Targeting incentives for policy: Linking bio-economic modelling to on-ground outcomes *

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Abstract

Environmental policy has assumed a high profile in Australia with recent policies addressing aspects of land degradation (National Heritage Trust and the National Action Plan), forest management (Regional Forests Agreements) and climate change (programs administered by the Australian Greenhouse Office), among others. These policies are often based on relatively little information about the likely benefits to be generated or cost borne. In this paper, the issue of policy development is directly addressed using a case study of wetland policy. The desirable scale of the policy response to environmental issues is informed by the development of the notion of threshold policy analysis. The suite of policy options that should be adopted is dependent on the scale and type of change desired from the policy. The degree of irreversibility and notion of environmental impact thresholds also affects 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: Environmental policy, optimal policy design

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1 Introduction

Environmental policy in Australia has assumed a high profile in Australia. Recent policies have been developed that seek to address aspects of land degradation (National Heritage Trust and National Action Plan for Salinity and Water Quality), forest management (Regional Forest Agreements) and climate change (programs administered by the Australian Greenhouse office) among others. The Commonwealth land degradation and climate change special programs have allocated over $4 billion dollars towards environmental management alone.[1]

Despite the scale of expenditure and the claims about the levels of benefits in press releases, 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 (Bennett 2001). For example, the National Action Plan allows for economic evaluations of regional plans but the National Action Plan is not supported by a similar analysis of the overall policies employed within the program. Pannell (2001) notes that the National Action Plan continues to place unrealistic expectations on the adjustment burden that farmers are willing and able to bear, as did the preceding programs the National Heritage Trust and the National Landcare Program. Furthermore, little quantitative modelling of outcomes has been undertaken at either the aggregate policy level or at the sub-program or regional level. Ironically, this includes much of the process of setting regional targets that may be inferior to business as usual outcomes unless based on detailed empirical analyses (Pannell 2001).

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 outline how one such tool, bio-economic modelling, can be used to help develop goals and target incentives in the policy development process. The bio-economic modelling process is demonstrated by way of case studies of wetland policies in the Upper South East (USE) of South Australia and on the Murrumbidgee River Floodplain (MRF) in New South Wales. Specifically, bio-economic modelling provides information on the scale and distribution of potential benefits that changes to environmental management are likely to achieve. Bio-economic modelling also provides information on the distribution and scale of costs that would be imposed on differing sections of the community if management was changed. Inclusion of this information in the policy development process provides for a more rigorous analysis and targeting of public investments.

Risk and uncertainty in policy development may arise from ignorance about elements of the bio-economic model or from ignorance about aspects of the alternative policy options.[2] Ignorance about elements of bio-economic modelling is due to a combination of biophysical ignorance and value ignorance. Biophysical ignorance is the result of a lack of adequate scientific knowledge about many aspects of environmental management. For example, uncertainty about how a particular species will react to a change in management. In particular, information is often lacking

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[1] National Heritage Trust $2.5b, National Action Plan for Salinity and Water Quality $0.7b and climate change programs $0.9b.

[2] A situation involving risk is defined as one in which the probability of different outcomes is known. Uncertainty is characterised by unknown probabilities.

about the degree of irreversibility and possibility of environmental impact thresholds inherent in production of the environmental outputs desired and the impacts of alternative management actions. Value ignorance arises from a lack of information about the size and extent of the values generated by environmental systems. Sensitivity analysis is used to take into account the biophysical and value ignorance in bio-economic modelling.

The difference between the actual level of environmental outputs and that which would maximise community welfare (as estimated in the bio-economic model) is caused by aspects of market or government failure in the production of environmental outputs. However, the policies developed to reduce the current level of market or government failure are themselves subject to market and government inefficiencies. Market and government inefficiencies are due to the costs of implementing institutional structures such as property rights, regulations, subsidies and other mechanisms. Uncertainty exists in the policy development process because of incomplete knowledge about the scale and distribution of these costs. Policy development uncertainty can be taken into account via policy threshold analysis. Policy threshold analysis is a comparison of the likely range of costs caused by market and government inefficiency against the net benefits of the additional environmental outputs so produced. Policy threshold analysis can be used to rank potential policies based on the likely relative scale of their net costs due to market or government inefficiencies. Comparing these net costs against the potential net benefits estimated in the bio-economic modelling facilitates selection of the policy mix that maximises the net benefits to society as a whole.

Taking value ignorance, biophysical ignorance and incomplete information about market and government inefficiencies of environmental policy into account means that policy timing is in part a function of quasi-option values – the value of postponing a decision in order to collect more information.

The rationale for this paper has been set out in this introduction. The paper is divided into two parts. In the first part the concepts of bio-economic modelling and its contribution to policy development are briefly described. The next section provides some background to the paper by briefly describing the bio-economic modelling process. The linkage between bio-economic modelling and policy is then discussed including how biophysical ignorance and value ignorance may be taken into account by sensitivity analysis. The impact of similar uncertainty generated by incomplete knowledge about market and government inefficiencies inherent in alternative policy options leads to the notion of policy threshold analysis for selecting appropriate combinations of policy options in the third section. The impact of more extreme versions of biophysical ignorance introduced by potential irreversibility or environmental impact thresholds is the focus in the fourth section of the paper.

These concepts are then briefly demonstrated with respect to two case studies of wetlands, in the USE of SA and MRF of NSW in the second part of the paper. The demonstration shows the basic outcomes of the bio-economic model and their ramifications for policy development, including the implications of biophysical ignorance and value ignorance. The impact of value ignorance and ignorance about market and government inefficiencies in deciding between alternative policy options and policy mixes is then briefly discussed. A brief summary of the main findings in this paper and their implications for economic research concludes the paper.

Part 1 Bio-economic modelling concepts and policy

Policy makers are faced with uncertainty about the scale of management change that is desirable and about the effectiveness of alternative policy options and packages when considering the design and adoption of environmental policy. Bio-economic modelling has the capability to reduce significantly the uncertainty about the appropriate scale of policy change that would generate the maximum net benefit to the community. The outputs of a bio-economic model also play a significant role in reducing the uncertainty associated with the effectiveness of alternative policy options. The contribution that can be made by bio-economic modelling is discussed in this part of the paper along with the use of policy threshold analysis to deal with the remaining uncertainty associated with policy effectiveness. A brief discussion of the implications of more extreme forms of ignorance about scientific information on bio-economic modelling and policy development completes this part of the paper.

2 Bio-economic modelling

Bio-economic modelling involves the quantitative assessment of the change in the net benefits to the community that result from changes to environmental management. The purpose of bio-economic modelling is to inform the decision making process. In doing this, the process reduces the uncertainty about the scale of management change that is appropriate. The concept of cost-benefit analysis underlies the definition of a 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 constraints.

Bio-economic modelling is a three-stage process:

1. Biophysical modelling – modelling changes in the biophysical status from changes to environmental management;
2. Economic modelling – modelling community values associated with alternative biophysical states; and,
3. Consolidation into a bio-economic model – modelling 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 private and social values.
2. The prediction of the outcomes, in terms of changes to biological factors, under different landscape scale management strategies.
3. The prediction 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 2 from step 3. 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 fencing of remnants 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'.[3] Despite the relatively small area that changes use it is posited to impact significantly on costs and benefits of the system being modeled.

Whereas biophysical modelling is the compilation and analysis of the biological factors that underlie private and social values, economic modelling is the compilation and analysis of the economic information required for a cost-benefit analysis. The economic modelling is the process of estimating the values of the costs or benefits of the marginal changes in the biophysical factors.

It is important to recognise that the economic modelling component refers to the change in total community benefits that would result from each potential management strategy and not only monetary changes. The concept of economic modelling is based on the theory of economic surpluses. An economic surplus occurs where either the producer or consumer receives a net benefit. The existence of both monetary and non-monetary values for wetland outputs complicates the economic modelling. 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. The management strategy that would lead to the highest net community benefit among the set of potential management strategies developed in the biophysical modelling phase can then be identified. The highest net benefit management strategy represents the goal of policy development. That is, cost-effective policies should be developed to achieve the highest net benefit outcome from the bio-economic model.

Uncertainty and bio-economic modelling

The highest net benefit strategy should not be referred to as the optimum because it remains subject to uncertainty. The uncertainty arises from biophysical ignorance and value ignorance. A further source of uncertainty arises because net benefits will change over time as individuals preferences shift. These forms of uncertainty are well known in cost-benefit analysis and are assessed via sensitivity analysis. Sensitivity analysis is conducted by changing assumptions about the key parameters in the bio-economic model and re-estimating the net benefit that is generated. The output from

[3] 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 analysis of changes introduces an element of uncertainty that should be taken into account in the policy development process.

the sensitivity analysis shows the likely range of the net benefit outcomes and which parameters have greatest leverage on the net benefits. The distribution of outcomes within the range assessed by the sensitivity analysis usually remains uncertain 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. Uncertainty within the bio-economic model dictates that policy goals are best expressed as an appropriate range to which environmental policies should be targeted towards achieving, rather than as an absolute value.

3 Policy selection, bio-economic models and threshold policy analysis

A bio-economic model can assist in setting appropriate environmental outcome goals for policy development as shown in Section two. The next logical step is selection of appropriate cost effective policy options to achieve these goals. Thus, in order to select appropriate policy options we need to know the relative cost-effectiveness of each option or package of options for achieving the environmental outcome goal. The components of bio-economic modelling provide significant input into selecting appropriate and cost-effective policy tools. However, considerable uncertainty remains about the relative cost-effectiveness of alternative policy options due to incomplete knowledge about market and government inefficiencies. One approach to dealing with such uncertainty is the concept of policy threshold analysis. The contributions of bio-economic modelling and policy threshold analysis to policy development are the focus in this section.

A bio-economic model identifies the beneficiaries and those who have costs imposed on them as a result of management changes along with the relative scale of the costs and benefits. This information is collected as part of the economic modelling phase of bio-economic modelling. This information can be used to target policy development towards those that are likely to significantly influence the decisions of environmental managers by impacting on the costs and benefits that result from management change. This information can also be used to quantitatively rank the relative potential of the policy options developed in terms of their relative potential to influence the costs or benefits of environmental managers. 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.

The cost-effectiveness of the range of policies developed from the bio-economic modelling information is subject to uncertainty. The uncertainty in policy development arises due to incomplete knowledge about market or government inefficiencies inherent in alternative policies. The causes of market inefficiencies include transaction costs and potential for market imperfections inherent in market based policy tools. Similarly, the causes of government inefficiencies include costs of information gathering, monitoring and derived externalities in redistributive policy tools. Bio-economic modelling only includes estimates of the direct costs or benefits to wetland owners and the wider community of management changes. It does not take into account the additional costs that may be imposed by government or market inefficiencies in attempting to achieve management changes. 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. Costs associated with collection and redistribution of funds, program management and the transaction costs to the farmer of seeking assistance are not included in the bio-economic model. Non-inclusion is because there are a number of policies that may achieve the management change, each of which has a different mix of market and government inefficiencies. For example, the inefficiency due to a tax rebate will differ from a materials grant program or a subsidy payment program.

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 costs of market and government inefficiencies against the 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 costs of market or government inefficiency would need to be in order for a policy 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 costs of market or government inefficiencies exceed the threshold is then judged and the conclusions about the relative ranking of policy options or packages altered accordingly. Judgement is based on qualitative and quantitative information including 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. Therefore, threshold policy analysis provides a consistent methodology for judging relative policy effectiveness.

An example of how threshold policy analysis works

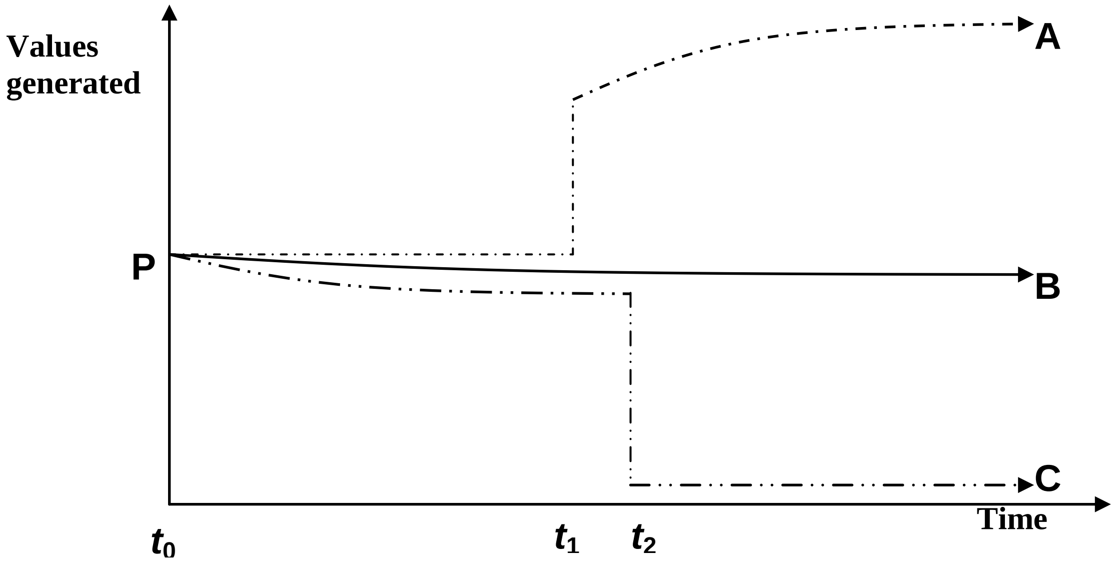
Assume a policy maker is required to develop policies for remnant vegetation conservation and is deciding between purchasing remnant vegetation and incorporating it into reserves or paying current landowners to manage remnants towards specified outcomes. The policy maker knows the costs of purchasing the remnants and the costs of their ongoing management as well as the losses inherent in tax collection. The policy maker would also know the likely average costs of private management from the bio-economic modelling process. The policy maker would also need to make judgements about factors of 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 assess the relative scales of the net impacts of market or government inefficiencies. The relative cost of each policy must then be compared against alternative policies and the net benefit range from changing management and assess the implications for policy selection and implementation.

4 Impact of environmental impact thresholds or irreversibility

The environmental management goal generated from the bio-economic model is not known with certainty. An important source of this uncertainty is the ecological data that were input into the model. In some instances, the behaviour of the system being modelled exhibits 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. The potential impact of these discontinuities is shown in Figure 1. Pursuing management strategy ‘A’ may achieve a positive environmental impact threshold being reached at t_1. Similarly, management strategy ‘C’ may lead to an irreversible decline in values at t_2.

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 achieve ‘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. The policy maker will attempt to reduce risk within the threshold policy analysis by adopting additional policies or expanding policy options. That is, more and larger policies will tend to 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 from achieving the policy goals. 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.

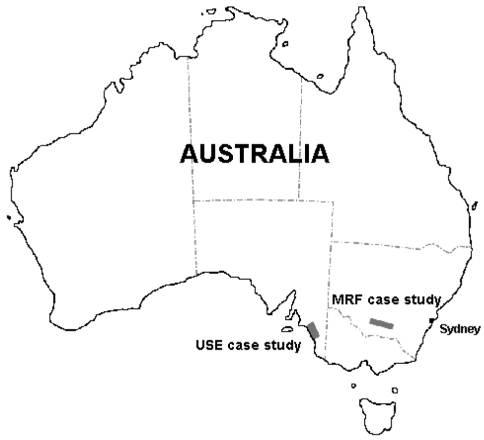
Part 2 Case study application

In this section the concepts developed above are briefly applied to two wetland case studies. Many of these results have been presented in greater detail in papers presented at previous conferences.[4] The results are used to illustrate the conceptual model discussed in part one and to demonstrate the importance of these aspects of policy development.

Case study areas

The policy discussion in this paper is illustrated by reference to two case study areas, the USE region of South Australia SA and the MRF in NSW. The approximate location of the case study areas is shown in Figure 1.

Location of case study areas



In the USE of SA he conversion of the landscape to pastoral production was motivated by the private values so obtained, mainly from pasture for agricultural production. Sixty-three thousand hectares of healthy wetlands, or less than seven percent of the original wetland area, remain in the USE region. The reduction in wetland area is further threatened by the impacts of dryland salinity. The reduction in

[4] The Research Reports describing the full findings of the Private and Social Values of Wetlands Research Project are also available from the authors or at the website: apsem.anu.edu.au/staff/jbennettr.html

wetland area has significantly reduced the private and social values generated by natural wetlands in the region.[5] The issue is whether the current net benefit is less than that which other management practices could deliver.

A similar pattern of wetland degradation has occurred on the MRF as a result of land and water management practices. 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 cropping 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 the 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 that present management.

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

5 Case study bio-economic models

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 1998). Supplementary information was gathered via a survey of wetland owners and managers in the case study area (Whitten and Bennett 2000b). The biological factors that drive these values were identified via the literature review and in consultation with scientists with expertise either in the region and/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 are likely to increase wetland values and provide underlying policy goals.

The next phase of the bio-economic modelling is to identify the impacts of potential management strategies on the biological states driving wetland values. To identify their impact the alternatives need to be compared to what would occur without changes to management – termed the 'business as usual' (BAU) case. Once a comparison point is established, an array of potential management strategies can be considered and quantified (see Whitten and Bennett 1999a). Five different and discrete management options were considered in the USE, namely:

- Improved management of existing wetlands – termed wetland retention (improved quality);
- Improved management of existing remnant vegetation – termed remnant retention (improved quality);

[5] These values include drought refuge for waterbirds from southeastern Australia, bird-breeding events, landscape appearance, recreation, fodder production and hunting.

- Conversion of agricultural pasture to wetlands – termed pro-wetlands (increased quantity);
- Conversion of agricultural pasture to revegetation – termed pro-remnants (increased quantity); and
- Large-scale adoption of farm forestry and other deep-rooted perennial species – termed targeted agro-forestry (improved quality).

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. The summary results of the biophysical modelling are shown in Table 1 (Whitten and Bennett 1999a). For example, changing wetland management to achieve the 'wetlands and remnants' option would involve reducing agricultural productivity by 257,700 dse but increasing the area of healthy wetlands by 28,400 hectares.

Table 1: Difference between 'BAU' and alternative strategies in the USE

Descriptive Attributes	**Unit**	**Wetland retention**	**Pro-wetlands**	**Wetlands and remnants**	**Cumulative farm forestry**	**Farm forestry alone**
Agricultural productivity	dse	-16,400	-79,800	-257,700	-341,100	-83,400
Farm forestry	ha	0	0	0	15,000	15,000
Environmental and management impacts						
Healthy wetlands	ha	12,600	25,300	28,400	31,600	3200
Healthy remnants	ha	0	0	51,300	51,300	0
Fencing required	km	450	950	2200	2400	100
Improved conservation status of species*	No.	15	17	22	22	0
Recreational impacts						
Number of ducks hunted	No.	3000	4800	5300	5800	1000
Total tourist numbers	No.	11,900	26,150	35,150	35,150	0

* Conservation status of threatened flora and vertebrate fauna species only

The outcome of the economic modelling and consolidation into bio-economic modelling is shown in Table 2 (Whitten and Bennett 2001a). The economic modelling shows that 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 $25.4m. However, when the additional costs of wetland management are considered ($22.9m) the option would generate a net loss to the community of $15.2m. Only 'wetland retention' generates net benefits to the community. An important aspect of bio-economic modelling is sensitivity analysis of the outcomes.

The last three rows of Table 2 show part of the sensitivity analysis performed within the bio-economic model. The conclusions from the bio-economic model are quite sensitive to the extrapolation of the non-market benefits generated by wetlands. A less conservative extrapolation generates a net benefit from all strategies except farm forestry alone. The results of the model are also quite sensitive to the number of endangered species that benefit. For example, if only half as many endangered species benefit then no management change 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 reintroduction is considered.

Table 2: Results of USE bio-economic model

Cost or benefit	Wetland retention	Pro-wetlands	Wetlands and remnants	Cumulative farm forestry	Farm forestry alone
Net change to agricultural producer surplus	-$1,166,000	-$3,210,000	-$13,368,000	-$12,517,000	$ 851,000
Management costs of wetlands and remnants					
Capital costs (earthworks, revegetation, fencing)	-$1,390,000	-$ 7,059,000	-$17,265,000	-$17,561,000	$ 221,000
Ongoing management costs	-$1,614,000	-$ 3,231,000	-$ 9,894,000	-$ 9,999,000	-$ 404,000
Sub-total	*-$3,004,000*	*-$10,290,000*	*-$27,159,000*	*-$27,560,000*	*-$ 751,000*
Environmental values generated – consumers' surpluses					
Duck hunting	$ 85,000	$ 220,000	$ 238,000	$ 257,000	$ 25,000
Tourism	$ 531,000	$ 972,000	$ 1,492,000	$ 1,492,000	$ 0
Non-use values	$8,029,000	$8,120,000	$21,217,000	$20,759,000	-$3,983,000
Sub-total	*$8,645,000*	*$9,312,000*	*$22,947,000*	*$22,507,000*	*-$3,958,000*
Environmental values generated – producers' surpluses					
Duck hunting	$ 17,000	$ 43,000	$ 46,000	$ 50,000	$ 5,000
Tourism	$ 750,000	$1,836,000	$2,367,000	$2,367,000	$ 0
Other wetland owner use values			*Not estimated*		
Sub-total	*$ 766,000*	*$ 1,879,000*	*$ 2,413,000*	*$ 2,417,000*	*$ 5,000*
*Total environmental values (conservative)**	*$9,411,000*	*$11,191,000*	*$25,360,000*	*$24,923,000*	*-$3,953,000*
Total changes valued	**$5,242,000**	**-$ 2,309,000**	**-$15,168,000**	**-$15,154,000**	**-$3,853,000**
Non-monetary benefits (less conservative estimates)*	$17,432,000	$17,664,000	$50,562,000	$49,288,000	-$12,341,000
Total net benefits (less conservative)	*$14,029,000*	*$ 6,043,000*	*$12,448,000*	*$11,627,000*	*-$11,668,000*
Total net benefits if 50% benefit to endangered species	*-$1,640,000*	*-$10,139,000*	*-$25,300,000*	*-$25,287,000*	*-$3,853,000*

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 survey non-respondents hold zero values and only extrapolate according to the survey response rate and only to SA residents. Less conservative assumptions extend the response rate to Victorian residents at 50 percent of the values held by SA residents (however, the underlying assumptions remain conservative). For more information see Whitten and Bennett (2001a).

A bio-economic model of MRF wetlands

A similar approach based on an initial literature survey followed by a survey of wetland owners and managers was followed in the MRF (Whitten and Bennett 1999b, 2000a). The key biological factors driving values can be summarised as the area (and type) of healthy wetlands (including size and healthy of buffer vegetation) and off-site water management. As in the USE, a comparison point of what would occur if 'business as usual' continues is established for comparison. Three different and discrete management options designed to improve wetland quality were then considered in the MRF, namely:

- Improved hydrological management of water – termed hydrological management;
- Improved management grazing practices in wetlands and buffer areas – termed 'grazing management'; and,
- Improved management of timber harvesting practices in wetlands – termed timber management.

Combining the three different options into a single strategy created a fourth option – 'combined strategies'. The summary results from the biophysical modelling of these options are presented in Table 3. For example, changing grazing management in the MRF would reduce agricultural production by 19,100 dse but increase the area of healthy wetlands by 6700 ha, the populations of wetland and woodland birds by 20% and the population of native fish by 25%.

Table 3: Difference between 'BAU' and alternative strategies on the MRF

Descriptive Attributes	Unit	Water management	Grazing management	Timber management	Combined strategies
Water transferred from irrigation	ML	41,700	0	0	41,700
Total agricultural production	dse	0	-19,100	0	-19,100
Sawn timber yield	m^3	0	0	-9,000	-9,000
Residual timber yield	m^3	0	0	-18,500	-18,500
Fencing required	km	0	700	0	700
Best information ecological outcomes of management changes					
Additional healthy wetlands	ha.	2,700	6,700	0	11,200
Additional wetland and woodland birds	%	33	20	20	75
Additional native fish	%	50	25	25	100

Note: Synergistic responses to management changes mean that the outcome of the 'combined strategies' is not simply the maximum of individual options or the sum of the individual options.

Table 4: Aggregate cost-benefit analysis of management strategies

Cost or benefit	Water management	Grazing management	Timber management	Combined strategies
Changes to agricultural activities				
Lost agricultural production	$ 0	-$3,136,756	$ 0	-$3,136,756
Cost of providing watering points	$ 0	-$ 191,549	$ 0	-$ 191,549
Lost timber production	$ 0	$ 0	-$4,677,775	-$ 4,677,775
Sub-total	*$ 0*	-$3,328,305	-$4,677,775	-$ 8,006,080
Management costs of wetlands				
Capital costs of water acquisition	-$18,161,201	$ 0	$ 0	-$18,161,201
Capital costs of wetland rehabilitation	-$ 1,151,129	$ 0	$ 0	-$ 1,151,129
Capital costs of fencing	$ 0	-$1,261,135	$ 0	-$ 1,261,135
Capital costs of wetland revegetation	$ 0	-$ 208,761	$ 0	-$ 208,761
Ongoing costs of wetland management	-$ 566,157	-$1,187,156	$ 0	-$ 2,072,250
Income from future water sales	$ 6,245,750	$ 0	$ 0	$ 6,245,750
Sub-total	-$13,632,736	-$2,657,052	$ 0	-$16,608,726
Environmental values generated – consumers' surpluses				
Recreation	$ 742,118	$ 1,841,551	$ 0	$ 3,078,414
Non-use values	$ 8,458,507	$ 9,211,723	$3,016,335	$11,832,400
Sub-total (Conservative)	*$ 9,200,624*	*$11,053,274*	*$3,016,335*	*$14,910,813*
Wetland owner use values		*not estimated*		
Total changes valued	**-$ 4,432,112**	**$ 5,067,917**	**-$1,661,441**	**-$ 9,703,993**
Non-monetary benefits (less conservative estimates)*	$26,602,000	$29,302,000	$7,093,000	$38,696,000
Total net benefits (less conservative)	$11,695,000	$23,724,000	$2,416,000	$13,215,000

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 survey non-respondents 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 (2001b).

Economic modelling of the changes to values that would result from the biophysical changes modelled in Table 3 are summarised within the bio-economic modelling results presented in Table 4. A similar conclusion can be drawn from Table 4 to Table 2. The costs of the production losses and wetland management outweigh the non-monetary benefits generated for all but one option, the ‘grazing management’ option. For example, the costs of acquiring and managing water under the ‘water management’ strategy ($13.6m) outweigh the non-monetary benefits generated ($9.6m). However, these results are also extremely sensitive to the degree of extrapolation of the non-monetary environmental values. The sensitivity tests of the MRF bio-economic model show that a less conservative extrapolation 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 4.

6 Case study policy development

Policies suitable for consideration in the USE and MRF must cost-effectively address the main elements of the costs or benefits from changing wetland management in the case areas. The degree of intervention with market mechanisms can also be ranked via use of Bromley’s (1997) division of policies between policies that facilitate, induce or compel change. In brief, 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. Facilitative policies seek to reduce (but cannot eliminate) market inefficiency, inducive policies reduce market inefficiency but at the cost of introducing at least some government inefficiencies and coercive policies remove market inefficiencies but are subject to higher degree of government inefficiency.

Policy threshold analysis involves assessing the likely scale of net benefits that would be generated via adoption of alternative policies relative to their implementation costs (due to market or government inefficiencies). The policy package that results should comprise the combination of policies that most cost-effectively achieves the closest outcome to the goal range that was determined from the bio-economic model. This trade-off involves elements of judgement about the scale of environmental values generated from changing management and 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.

Facilitative policies are subject to relatively few elements of government inefficiency and seek to reduce market inefficiencies. They are likely to rank highly in terms of cost-effectiveness but are unlikely to achieve the policy target in isolation due to remaining market inefficiencies. Therefore there is likely to be opportunity for cost-effective use of inducive policies that may generate a net benefit to the community but due to the greater degree of government inefficiencies that are introduced, are unlikely to achieve the policy target. Hence, an initial policy threshold analysis ranks facilitative policies as most cost effective, followed by inducive and coercive. More detailed ranking requires attention to the differences in the market or government inefficiency costs introduced by individual policies within each category.

Potential policies for changing wetland management in the USE and MRF

An array of policies that are likely to cost-effectively address the costs or benefits of management change in the case study areas was developed by Whitten and Bennett (2001c). These policies are applicable at alternative jurisdictional levels and are subject to different elements of market and government inefficiency.

At the local government level most important facilitative incentive is streamlining development applications that result in improved conservation (for example sales to conservation groups or rehabilitation activities). A local government rate exemption for conservation management would provide a strong signal to wetland owners of the importance placed on their wetland management by the wider community.

Facilitative incentives suggested at the state level include:

- strengthening information provision schemes to wetland owners interested in conservation management (such as the 'Land for Wildlife' scheme in Victoria and the 'Wetland Carers' scheme operated by Wetland Care Australia);
- encouraging farmers to complete farm management courses and plans;
- signalling the importance of wetlands in regional management plans and by listing wetlands at the appropriate state, national and international levels; and
- allowing private sector organisations to write and hold conservation covenants over wetlands (similar to Heritage Agreements in SA or Voluntary Conservation Agreements and Registered Property agreements in NSW – thus facilitating flexibility, innovation and competition in a protecting wetlands using conservation covenants).

More direct incentives at the state level include: consideration of exemptions on state government taxes and charges on land sales to conservation groups; and, direct subsidisation of the costs of changing wetland management or the ongoing costs of wetland management. It is also suggested that consideration be given to modifying current laws to simplify and reduce the costs of wetland rehabilitation.

Possible federal government incentives are focused on the potential for taxation incentives to reduce the costs of wetland management and increase the potential for additional contributions to wetland management via non-government organisations (such as Wetland Care Australia and Wetlands and Wildlife).

Other suggestions include the introduction of revolving funds and broader use of real estate tools by conservation groups in Australia.

The USE wetlands have significant potential to become an eco-tourism destination, however there is currently no promotion at the regional or state level. There may also be opportunities to benefit USE wetland conservation via management agreements with wetland owners over duck hunting that facilitate additional returns to wetland owners.

Rehabilitation of MRF wetland areas may be complicated by restrictions to construction works on floodplains and other development ordinances – it is suggested that these are streamlined to facilitate wetland conservation projects. Commonwealth tax incentives for construction of water storages should not apply where wetlands are destroyed in the process.

The State government manages many resources in the MRF including grazing and timber harvesting over about a quarter of the floodplain and water licensing and management. The State government should ensure appropriate pricing, ownership and management organisations are in place to manage these resources in the MRF.

7 Conclusions

The potential for bio-economic modelling to influence development of environmental policies is significant. Bio-economic models are able to assist with setting appropriate targets for management change that would maximise community welfare. Uncertainty remains in bio-economic modelling output from value ignorance, biophysical ignorance and preference changes over time. 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 developed from the bio-economic modelling framework are more difficult. The difficulty arises from incomplete knowledge about the market or government inefficiencies 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.

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 possibility can be considered in part by comparing likely policy costs against the target policy range suggested from the bio-economic model.

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