wildlife (wetlands)	Facilitate	State	Low	High	Likely
Education/ extension	Facilitate	State	Low	High	Likely
Broaden tax incentives – management	Induce	Federal	Medium	High	Likely
Revolving funds	Facilitate	State	Medium	High	Likely
Non-profit organisation support	Facilitate	Federal	Low	Medium	Likely
Repair/investment subsidies - USE	Induce	State	High	High	Unclear
Repair/investment subsidies - MRF	Induce	State	High	High	Unclear
Stewardship payments	Induce	State	High	High	Unclear
Farm management planning	Facilitate	State	Medium	Medium	Unclear
Extend rate rebates / concessions	Induce	Local	Medium	Medium	Unclear
Simplify wetland rehabilitation/ re-creation	Facilitate	Local/State	Medium	Medium	Unclear
Broaden tax incentives – non-profits	Induce	Federal	Medium	Medium	Unclear
GIS database access	Facilitate	State	Low	Low	Unclear
Signalling measures	Facilitate	State	Low	Low	Unclear
Conservation wetland sales	Facilitate	Local/State	Low	Low	Unclear
Real estate tools for Non-profits	Facilitate	Non-specific	Low	Low	Unclear
New enterprise development training	Facilitate	Local	Low	Low	Unclear
Zoning regulations	Compel	Local	High	Medium	Unlikely
Ranching for wildlife	Facilitate	State	High	Medium	Unlikely
Operationalise ‘duty of care’	Compel	State	High	Medium	Unlikely
‘Safe harbour’ scheme	Compel	State	High	Medium	Unlikely
Resource based zoning	Compel	State	High	Low	Unlikely
Accreditation or licensing schemes	Compel	State	High	Low	Unlikely
Stamp duty exemptions & development fee waivers	Induce	State	Medium	Low	Unlikely
Access to legal systems	Facilitate	State	Medium	Low	Unlikely
<b><i>USE specific policy instrument</i></b>					
Wetlands tourism promotion	Facilitate	Local	Low	Medium	Likely
Wetlands tourism infrastructure	Facilitate	Local	Low	Medium	Likely
<b><i>MRF specific policy instrument</i></b>					
Flexible management of environmental water allocation	Facilitate	State	Medium	High	Likely
Water covenants	Facilitate	State	Low	High	Likely
Devolved management of environmental water	Facilitate	State	High	High	Unclear
Remove perverse tax incentives	Induce	Federal	Medium	Medium	Unclear
Remove perverse subsidies	Induce	State	Medium	Medium	Unclear
Markets for timber preservation	Facilitate	State	Low	Low	Unclear
<b><i>Existing policy instruments</i></b>					
Drainage levy rebate	Induce	Regional	Low	Medium	Likely
Greening Australia materials grants	Induce	Regional	Medium	High	Likely
Management advice - government agencies	Facilitate	State	Medium	Medium	Unclear
Landcare tax rebate / deduction	Induce	Federal	Low	Low	Unclear
Local council rate rebate	Induce	Local	Low	Low	Unclear
Grants from NHT	Induce	Federal	High	Low	Unlikely

#### 4 Discussion of case study results

Several conclusions can be drawn from the summary data presented in Table 9. Policies that are evaluated as ‘likely’ to meet the threshold policy analysis test have been judged to have total costs (including transaction costs) lower than their benefits. Hence, these policies are considered ‘best bets’ for adoption in the case study areas. Where concerns about the extent of the transaction costs associated with these policies remain, estimation of these costs should be undertaken. That is, the threshold policy analysis is acting as a ‘filter’ to identify policies worthy of further investigation. Similarly, policies evaluated as ‘unlikely’ to be cost-effective should not be considered for adoption with respect to the case study areas because they are considered highly unlikely to generate a net benefit. Finally, the analysis shows that only two of the current USE and MRF policies are regarded as likely to be cost-effective and one is regarded as not cost-effective in its current form.

The conclusions regarding policies ranked as ‘unclear’ in Table 10 are more complex. In these cases, the relatively coarse, judgement-based information is insufficient to identify whether these policies are likely to generate a net benefit when all transaction and production costs and benefits are considered. Additional information is required to assist in judgements about their likely cost effectiveness. In some cases, part of this information may be available from the bio-economic model together with estimates of transaction costs from other sources.

As an example, consider the case of repair/investment subsidies in the USE and MRF. The capital costs associated with wetland management changes to achieve the desired strategies amount to \$1.4m in the USE and \$1.7m in the MRF. Evidence from existing schemes can be used to anchor transaction cost estimates in a similar fashion to the use of benefit transfer in identifying values for non-market goods and services when other estimates are not available. USE program evidence suggests that a team of two individuals could implement, administer and monitor a \$1m to \$2m program over approximately two years (costing at least \$400,000) (USEDSEMPSC 1999). Approximately, an additional 20 percent of transfer values (\$360,000 in the USE and \$420,000 in the MRF) will be incurred in tax collection and redistribution according to Campbell and Bond (1997). Finally, landowners will also incur transaction costs in applying for and receiving subsidies. If these are also in the order of ten to twenty percent of the subsidy, total program transaction costs will be approximately eighty percent of the value of the subsidy. That is, total costs (including transaction costs) amount to 1.8 times capital costs. This generates a benefit-cost ratio of about 1.4:1 in the USE and 1.1:1 in the MRF. Recalling that the benefit estimates in Tables 3 and 5 are conservative due to the restrictive assumptions about extrapolation of environmental values (see for example the impact of sensitivity tests) and because wetland owner non-monetary benefits are not included, repair/subsidy programs are likely to be cost effective in the USE and MRF regions. Similar analyses could be performed for other policy instruments classified as ‘unclear’ in Table 9.

In a broader context, policy threshold analysis does not inform which policies will work in a synergistic fashion and which may conflict with each other. However, the compatibility of potentially cost-effective policies should be taken into account in selecting the policy package implemented. This trade-off is limited by the jurisdictional powers of the policy maker, although policy makers may also be able to influence decisions beyond their jurisdiction.

Policy makers must also make judgements about the likely costs of postponing a decision, and collecting and analysing additional information per the discussion on possible irreversibilities or environmental impact thresholds. It is useful to consider the results from the policy threshold analysis in this context. The sensitivity analysis shown in Table 4 indicates that the benefits from changing wetland management are much lower if changes to management are insufficient to produce benefits to threatened species in the region as estimated. These benefits include preventing the regional extinction of some species, that is, a regional threshold. In this context it may be prudent to adopt more policies that have been found 'likely' to be cost effective in order to avoid the negative benefits generated by the species threshold being breached. A similar example in the MRF may be the population of native fish. Once the population reaches a particular (unknown) threshold, recreational fishing values may increase significantly. In this case it may be prudent to enact more policies earlier than would be the case if no thresholds existed.

However, policies ranked unlikely or unclear should still not be adopted because they are not likely to generate net benefits to the community. Nor should the total policy transaction costs associated with the mix of policies implemented exceed the net benefits generated.

Finally, there are also likely to be high quasi-option values associated with additional information about the effectiveness of wetland policies in achieving goals involving thresholds such as regional species conservation in the USE. There are also likely to be high quasi-option values associated with information that identifies biophysical or value thresholds and allows adjustment of policy to the potential range of outcomes that may occur. For example, identification of whether there are fish population thresholds with respect to recreational fishing values.

## **5 Conclusions**

The potential for bio-economic modelling to influence the identification and implementation of environmental policies is significant. Bio-economic models are able to assist with setting appropriate targets for management change that would increase community welfare. Uncertainty remains in bio-economic modelling outputs due to incomplete biophysical and economic information. The implication of the remaining uncertainty is that the appropriate policy target is a range rather than a single optimal value. The information contained in bio-economic models can also be used to develop policy instruments that most effectively influence the costs or benefits of changing environmental management. Hence, it is useful to undertake bio-economic modelling prior to developing potential policy solutions.

Cost effectiveness comparisons between the alternative policy options identified and developed from the bio-economic modelling framework are more complex. The difficulty arises from incomplete knowledge about the transaction costs inherent in alternative policy approaches. One response is to adopt policy threshold analysis as a consistent methodology for judging between alternative policy options in the presence of such uncertainty. Policy threshold analysis has been shown to act as a useful filter in identifying the 'best bet' policy options for further consideration while systematically rejecting those for which adoption is unlikely to yield a net benefit to the community.

A more extreme form of uncertainty may also arise in the presence of potential irreversible or threshold environmental impacts in the biophysical modelling. In these cases, a discontinuity in the values generated from the bio-economic model arises that must be taken into account within any threshold policy analysis undertaken by the policy maker. One important impact is that the quasi-option value of postponing a decision in order to collect and analyse more information may be significantly reduced and the optimal schedule of policy adoption may be shortened. The danger in such an approach is that the increased policy costs will outweigh the net benefits available. This danger can be reduced, in part, by restricting consideration to those policies that are likely to prove cost-effective.

Finally, there are a number of areas of future research that emerge from this paper. First, improving the estimation of both biophysical and economic outcomes will enhance policy makers' ability to target policy. Second, while policy threshold analysis can filter 'best bet' policies, there is still much to be done to develop methodologies to estimate transaction costs of alternative policies. Finally, very little is known about the specific impacts of mixing policies on policy transaction costs and effectiveness in achieving wetland management change.

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## Appendix 1: Transaction cost assessments for alternative wetland policy instruments

*Table A1: Qualitative assessment of the transaction costs of alternative policy instruments*

Policy instrument	Government policy transaction costs							Market policy transaction costs				Dynamic impacts	Change to BAU TCs	Bio-physical wetland protection
	Design	Enact	Implement	Administrate	Detection	Prosecution	Risk	Direct costs	More info	Contract	Detect/protect			
<i>New policy instruments</i>														
Accreditation or licensing schemes	Medium	Medium	High	Medium	High	High	Medium	Medium	High	High	Medium	Medium	Negative-medium	Low
Assistance in legal cases	Medium	High	Low	Low	n.a.	n.a.	Medium	Low	High	High	Low	High	Positive-low	Low
Broaden tax incentives – management	Medium	High	Low	Medium	Low	Medium	Low	Low	Medium	Medium	Low	Medium	no change	High
Broaden tax incentives – NPOs	Medium	Medium	Low	Medium	Low	Medium	Low	Low	Medium	Medium	Low	Medium	no change	Medium
Conservation covenants	Low	High	Low	Low	Medium	Medium	Low	Low	High	High	Low	Medium	Positive-high	High
Conservation wetland sales	Low	Low	Low	Low	Low	Medium	Medium	Low	Medium	Medium	Low	Medium	Positive-low	Low
Devolved management of environmental water	High	High	High	Medium	Medium	Low	High	Low	Low	Low	Low	High	no change	High
Education / extension	Medium	Low	Medium	Medium	Low	Low	Low	Low	Low	Low	Low	High	Positive-high	High
Extend rate rebates / concessions	Low	Low	Low	Medium	Medium	Medium	Low	Low	Medium	Medium	Low	Low	no change	Medium
Farm management planning	Low	Low	Low	Low	Low	Low	Low	Low	Medium	Medium	Low	Medium	Positive-medium	Medium
Flexible environmental water allocation	Medium	High	Medium	Low	n.a.	n.a.	Medium	Low	Low	Low	Low	Medium	no change	Medium
GIS database access	Low	Low	Medium	Low	Low	Low	Medium	Low	Low	Medium	Low	High	Positive-medium	Medium
Land for wildlife (wetlands)	Low	Low	Medium	Medium	Medium	Low	Low	Low	Low	Medium	Low	High	Positive-high	High

Policy instrument	Government policy transaction costs							Market policy transaction costs				Dynamic impacts	Change to BAU TCs	Bio-physical wetland protection
	Design	Enact	Implement	Administrate	Detection	Prosecution	Risk	Direct costs	More info	Contract	Detect/protect			
Markets for wetland preservation	Medium	High	Low	Low	Low	Medium	Medium	Low	Low	High	Low	Low	Positive-medium	Low
New enterprise training	Low	Low	Low	Low	Low	Low	High	Low	Low	Low	Low	High	Positive-low	Low
NPO fee waivers	Low	Medium	Low	Low	Low	Medium	Low	Low	Medium	High	Low	Low	Positive-low	Low
Non-profit organisation support	Low	Low	Medium	Low	Low	Low	Medium	Medium	Low	Low	Low	High	Positive-medium	Medium
Operationalise 'duty of care'	High	High	Low	Low	Medium	High	Medium	Low	High	Low	High	High	Negative-low	Medium
Ranching for wildlife	High	Medium	Medium	High	Medium	High	High	Low	High	High	Low	Medium	Positive-high	Medium
Real estate tools for NPOs (training)	Low	Low	Low	Low	Low	Low	High	Low	Low	High	Low	High	Positive-low	Low
Remove perverse tax incentives	Medium	Medium	Low	Low	Low	Medium	Low	Low	Low	Low	Low	Low	no change	Medium
Remove perverse water subsidies	Medium	Medium	Low	Low	Medium	Medium	Medium	Low	Medium	Medium	Low	Low	no change	Low
Repair / investment subsidies – MRF	Medium	Medium	Medium	High	Medium	High	Low	Low	High	High	Low	Low	no change	High
Repair/ investment subsidies – USE	Low	Low	Low	High	Medium	Medium	Low	Low	High	High	Low	Low	no change	High
Resource based zoning	Medium	High	Low	Medium	Medium	High	High	Low	Low	Low	High	Low	Negative-medium	Low
Revolving funds	Low	Medium	High	Medium	Medium	High	Low	Low	Medium	High	Low	Medium	Positive-high	High
'Safe harbour' scheme	Medium	High	Medium	Medium	Medium	High	Medium	Low	High	High	Low	Medium	Positive-medium	Medium
Signalling measures	Low	Medium	Medium	Medium	Medium	Low	High	Low	Low	Low	Low	High	no change	Low
Simplify wetland rehabilitation / re-	Low	Medium	Low	Medium	Medium	High	Medium	Low	Medium	Medium	Low	Medium	Positive-medium	Medium

Policy instrument	Government policy transaction costs							Market policy transaction costs				Dynamic impacts	Change to BAU TCs	Bio-physical wetland protection
	Design	Enact	Implement	Administrate	Detection	Prosecution	Risk	Direct costs	More info	Contract	Detect/protect			
creation														
Stewardship payments	Medium	Medium	Medium	High	High	Medium	Medium	Low	High	High	Low	Medium	no change	High
Water covenants	Medium	Medium	Low	Low	Low	Medium	Medium	Medium	Medium	High	Low	Medium	Positive-high	Medium
Wetlands tourism infrastructure	High	Low	Low	High	n.a.	n.a.	High	Low	Medium	Low	Low	High	Positive-high	Medium
Wetlands tourism promotion	Medium	Low	Low	Low	n.a.	n.a.	High	Low	Low	Low	Low	High	Positive-high	Medium
Zoning regulations	High	High	Low	Medium	Medium	High	Medium	Low	Low	Low	High	Low	Negative-high	Medium
<i>Current policy instruments*</i>														
Management advice	n.a.	n.a.	n.a.	Medium	Low	Low	Medium	Low	Low	Low	Low	Medium	Negative-medium	Medium
Landcare tax rebate / deduction	n.a.	n.a.	n.a.	Low	Low	Medium	Low	Low	Medium	Low	Low	Medium	No change	Low
Local council rate rebate	n.a.	n.a.	n.a.	Low	Low	Medium	Low	Low	Low	Medium	Low	Low	No change	Medium
Drainage levy rebate	n.a.	n.a.	n.a.	Low	Low	Medium	Low	Low	Low	Medium	Low	Low	No change	Low
Govt materials grants	n.a.	n.a.	n.a.	Medium	Medium	High	Low	Low	Low	High	Low	Low	No change	Medium
Greening Australia materials grants	n.a.	n.a.	n.a.	Medium	Medium	Medium	Low	Low	Low	Medium	Low	Medium	No change	High
Grants from NHT	n.a.	n.a.	n.a.	High	Medium	High	Medium	Low	Medium	High	Low	Medium	No change	Low
Wetlands Waterlink grants	n.a.	n.a.	n.a.	High	Medium	High	Low	Low	Low	High	Low	Low	No change	High

**Note:** 'Wetlands Waterlink' and 'drainage levy rebate' combined as 'USE specific grants' in Table 7.

- Design, enactment and implementation costs are sunk costs for current policies.

**Table A2: Transaction cost weights – aggregation and sensitivity analysis**

Transaction cost type		Base Case	Equal group weights <sup>#</sup>	Equal cost weights	Admin weight halved	Current TC weight halved	Current TC weight doubled
<b>Government policy transaction costs</b>	<b>Design</b>	1	1	1	1	1	1
	<b>Enact</b>	1	1	1	1	1	1
	<b>Implement</b>	1.5	1.5	1	1.5	1.5	1.5
	<b>Administrative</b>	4	4	1	2	4	4
	<b>Detection</b>	1	1	1	1	1	1
	<b>Prosecution</b>	0.5	0.5	1	0.5	0.5	0.5
	<b>Risk</b>	0.5	0.5	1	0.5	0.5	0.5
<b>Market policy transaction costs</b>	<b>Direct costs</b>	1	2.71	1	1	1	1
	<b>More info</b>	1.5	4.07	1	1.5	1.5	1.5
	<b>Contract</b>	1.5	4.07	1	1.5	1.5	1.5
	<b>Detect/ protect</b>	1	2.71	1	1	1	1
<b>Dynamic</b>	-1.5	-4.07	-1	-1.5	-1.5	-1.5	
<b>Change to BAU transaction costs</b>		3	9.5	1	3	1.5	6

<sup>#</sup> Equal group weights: equal weighting to government, market (including dynamic) and current transaction costs as bundles.