INFORMING NATURAL RESOURCE MANAGEMENT POLICY THROUGH COST–BENEFIT ANALYSIS:

A CASE STUDY OF SEDIMENT REDUCTIONS FROM RANGELAND GRAZING IN THE GREAT BARRIER REEF CATCHMENTS

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This thesis is presented for the degree of Doctor of Philosophy of CQUniversity

School of Business and Law

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Certificate of authorship and originality of thesis

The work contained in this thesis has not been previously submitted either in whole or in part for a degree at CQUniversity or any other tertiary institution. To the best of my knowledge and belief, the material presented in thesis is original except where due reference is made in text.

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Abstract

A key challenge in improving water quality entering into the Great Barrier Reef (GBR) from rangelands grazing is to understand the economic trade-offs in changing management practices and subsequent sediment reductions. In grazing lands in the Fitzroy and Burdekin catchments this can be achieved by improving land condition and reducing stocking rates to improve ground cover. A key conceptual problem facing policymakers is a lack of quantitative models and procedures to assess the costs and benefits of changing practices and to improve policy and program mechanisms.

A cost–benefit framework provides an appropriate methodology to estimate the trade-offs to achieving targeted reductions in sediments. To complete the framework the costs were estimated using a bioeconomic modelling approach and the community benefits of achieving sediment reductions were estimated through a choice modelling approach. The methodology integrated current policies and plans to consider the approaches in a policy framework.

The results bring together both the bioeconomic modelling and choice modelling, demonstrating that the community benefit for improved GBR health is significantly higher than the current level of government investment. The large heterogeneity in costs creates challenges in the complexity of administering efficient policy mechanisms but allows for clear prioritisation and targeting in the landscape.

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LIST OF PUBLICATIONS

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Star, M., Donaghy, P. and Rolfe, J. (2011). Economically viable land regeneration in Central Queensland and improved water quality outcomes for the Great Barrier Reef, *The Rangeland* Journal 33, 267-276.

Star, M., Donaghy, P. and Rolfe, J. (2011). Bioeconomic Modelling- Do we have the complete picture?, *Australian Agricultural and Resource Economics Society 55th Conference*, Melbourne.

Star, M., Rolfe, J, Windle, J. and Donaghy. (2013) Valuing agricultural pollutants emissions for improved Great Barrier Reef Health: A choice modelling case study approach. *Australian Agriculture and Resource Economics Society*, 57th Annual Conference Adelaide.

Star, M., Rolfe, J., Donaghy, P., Beutel, T., Whish, G. and Abbott, B. (2013). Targeting resource investments to achieve sediment reduction and improved Great Barrier Reef health, *Agriculture, Ecosystems & Environment* 180, 148-156.

Star, M., Rolfe, J., Long, P. Donaghy, P. (2014) The certainty of private benefits- Applying policy mechanisms for sediment reductions in the Great Barrier Reef catchments. *Australian Agriculture and Resource Economics Society*, 58th Annual Conference Port Macquarie.

CHAPTER ONE: INTRODUCTION

The Great Barrier Reef (GBR) is an iconic natural ecosystem comprising more than 2,900 reefs and covering an area of more than 348,000 km^2 (Haynes *et al.* 2007). The GBR extends for more than 2,300 km along Australia's north-east coast (Stoeckl *et al.* 2011) and is a World Heritage–listed area representing about 17 per cent of the world's coral areas. The GBR Marine Park includes extensive areas of reefs, seagrass beds and mangroves (Packett *et al.* 2009).

It is estimated that the GBR attracts approximately two million visitors per year for snorkelling, recreational fishing and other reef-related activities. These activities contribute approximately $3.5 billion annually to the Australian economy (Gordon 2007; Farr *et al.* 2011). There are, however, significant pressures on the GBR from poor water quality, climate change and fishing impacts, prompting government programs and policies to improve reef health.

In the early 2000s existing scientific evidence was collated, which led to the definition and agreement of issues affecting the GBR (Queensland Government 2009; Waterhouse *et al.* 2011). This led to agreement between the Australian and Queensland Governments that action was required to address water quality issues in the GBR and its catchments, resulting in the development of the *Reef Water Quality Protection Plan* (*Reef Plan*) in 2003 (Queensland Department of the Premier and Cabinet 2003). The goal of *Reef Plan* was to halt and reverse the decline of water quality entering the GBR within 10 years (i.e. by 2013) through reducing sediment, nutrient and pesticide loads. While the actions in *Reef Plan* were progressed to some extent in the first five years, it was not until 2008 that the Australian Government announced a major incentives program called 'Reef Rescue', which was designed to improve agricultural on-ground management practices. This incentive program resulted in large-scale changes in catchment management. *Reef Plan* was also updated in 2009 (Queensland Department of the Premier and Cabinet 2009) to include more specific targets and an additional overarching goal related to the long-term health of the GBR. The Queensland Government also introduced regulations to target sugarcane and grazing activities in priority areas to reduce the amount of

sediment, nutrients and pesticides discharged to the GBR (Queensland Department of the Premier and Cabinet 2009).

Land uses such as rangelands grazing and clearing of native vegetation for agriculture have increased pollutant transport to receiving water bodies to a level many times greater than the natural rate for oceans, coral reefs, lakes and reservoirs globally (Bejranonda *et al.* 1999; Ruitenbeek *et al.* 1999; Paris and Chérubin 2008), including the GBR in Australia (Fabricius and De'ath 2004; Gordon 2007; Haynes, Brodie *et al.* 2007). It is estimated that since European settlement, sediment and nutrient discharge levels from rivers have increased six- to nine-fold (Fabricius *et al.* 2011).

The *Reef Plan* targets of a 20 per cent reduction in sediment and a 40 per cent reduction in dissolved inorganic nitrogen are to be achieved from the grazing and sugarcane industries. Changes to management practices in both industries have been identified to achieve reductions in the pollutants (Queensland Department of the Premier and Cabinet 2009).

Sugar production is a prevalent agricultural industry in the reef catchments, operating over 5,700 km^2 in mainly the Mackay Whitsunday, Wet Tropics and Burdekin catchments (Figure 1.1) (Brodie *et al.* 2011). Sugar production systems use fertilisers – predominantly nitrogen – to improve soil fertility. However, in extreme rainfall events and particular soil types, there is nutrient run-off. Increased nutrient impacts the GBR through increasing phytoplankton blooms, which the crown-of-thorns starfish feed off, and is detrimental to seagrass health (Brodie *et al.* 2011). These confounding factors can result in decreased growth and survival and consequent changes to the coral population, structure and colony size (Haynes *et al.* 2007).

There is substantial interest in reducing discharges of sediments to the reef, particularly from the grazing industry (Karfs *et al.* 2009). Increased bare ground from grazing pressure contributes significant sediment when summer-dominant rainfall occurs and there is little ground cover to stabilise the soil (Karfs *et al.* 2009). Excessive sediment loads can impact on corals through smothering when particles settle out and by decreasing light availability, coral photosynthesis, and growth

(Fabricius and De'ath 2004; McKergow *et al.* 2005b; Bartley *et al.* 2014). There is particular concern regarding the condition of inshore reef health. Research has demonstrated that inshore reefs are collapsing under present land-use conditions, with sediment and nutrient run-off seriously affecting coral diversity (Fabricius *et al.* 2011).

Grazing is listed as the prime determinant of changes in water quality regarding sediment, with beef production accounting for the largest single industry by land use, covering 90 per cent of the relevant land area (Karfs *et al.* 2009). Across Australia extensive beef production contributes over $1 billion dollars to the national economy annually and employs over 9,000 people, many in rural communities (Gordon 2007).

The agricultural sector is a contributor to the declining water quality. However, the size and economic importance of the industry means it may be costly to reduce impacts. The focus of this study is on sediment reductions from rangelands grazing in the two largest catchments (Burdekin and Fitzroy – Figure 1.1) adjacent to the GBR.

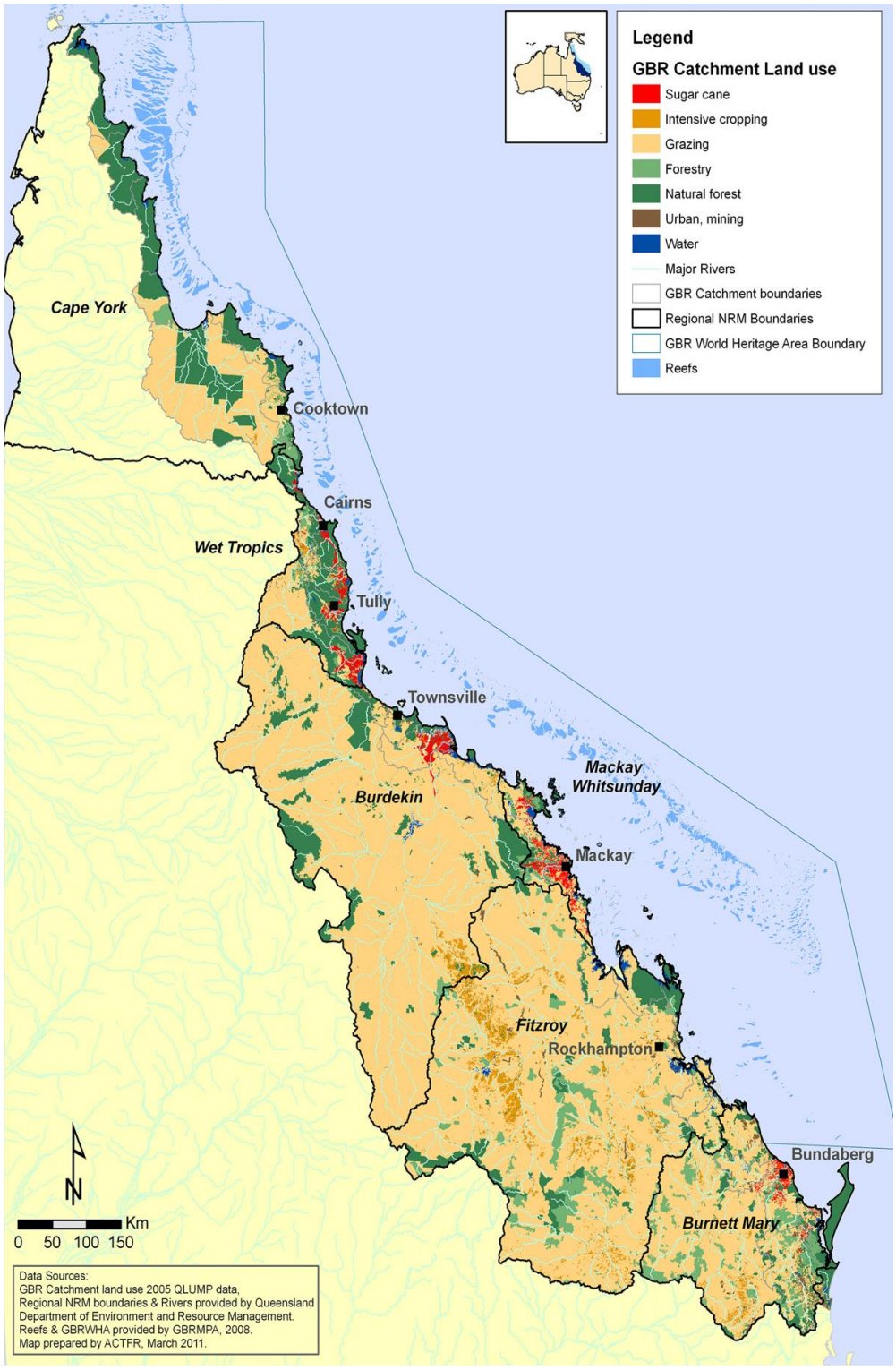

Figure 1.1 Map of the Great Barrier Reef catchments and reef position

1.1 Management practice change and certainty of outcomes

Improved management practices for land-based activities can improve water quality and improve the resilience of the GBR (Waterhouse *et al.* 2011). *Reef Plan*

(Queensland Department of the Premier and Cabinet 2009) actions are associated with improved land management practices, and the implementation of the 'Reef Rescue' program provided incentives to facilitate land management change. *Reef Plan* (2009) also set targets for reductions in sediment and nutrient pollutant loads through changing agricultural management practices, and improving catchment condition (Queensland Department of the Premier and Cabinet 2009).

The desirable management practices for grazing are primarily focused on sediment reductions. Improved practices include wet season spelling, fencing to land type, fencing off riparian areas, regenerating ground cover and retention of adequate ground cover through appropriate long-term stocking rates (Queensland Department of the Premier and Cabinet 2009).

Although there is clear evidence that sediment and nutrient are impacting on ecosystem and GBR health (Brodie *et al.* 2003b; McKergow *et al.* 2005b; Haynes *et al.* 2007; De'ath and Fabricius 2010; Brodie *et al.* 2011; Fabricius *et al.* 2011; Kroon *et al.* 2011; Stoeckl *et al.* 2011; Waterhouse *et al.* 2011), the marginal reduction in sediment and nutrients from improved practices has not been evaluated and there is uncertainty as to how much of a reduction will actually occur. Similarly, the marginal improvement in GBR health as a result of a marginal reduction in sediments or nutrients has not yet been clearly established (Fabricius 2010). These confounding factors present a level of uncertainty about whether targets will be achieved and the impact that the marginal changes will have on improving GBR health.

Extensive policies and programs have been developed to improve water quality through Federal and State funding arrangements (Queensland Department of the Premier and Cabinet 2009). In the past, government policy has been poorly implemented and often contradictory (Laurence *et al.* 2004). Recent programs, such as Caring for Our Country and the Natural Heritage Trust, have faced criticism due to the absence of measurable outcomes, lack of prioritisation and for the lack of success in combining biophysical and economic outcomes for making investment decisions (Pannell 2009a; Morrison *et al.* 2010; Pannell and Roberts 2010).

A core matter of concern has been the inefficiencies in selecting suitable projects for funding, which have been attributed to poor program design. Many government

support programs implement grants which involve a set payment for an action irrespective of the opportunity cost borne by the landholder, resulting in an inefficient allocation of funds (Rolfe *et al.* 2007). Other inefficiencies relate to the poor links between payments and actions, where most programs focus on inputs (such as the supply of riparian fencing) rather than on outcomes (improved water quality) (Rolfe *et al.* 2007). This therefore raises the issues of offering the correct support for the NRM problem and effectively allocating the investment.

There has been increased pressure for natural resource management (NRM) groups to spend money efficiently and demonstrate measurable outcomes to justify where funds have been allocated (Pannell 2009a). The knowledge required to do this involves understanding the interactions between biophysical components of the natural resource, opportunity cost and management implications. Determining which policy option allows targeted investment to achieve improved water quality outcomes requires understanding the associated costs and benefits (Bennett 2005a; Whitten and Bennett 2005b; Mazur and Bennett 2008).

Many of the costs and benefits associated with improved GBR water quality are not directly derived from markets. This means improvements are not able to be bought or sold in the open market. Costs can be inferred from market transactions, but benefits are difficult to quantify. To reduce sediment loads to an acceptable level is costly, involving public and private investments. The predicament of choosing between costly intervention measures now or risking unfavourable ecological and economic outcomes in the future is the dilemma currently facing Federal and State governments. To further inform these decisions, information regarding the costs and benefits of intervention measures is necessary.

Currently, there is limited understanding of the costs to improve water quality from grazing lands or the benefits that the improved water quality will generate. This thesis uses a cost–benefit analysis to evaluate the allocation of funds to achieve water quality improvement outcomes. It also assesses the benefits of water quality improvements and pollutant reduction initiatives.

In this thesis, the broad question is:

- What is the net social benefit of improving water quality for Great Barrier Reef health through changed grazing management practices?

Some of the key questions to be addressed include:

- What are the costs associated with reducing sediment from different land types within the Fitzroy and Burdekin Basins?

- What are the benefits from improving water quality resulting from management practice changes and achieving Great Barrier Reef water quality targets?

- What is the net benefit to the community of improving water quality?

1.2 Aim of the study

The aim of this study is to determine the net social benefit of improved water quality from sediment reductions to improve the health of the Great Barrier Reef. To address this aim, firstly a review was conducted of the relevant literature and past grazing studies, NRM policy mechanisms and non-market valuation studies leading up to the implementation of the current programs, and the underpinning science.

Secondly, based on this review, a bioeconomic model was developed to empirically estimate the cost of a tonne of sediment reduction generated by changing management practices from grazing lands in the Fitzroy and Burdekin Basins. Thirdly, a choice modelling experiment determined the value that society holds for improved water quality through changed management practices.

To achieve the aim of the study, the research:

- identified which land type and land characteristics offer the cheapest method of sediment reductions

- estimated the community benefit of water quality improvements from sediment reductions.

1.3 Research objectives

To improve the design of NRM policy mechanisms and level of funding to achieve improved health of the GBR, the core research objective to be addressed can be summarised as:

> To estimate the net social benefit of water quality improvements for the Great Barrier Reef from grazing lands to ensure this is reflected in the current level of government investment.

To understand this objective requires an understanding of the level of community benefit from increased protection, and the costs involved. To achieve this objective, two broad areas are addressed in this thesis. First, past NRM policies and sustainable grazing were examined to develop a bioeconomic model. A bioeconomic model allows the trade-offs for changing grazing management practices to be captured, identifying the costs of sediment reductions. A bioeconomic model enabled the costs of sediment reductions to be evaluated through combining the economic and biophysical aspects of grazing. Specific objectives are to identify how:

1. Land types and pasture utilisation levels impact the cost of sediment reductions
2. The variance of costs of sediment reductions impact on the required policy mechanism

The second major area of interest concerns the community (coastal towns along the Great Barrier Reef and the Brisbane population) that benefit from water quality improvements, where choice modelling experiments were developed to estimate community willingness to pay for pollutant reductions and subsequent improvements in GBR health. This then leads to the specific objectives of identifying whether:

3. The community willingness to pay for water quality improvements from changed management practices is higher than current investments.
4. The community willingness to pay for water quality improvements is higher than the costs of implementing the management changes.

By understanding these four broad objectives, the thesis will make a contribution to the net social welfare of water quality improvements from sediment reductions in grazing lands in the Fitzroy and Burdekin Basins.

1.4 Thesis structure

The structure of this thesis is as follows. Chapter Two provides background to previous NRM issues and a review of the policies and programs, followed by the GBR issues and how they link to the Fitzroy and Burdekin Basins and rangelands grazing. Chapter Three provides insights into the underpinning economic theory of NRM issues and the relevant policy mechanisms, as well as framing the bioeconomic modelling and choice modelling methodologies. The fourth chapter provides insights into previous bioeconomic models and the requirements to ensure the costs of sediment reductions are sufficiently captured. The fifth chapter explains the methods for the case study application, and the sixth captures the results of the bioeconomic modelling. Chapter Seven then describes the different approaches to estimating benefits for environmental goods and provides choice modelling theoretical underpinnings. Chapter Eight explains the case study methodology and Chapter Nine the results. Chapter Ten completes the cost–benefit analysis and subsequent policy recommendations, discusses the hypotheses and identifies future research opportunities.

1.5 Research method

In contributing towards informing future NRM policies and programs and gaining an understanding of the interaction between their costs and benefits, a framework was applied in this thesis. The framework draws on evidence and methodology for the following:

- review of past policies and programs of Australian NRM
- review of sustainable grazing practices
- estimating costs through completing a bioeconomic model
- estimating benefits through completing a choice modelling experiment

- implementing a cost–benefit analysis to estimate net social welfare.

This section outlines the research methodology and techniques adopted in this thesis. Clarifying these aspects of the research design assists to satisfy research principles that are important for maintaining an appropriate standard of research, which involves (Howe and Eisenhard 1990):

1. understanding one's values through alertness to, and coherence of, background assumptions

2. congruity between research questions and design

3. the effective use of relevant data collection and analysis techniques.

The research begins with a literature review of the sustainable grazing strategies for rangelands, rangelands economics and past NRM policies and programs. The study also reviews economic theories to determine the theoretical frameworks suitable for undertaking an assessment of the costs and benefits of improved water quality for the GBR from changes to grazing practices.

The next component of the research involves a series of case studies based on the Fitzroy and Burdekin Basins. In particular, the focus was on the different land type characteristics within the two catchments and the community that values the GBR.

The following sections outline the rationale for the case study approach adopted in this study.

1.6 Case study approach

The research design applies a hybrid methodological approach based on incorporating quantitative measures into a case study methodology. The quantitative measures refer to collection of data and interpretation of the results to determine the implications. The interaction between the case study approach and the quantitative measures allows for insights into the analysis of the case study, while the case study provides some degree of confirmation of the theories.

The case study of interest is the Fitzroy and Burdekin Basins, which are the two largest catchments for the GBR. The research questions were addressed by applying quantitative techniques, which were based on simulated data and property-level data, along with a survey of the general community. The use of measurement allows more precise estimates of the degree of relationship between concepts (Bryman 2004).

Case study research is at a scale and depth that can identify causal relationships and is appropriate for collecting evidence to support and contribute to development of contemporary economics for future NRM policies and programs. The use of case study research methods has been the foundation for much work in the social sciences, and research into the economics of grazing management at a regional case study level is considered appropriate for this study.

The quantitative measure aspect of the research allows concepts to be further explained and relationships fully explored. This research used quantitative measures to value the costs and benefits of sediment reductions. The framework allows further explanation of the concept of net social benefit and the relationship that occurs between the costs and benefits. The costs are estimated by simulated data and scientific formulas in conjunction with an economic case study. The benefits are measured from the responses to a survey in a choice modelling experiment.

The use of a case study approach and quantitative analysis involves a number of methodological issues that need to be considered; these are explained below.

1.6.1 Validity

Experimental design requires both internal and external validity. Internal validity refers to the authenticity of the cause and effect relationships, or how well the findings fit with reality. External validity is how the research applies to the external environment and whether the theoretical propositions arising from it are corroborated by other research.

The hybrid methodological approach adopted in this study offers a trade-off between single-unit analysis (with descriptive details, depth and emphasis on causal mechanism) and cross-unit studies (useful for causal inference, breadth of proposition, representativeness, emphasis on cause–effect, and ‘theory testing’ research strategy) (Gerring 2004).

The case study analysis in this research aims to contribute to an increased depth of understanding of the mechanisms underpinning NRM policies and programs in the context of the GBR. The research provides empirical evidence to inform future policies and programs of Australian NRM. Mazur and Bennett (2008, p. 3) support this approach:

> This integration of bio-physical modelling and economic valuation within the rigorous conceptual framework of benefit cost analysis affords a more complete information base on which NRM decisions can be made.

The present study seeks to add to this more complete information base. Given the objectives and constraints of the research, a case study approach was considered suitable to help inform and validate the theoretical and conceptual underpinnings of regional NRM policies and programs.

1.7 Net social welfare

A social welfare framework is implemented as the basis for estimating the economic value of environmental improvements from reduced sediment entering the GBR. A key factor in establishing an economic framework is to assess the net social benefit of sediment reduction from grazing lands. Developing such a framework poses several challenges. Firstly, not all environmental improvements associated with reduced sediment entering the GBR from grazing lands have readily observable markets and prices. Secondly, it remains difficult to value the changes of the choice being made. A measured dollar estimate is required of the cost of reducing sediment and the benefit gained by society.

Estimating costs allows biophysical and economic information to be incorporated into bioeconomic modelling. Choice modelling provides an estimate of the social benefit a community derives from improved water quality due to changed grazing systems. The interaction of and ability to estimate the demand and supply curves allow the cost–benefit framework to be completed and the net social benefit to be determined.

This research intends to contribute to the knowledge base used to determine policy and program design and to improve the ability of governments to target NRM investments efficiently for achievable and measurable outcomes.

1.8 Cost–benefit analysis

This study involved both a desktop simulation analysis[1] and the collection of data from the general public. Information used to inform this research was collected by a range of methods, including stakeholder engagement and field visits[2] in the study region.

The data for the study are reported in two sections. The first section determines the cost of water quality improvements from reduced sediment exported to the GBR through developing and implementing a bioeconomic model for grazing systems in the two catchments. These costs underpin the mechanism with the most efficient method of achieving NRM outcomes. The second part of the research used choice modelling to estimate the community benefit of sediment reduction. The value of these benefits informs the required amount of investment. The completion of the two sections allows for the data to be analysed in a cost–benefit analysis and the net social welfare of the water quality improvement to be estimated. The rationale for this methodology was to address the lack of empirical data on the net social benefits of NRM programs and policies.

The bioeconomic component of the research was developed through the integration of a biophysical model that simulated pasture growth and animal interaction and an economic model that explored the profitability of the enterprise under numerous biophysical scenarios. The bioeconomic model development followed some of the techniques described by Cacho (1997). There was integration between the two models for various parameters and variables, to enable an estimate of cost per tonne of sediment reduced to gain an understanding of how sediment can be reduced by particular management practices.

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[1] During the course of this study, the researcher was working with the Integrated Regional Planning Team in the Department of Agriculture Fisheries and Forestry, a lead state government agency responsible for at the time collaborating and delivering NHT2 and Caring for Our Country programs in Queensland as a part of state and regional NRM arrangements.

[2] Field visits in the Fitzroy and Burdekin catchments study area were conducted during the course of this research between 2009 and 2011. During this period, the researcher observed and participated in meetings with key NRM stakeholders ranging from different levels of government, regional organisations, industry peak bodies, researchers and landholders.

The choice modelling component of the research underwent lengthy research design to ensure that the available science on improved water quality was integrated into the survey. The choice modelling survey was collected through an internet survey that targeted a random sample of participants through the form of internet-based distribution. The experimental design of the choice modelling methodology allowed a benefit estimate to be estimated.

The study methodology allowed the relationship between the cost of sediment reductions and the benefit of improved water quality to be integrated in a cost–benefit analysis that explores the utility or welfare of society and enables a valuation of the net social benefit to occur on a locally specific level.

1.9 Ethical considerations

This research was carried out in accordance with the ethical guidelines prescribed by Central Queensland University. The purpose and intent of the study was overt and open to all participants. Participants were also informed that their involvement was voluntary and withdrawal from the study was without prejudice.

It was essential in the process of data collection that every effort was made to ensure the anonymity of key informants and to assure participants that any information provided was to be held in strict confidence in accordance with Central Queensland University ethical research policies.

CHAPTER TWO: UNDERSTANDING THE PAST AND FRAMING THE FUTURE

This chapter discusses the framing of past NRM issues and subsequent policies and programs. The chapter aims to provide insights into how NRM policies in Australia have been developed and their subsequent strengths and weaknesses. It explores the NRM issue of the declining health of the GBR and highlights the role of the NRM organisations relevant to this thesis: the Fitzroy Basin Association and NQ Dry Tropics. Finally, a review of rangelands grazing studies and previous work is given.

2.1 Natural resource management in Australia

In the 1970s the Federal Government introduced initiatives to create World Heritage sites in a number of areas (including the GBR). In the 1980s, scientific evidence indicated significant productivity loss in agricultural lands due to salinity, soil degradation, pollution of waterways and poor land management practices (Wallington *et al.* 2008). This gave rise to the Federal Government funding a program called *National Landcare Program*, which was launched in 1989.

In 1992 the *National Strategy for Ecologically Sustainable Development* was designed and implemented to underpin future government NRM policies and programs. Under this strategy, the Federal government committed to three core objectives:

- Enhancing individual and community wellbeing and welfare by following a path of economic development that safeguards the welfare of future generations
- Providing for equity within and between generations
- Protecting biological diversity and maintaining essential ecological process and life support systems.

In 1996 the Natural Heritage Trust (NHT) program was announced. Based on a partnership agreement between the Federal and State Governments it operated with

funding investments matched by both governments. The intent of NHT was to stimulate investment in conservation and sustainable use of environmental, agricultural and natural resources into the future (Crowley 2001). The framework of the NHT also fostered partnerships between communities, industry and government and saw the first generation of regional NRM groups. NHT was phased out in 2008 (Wallington *et al.* 2008).

Caring for Our Country (CfOC) is the latest national NRM program and was implemented in 2008 and completed in 2013. A budget of $2.25 billion over five years has been allocated for six articulated priority areas. The program was developed using a more business-like approach through the development of a business plan, with outcomes on a structured timeline.

Table 2.1 provides a timeline of programs previously implemented and the approach taken.

Table 2.1 Timeline of programs implemented in Australia

Program	Time frame	Approach	Funds
National Landcare Program (NLP)	1989–97	Individual and property-level change via extension and education; limited formal evaluation	$340 million
Natural Heritage Trust 1 (NHT1)	1997–2001	As above, with addition of competitive devolved grants to local community groups for small projects	$1.25 billion
Natural Heritage Trust 2 (NHT2)	2002–08	Funding for regional bodies with approved plans, further participatory evaluation and some monitoring of general trends in resource condition	$1.75 billion
Caring for Our Country (CfOC)	2008–13	Base-level funding for regional bodies plus competitive grants for priority targets, a wider range of organisations and a new stewardship package	$2.25 billion

2.2 Review of past natural resource management programs

2.2.1 National Landcare Program

Landcare groups were based on catchments or neighbourhoods, rather than interest groups, and were initiated by the community with support from government. The responsibility and ownership of projects were held by the community rather than the government (Landcare 1997).

The program was based on education and changing attitudes. It focused on self-help initiatives, such as the provision of information, planning assistance, capacity-building opportunities and market-based incentives in order to internalise environmental and social costs (Wallington *et al.* 2008). The Federal Government commitment to Landcare groups for land conservation activities and property planning at a total cost of $340 million over 10 years.

Curtis and De Lacy's (1996) review of the program found that landholders benefited from being able to share ideas, work together to solve problems, have opportunities to learn about land management, have the capacity to plan at a property and at a catchment scale, and obtain technical information which was otherwise not available to them. It was also suggested that peer pressure may have encouraged adoption of management practices and overcome the free-rider problem. Their review found that participants in Landcare:

- were significantly more aware of many land degradation issues then previously, particularly those that are less obvious
- reported significantly higher levels of knowledge of resource topics
- had significantly higher levels of adoption of almost all best practices surveyed than before participating in the program.

2.2.2 Natural Heritage Trust

The Natural Heritage Trust (NHT) focused on integrated conservation, sustainable use and repair of biodiversity, on which Australia's long-term economic prospects depend. The premise for the capital spending was to achieve NRM and sustainable agriculture outcomes consistent with national strategies, by developing partnerships across all tiers of government and the community (Crowley 2001).

The stated purpose of the NHT was:

> … to stimulate significant investment in the conservation, sustainable use, and repair of Australia's environmental, agricultural, and natural resources into the 21st century. The NHT will provide a framework for strategic capital investment in the natural environment, achieve complementary environmental, natural resource management and sustainable agriculture outcomes consistent with agreed national strategies and will foster partnerships between the communities, industry and all levels of government. (DPIE & Environment Australia 1997)

Following a report by the Industry Commission 2000 (Wallington *et al.* 2008) and a report by Landcare in 1997 (Landcare 1997), which both suggested strengthening of regional; and catchment-level organisations, integrated NRM at a regional scale became the preferred approach for addressing sustainable development in Australia (Wallington *et al.* 2008). The framework of the NHT acknowledged these recommendations and fostered partnerships between communities, industry and government and saw the first generation of regional NRM groups. 90 per cent of NHT was allocated to rural and regional communities, between 40 per cent and 60 per cent of funds bypassed state and local government and went directly to community-based projects (Crowley 2001). A further $1,400 million was committed by the federal and state governments to water quality and salinity in the *National Action Plan for Salinity and Water Quality* (*NAPSWQ*).

A review of NHT and Landcare (Curtis and Lockwood 2000) suggested that state-sponsored community participation can work successfully if a number of criteria exist: (1) the separate roles of regional bodies and local organisations are transparent, (2) established relationships and collaboration between all stakeholders exist, (3) there are benefit-based cost-sharing mechanisms for public investment on private property, and (4) there are flexible policy packages and economic incentives for landholders to supply public benefits (Curtis and Lockwood 2000; Kahn *et al.* 2005).

In a review of *NAPSWQ* and its implementation, Morrison *et al*. (2004) identified that fragmentation of policy implementation between Federal and State Governments, and within State Governments, obstruct better NRM in Australia. Through the structure and portfolio of governments, certain agencies or departments

have lead roles in particular policy objectives, some of which are different or competing (Morrison *et al.* 2004). The authors recommended that to improve NRM programs and policy effectiveness, strategic structural changes are required along with rationalisation of organisations and entities. Responsibilities from state organisations should be delegated regionally, with participatory decision-making and inclusive information exchange between all entities involved (Morrison *et al.* 2004).

The NHT was similarly criticised in regards to achieving desired goals because investments were not prioritised with integrated biophysical and economic data (Pannell 2009a):

> Environmental problems are often technically complex and uncertain. Sound decisions about their management [are] required to be based on good knowledge regarding (a) the degree of threat or damage to environmental assets at risk, and (b) the extent to which this threat or damage can be reduced by particular changes in management. In many cases, generic knowledge is not sufficient – we need locally specific knowledge. (Pannell 2009a, p. 3)

The NHT was also criticised by (Pannell 2009a) for the outcomes that were not achieved as the policy mechanisms were not appropriate for the circumstances, and the incentives and program rules did not encourage program managers to cost-effectively pursue environmental outcomes.

2.2.3 Caring for Our Country

In 2008, a newly elected Federal Government announced Caring for Our Country (CfOC), a natural resource program to replace the NHT. A budget of $2.25 billion over five years was allocated. The goal outlined by CfOC was:

> Caring for our Country is the Australian Government's new environmental management initiative. It aims to achieve an environment that is healthier, better protected, well managed, resilient, and provides essential ecosystem services in a changing climate. (Australian Government Land and Coasts 2008, p. 2)

The program identified six national priority areas to improve strategic outcomes from July 2008 to June 2013. These included the National Reserve System; biodiversity and natural icons; coastal environments and critical aquatic habitats; sustainable farm practices; natural resource management in northern and remote Australia; and community skills, knowledge and engagement (Australian Government Land and Coasts 2008, p. 3).

For each of these national priority areas, target outcomes to be achieved by 2013 were stated. These outcomes include:

- expand the area that is protected within the national reserve system to at least 125 million hectares (a 25 per cent increase), with priority to be given to increasing the area that is protected in under-represented bioregions (Australian Government Land and Coasts 2008, p. 5).
- Increase, by at least one million hectares, the area of native habitat and vegetation that is managed to reduce critical threats to biodiversity and to enhance the condition, connectivity and resilience of habitats and landscapes (Australian Government Land and Coasts 2008, p. 11).
- Reduce the discharge of dissolved nutrients and chemicals from agricultural lands to the Great Barrier Reef lagoon by 25 per cent (Australian Government Land and Coasts 2008, p. 17).
- Reduce the discharge of sediment and nutrients from agricultural lands to the Great Barrier Reef lagoon by 10 per cent (Australian Government Land and Coasts 2008, p. 17)
- Assist at least 30 per cent of farmers to increase their uptake of sustainable farm and land management practices that deliver improved ecosystem services (Australian Government Land and Coasts 2008, p. 25).

Pannell (2009a) explored the prospects for CfOC and highlighted the changes in priorities from the NHT program; salinity, in particular, was viewed in 2000 as a 'national crisis', yet under CfOC it did not attract a single project focused on the issue. Pannell (2009a) also suggested that intended outcomes should not only be clearly articulated but also be based on sound and comprehensive asset value, levels and timing of environmental damage; the technical and socio-economic feasibility of reducing damage; and costs. This, however, has not occurred. Pannell's (2009a)

recommendation of a more business-like approach was incorporated with a business plan outlining an approach to investment, with outcomes, priorities and improved accountability.

Morrison *et al*. (2010) reviewed the CfOC process and noted that it included a broader range of policy tools, with less reliance on extension and small grants and more emphasis on intervention around key assets. CfOC also tendered funding to a wider range of organisations and only provided base funds to regional NRM organisations. The program also streamlined previous NRM programs and promised to focus on monitoring, evaluation, reporting and improvement (MERI).

CfOC, however, has been criticised as the expectations set by the business plan may be unrealistic relative to the amount of funds available (Morrison *et al.* 2010). As a result of these limited funds, only 57 out of the 1,300 projects submitted in the first year were able to be funded. The second area of concern was the lack of transparency to determine targets and investment priorities. The criteria used to allocated funds lacked transparency in regards to how the projects met the business plan objectives. Finally, the role of local government had reduced further and appears to be excluded from decision-making processes in relation to NRM; this may hinder long-term community engagement and capacity building (Morrison *et al.* 2010).

2.2.4 Great Barrier Reef programs and targets

In 2003 under the NHT Program (ongoing commitment to the GBR was continued through CfOC) a *Reef Water Quality Protection Plan* (*Reef Plan*) was developed by the Australian and Queensland Governments and updated in 2009 and again in 2013 (Queensland Department of the Premier and Cabinet 2009; Queensland Department of the Premier and Cabinet 2011). *Reef Plan* states the goal of "halting and reversing the decline in water quality entering the Reef within 10 years" (Queensland Department of the Premier and Cabinet 2013 2000, p. 7). A key objective was to reduce the load of pollutants (sediment, nutrient and pesticides) from diffuse sources entering the GBR. The strategies outlined for landholders include:

- self-management approaches

- education and extension
- economic incentives.

(Queensland Department of the Premier and Cabinet 2013)

Reef Plan outlines self-management approaches for landholders to implement practices through programs such as best management practices, property resource management planning, and environmental management systems (Queensland Department of the Premier and Cabinet 2013). Education and extension services are identified in the strategy to encourage collaboration between government departments and landholders to increase the adoption of sustainable agricultural practices. Through this extension work, the plan targets, as a priority, sediment contributions from grazing cattle in high risk catchments (Queensland Department of the Premier and Cabinet 2003). The target for sediment reduction is a 20 per by 2020 (Queensland Department of the Premier and Cabinet 2013).

As an extension of *Reef Plan* and in line with the national priority areas of CfOC, 'Reef Rescue' was formed. Reef Rescue has the objective "to improve the water quality of the Great Barrier Reef lagoon by increasing the adoption of land management practices that reduce the run-off of nutrients, pesticides and sediments from agricultural land" (Queensland Department of the Premier and Cabinet 2009, p. 27).

Over $2 billion has been allocated to fund this objective, through the following five main components:

- Water Quality Grants ($146 million over five years)
- Reef Partnerships ($12 million over five years)
- Land and Sea Country Indigenous Partnerships ($10 million over five years)
- Reef Water Quality Research and Development ($10 million over five years)
- Water Quality Monitoring and Reporting, including the publication of an annual Great Barrier Reef Water Quality Report Card ($22 million over five years).

(Queensland Department of the Premier and Cabinet 2009, p. 27)

The Water Quality Grants and Reef Partnerships components have been key for funding on-ground practice change administered through the respective NRM groups, which already had activities addressing water quality.

Reef Plan identifies the Fitzroy and Burdekin Basins as 'high risk' for the categories of biophysical risk, social risk, development risk and risk to marine industries. The *Great Barrier Reef Protection Amendment Bill 2009* (Qld) was introduced to form a regulatory structure that aims to reduce the impact of agricultural activities on the water quality entering the GBR and contribute to *Reef Plan*. The legislation requires landholders to change land management practices for improved water quality and reef health, through activities such as monitoring of bare ground at the end of the dry season, completing an environmental risk management plan and further record keeping.

The legislation is only for three of the catchments for the GBR: the Wet Tropics, Burdekin and the Mackay Whitsunday catchments. The legislation applies to commercial sugarcane and grazing businesses greater than 2,000 ha, and has resulted in additional employees to provide extension services, regulation and training (Queensland Department of the Premier and Cabinet 2009).

2.2.5 Regionalisation and natural resource management

Since 2000, responsibility of public expenditure per annum has been devolved to 56 catchment-based NRM regional bodies. The organisations have played an increasingly important role in regionalised NRM issues and programs through taking an Integrated Catchment Management approach to prioritising issues and making investment decisions (Jennings and Moore 2000). 'Integration' involves collating information for prioritisation and planning. This information can take a number of forms, including biophysical, economic, social and local knowledge, and elements of these decisions may be made by staff, the board, peak committee, subcommittees or working groups (Seymour *et al.* 2008). The collation of information then forms the basis for 'regional investment strategies' for NRM groups in Queensland, and are termed similarly in the other states (Pannell *et al.* 2008).

In 2002 with the development of NHT2, regional bodies were required to develop a 'strategy and investment plan'. This involved identifying assets; ranking and prioritising assets after accounting for risk; establishing and prioritising goals, objectives and targets for realistic achievement through the investment planning process; and consulting the public (Farrelly and Conacher 2007).

These tasks are complex and require expertise in gathering and using science and information, local and practical knowledge, and an understanding of public values. These skills and knowledge are reported to be lacking (Seymour *et al.* 2008), but are critically needed to make prioritisation decisions. The spatial and temporal impact of a decision can vary dramatically from an individual field to a whole region and from a year to a whole century, based on the complexity and geography of the NRM issue.

These information gaps can pose be a constraint on the quality of decision making by a regional body (Seymour *et al.* 2008). A survey completed by 18 regional bodies identified that there was limited use of economic and social information. Integration of information was also lacking, and evaluation and tracking of outputs and activities was minimal. This reflects the lack of clear reporting guidelines to government funders and provides an opportunity for a framework to be developed (Seymour *et al.* 2008).

A key challenge for NRM groups in developing their strategy and investment plans was the critical gaps in the resource condition information base. This resulted in large numbers of NRM groups proposing investment in data collection. All plans set long-term resource condition targets and short-term management actions; however, the links between the asset resource base and the subsequent short-term management strategy was depicted with various levels of clarity. The complexity of the plans highlight the importance of sharing resource information between governments, scientists and NRM groups (McAlpine *et al.* 2007).

Integrated resource management has been the approach for many NRM programs to be implemented as it provides an integration of community involvement, technical knowledge and organisational structure. This collaboration approach provides a more adaptive and flexible approach to address uncertainty, complexity and

interconnectedness associated with NRM then previous programs. However, there is little information and few resources dedicated to the evaluation of the approach, as a result of the complexity associated with difficult trade-off decisions made in integrated resource management. The difficulty arises because NRM programs provide intangible services with broad social functions, which makes it impossible to measure or calculate the marginal economic benefits associated with the program (Bellamy *et al.* 1999).

2.3 Rangelands management in the Fitzroy and Burdekin Basins: a case study

The two largest and grazing-dominant catchments are the Fitzroy and the Burdekin – which are the case study catchments for this research. The challenges facing sustainable rangelands management in these case study catchments are investigated, through reviewing previous research to gain an understanding of the causes of degradation and the limiting factors in regeneration and through modelling pasture and animal production.

A key driver of adoption of best management practices is the economic implications and trade-offs. Currently, there is limited knowledge of the economic components of NRM issues, with little information on the biophysical and economic drivers of degradation, or economic insight into the most efficient methods to correct the problem through improved policies and programs (John *et al.* 2005; MacLeod and McIvor 2007; House *et al.* 2008; Roebeling *et al.* 2009; MacLeod *et al.* 2010). Australia's protected areas make up of only six per cent of the continent. Thirteen per cent of land is privately owned, and approximately 75 per cent of agricultural activity takes place on long-term government leases. Positive cooperation and active management by landholders is required to achieve effective conservation (Curtis and Lockwood 2000; Department of Sustainability Environment Water Population and Communities 2002; Earl *et al.* 2010).

2.3.1 Fitzroy Basin Association

The Fitzroy Basin has an area of 156,000 km^2, encompasses the tributaries of the Mackenzie, Isaac and Connors, Dawson, Comet and Nogoa Rivers and occupies one-tenth of the land mass of Queensland. All the tributaries enter into the Fitzroy River,

which then drains into the World Heritage–listed GBR. The Fitzroy Basin Association (FBA) also covers adjacent catchments of the Boyne and Calliope Rivers (which drain directly to the GBR), although they are not strictly part of the Fitzroy River system (Figure 2.1).

The basin is dominated by the use of grazing (90%), while also incorporating broadacre grain cropping and irrigated cotton (6%) (Karfs *et al.* 2009). The region also has mining as a key industry, with the Bowen Basin coal reserves contributing more than half of the region's income. The climate is sub-tropical and semi-arid, with highly variable summer rainfall and drought being a reoccurring feature.

The basin has experienced extensive modification, with clearing of Brigalow (*Acacia harpophylla*)–dominated woodland communities for grazing and cropping. By 1996, approximately 60% of all remnant vegetation had been cleared or substantially altered, impacting significantly on the amount of soil run-off (approximately doubling it) exported from native vegetation (Packett *et al.* 2009). These characteristics, combined with the impacts of the coal, cropping and grazing industries, have raised concerns about water quality and the present and future health of the GBR (Webster 2008).

Recent estimates of modelled post-development, long-term annual suspended sediment export from the Fitzroy River Basin to the GBR lagoon range from three to four and a half million tonnes per year (Packett *et al.* 2009; Waterhouse *et al.* 2011). The key source of sediment pollutant entering the GBR is an increase in bare ground from grazing lands in the catchments. Karfs *et al.* (2009) also recognised that increased ground cover particularly at the end of the dry season and improved land condition can prevent excessive amounts of sediments entering streams and rivers. With such heterogeneity between land types regarding soil characteristics, land productivity and slope, the sediment exported varies significantly throughout the catchment and its land types (Silburn 2011; Silburn *et al.* 2011b).

Figure 2.1 Fitzroy Basin

The FBA's regional investment strategy, *Central Queensland Strategy for Sustainability 2004 and beyond* (Christensen and Rogers 2004), encompasses four key areas: (i) regional aspirations, targets and actions, (ii) implementation, (iii) monitoring and evaluation, and (iv) a regional profile. The plan sets a number goals focused in these key areas for the region's assets.

To achieve these goals, targets were determined and prioritised, programs were devised, partnerships identified, actions stipulated, and how the targets would be measured was also articulated. A number of programs were developed, such as the biodiversity and vegetation program, corals and coasts program, healthy region program, healthy rivers, waterways and wetlands program, protecting our heritage program, salinity program, sustainable landscapes program, and water allocation and management program (Christensen and Rogers 2004).

Acting on *Reef Plan*, the Fitzroy Basin Association released the *Fitzroy Basin Water Quality Improvement Report (*Fitzroy Basin Association 2008). In this report, the Association sets its long-, intermediate- and short-term objectives. The report promotes a self-management approach strategy through education and extension and identifies voluntary adoption of best management practices as the best way to

improve water quality. An objective of best management practices is to encourage optimal pasture utilisation rates to improve the condition of land that has chronic low ground cover and of land types susceptible to erosion. The report also identifies a short-term goal of reducing suspended sediment concentrations to 13 mg/L at the high peak flow of the wet season by 2014. Current concentrations are at 19 mg/L (Fitzroy Basin Association 2008).

The report has a strong emphasis on improved water quality for the health of the GBR, allowing for land asset and water asset management. The FBA has reported its success in encouraging landholders to participate in programs such as best management practice workshops, projects such as fencing to land type, and increasing the area of protected land (Fitzroy Basin Association 2010).

2.3.2 NQ Dry Tropics

The Burdekin Basin (Figure 2.2) consists of six sub-catchments: Belyando, Bowen Brocken Bogies, Cape Campaspe, Lower Burdekin, Suttor and Upper Burdekin, covering 135,188 km^2. The two main land uses, commercial sugarcane and grazing, use 87% of the area and include diverse tropical landscapes such as semi-arid drylands, wooded grasslands, mountains, tropical rainforests, coastal plains and wetlands.

NQ Dry Tropics is a community-based, not-for-profit company established in 2002 to deliver land and water management activities and enhance the community's involvement in issues throughout the region. NQ Dry Tropics has invested in on-ground actions to improve awareness, attitude, practice change, and the way the catchment is managed. The *Burdekin Natural Resource Management Plan* (Board 2005*)* was developed with a number of targets for the grazing industry. These targets were 50 per cent of 'fair condition' land to be increased to 'good condition' land, 20 per cent of 'poor condition' land to be increased to 'fair condition', and 50 per cent of 'very poor' condition land to be rehabilitated by 2024.

A broader vision of outcomes and targets were also developed include:

1. Freshwater, estuarine and marine ecosystems are ecologically healthy, productive, resilient, enjoyed and valued
2. Terrestrial ecosystems are sustainably managed for good water quality
3. Surface and ground water are sustainably managed for good water quality
4. Less pollutants in surface and ground water (sediment, nutrients and pesticides)
5. Hydrological conditions are able to support functional aquatic systems, appropriate water levels and productive uses
6. Rivers and wetlands are in good ecological condition.

Figure 2.2 Burdekin Basin

The Burdekin Catchments NRM organisation, NQ Dry Tropics, has also released targets to be achieved by 2014 (Burdekin Dry Tropics Board 2005). To reach these targets, programs that the Fitzroy Basin and NQ Dry Tropics have taken part in

include the NHT programs, the National Action Plan for Salinity and Water Quality (NAPSWQ), and, more recently, Caring for Our Country programs (CfOC).

2.6 Land degradation and grazing management

Forty-three per cent, or 3.2 million km^2, of Australia is rangeland, and degradation has occurred on a significant portion of this land (McKeon *et al.* 2004). Degradation can be described as the loss of 'desirable' perennial grasses and shrubs (in relation to ability to feed livestock), increased susceptibility to erosion, soil structure decline and infestation of woody weeds (McKeon *et al.* 2004). A recent survey found that 44 per cent of northern Australia's rangelands had experienced some change and 12% was badly affected by soil deterioration and dominance of undesirable species (McIvor 2012). Rangelands consist of relatively low-level soil fertility and poor soil structure, therefore when extreme weather patterns occur (drought or intense wet periods) Australia's rangelands are vulnerable to rapid deterioration (McIvor 2012).

The pastoral industry in northern Australia is characterised by low input, low output enterprises that operate in a harsh and variable climate which is high in variation of both annual and seasonal rainfall. Rainfall is summer-dominant (Ash and Smith 2003), leading to large fluctuations in productivity and quality of herbage (MacLeod *et al.* 2004). Pastures consist mainly of native species, and perennial grasses to provide valuable ground cover, which prevents soil decline and erosion. Drought and heavy grazing, either singularly or in combination, can result in the decline and death of perennial pasture species (McIvor 2001).

During the dry periods as the pasture species deteriorate and there is increased bare ground rangelands are vulnerable to soil movement when heavy rain does occur. This soil movement eventually enters into waterways and is exported to the GBR lagoon in the case of the Fitzroy and Burdekin (Karfs *et al.* 2009).

There has been increased research into sustainable rangelands management with a focus on pasture species and grazing management. Stocking numbers and management strategies have proven to be the most significant variables affecting long-term productivity and sustainability. The importance of stocking rates had resulted in a number of studies concerning rangelands degradation and viability

(McIvor and Monypenny 1995; Ash and Stafford Smith 1996; Chilcott *et al.* 2005; Hamilton *et al.* 2008).

The correlation between stocking numbers and land degradation was realised early in Australia's pastoral history. The inability to match appropriate stocking numbers to pasture availability and to take account of seasonal fluctuations has been commonly cited as the reason for the collapse of perennial pasture species (Epps and Crittenden 1992). Recent studies have contributed to further understanding the key components in land degradation (Table 2.2). Chilcott *et al*. (2005) define pasture utilisation as 'the proportion of potential pasture growth that is consumed by livestock'. It is from this definition that the safe long-term (5–10-year) carrying capacity is calculated (Chilcott *et al.* 2005).

Table 2.2 Recent literature on grazing land management approaches

Paper reference	Research design	Variables included	Research conclusions
Ash *et al.* (2002)	Grazing trial, Northern Australia, Dalrymple	Stocking rate, burning, wet season spelling, property development	• Grazing management is the main variable affecting land condition • Early wet season spelling resulted in a higher rate of pasture utilisation and enabled increased cash flow to be allocated to increased watering and fencing infrastructure
Northup *et al*. (2005)	Grazing trial, Northern Australia	Stocking rates, basal area, vegetation, soil	• Increased grazing pressure led to less standing crop, and soil properties such as carbon and nitrogen were more widely dispersed
Campbell *et al.* (2006)	Rangelands grazing strategy Bio-economic modelling	Environmental variability and predictability Degradation and land type resilience Property rights, discount rates and market stability and prices	• A conservative stocking strategy is optimal to reduce degradation and environmental variation • Conservative stocking rates are optimal when there is high risk in rainfall and weather events • If there is prolonged drought (1 year or longer), stock fetch low prices due to the contributing factors of poor condition, market prices and high supply • Discounts rates and market stability both impacted on stocking strategy and the variation in economic return that can occur.
Orr and Yee *et al*. (2006)	Grazing trial Central Queensland	Stocking rate, burning, rainfall, black spear grass	• Exclusion of stock for short periods of time (of up to 12 months), especially during winter and in years when rainfall is average or below, will not ensure pasture condition with perennial native species improves • Rainfall was a significant variable in pasture recovery

Paper reference	Research design	Variables included	Research conclusions
Macleod and McIvor (2008)	Case study, Northern Australia, in monsoonal grasslands and subtropical woodlands	Riparian fencing, tree clearing and stocking rate	• Riparian fencing for the monsoonal grassland case study had overall more public benefits (environmental benefits) than costs, although it was an expensive task for the landholder to undertake alone • Tree clearing resulted in large economic returns but there were also large negative environmental impacts
O'Reagain (2009)	Grazing trial in Charters Towers	Stocking rates	• Live weight gain in the heavy stocking rate enclosures was lower per head, and there were increased costs of drought feeding and management costs in years of low rainfall • Lighter stocking rate had good individual production performance and did not require drought feeding
Orr *et al.* (2010)	Grazing trial, Central Queensland Black spear grass	Stocking rates, burning	• 30% pasture utilisation or 4 and 5 ha per steer was the sustainable pasture utilisation rate • Light spring burning also encouraged seed recruitment
Scanlon *et al.* (2013)	Bioeconomic modelling based on grazing trial in Charters Towers	Stocking rate, climate, profitability	• Moderate stocking rate proved to be the most profitable over the 30-year time frame explored

Rangelands are complex ecosystems which vary significantly depending on a number of variables such as the characteristics of the land type, rainfall and land-type resilience. However, the results of previous research demonstrate a strong correlation between stocking rates ensuring that land degradation does not occur. This previous research also reflects the heterogeneity of land types in Queensland and the challenges through a number of variables that landholders are faced with regarding land degradation and regeneration.

2.7 Land condition

Land condition changes have often been explained as ecological responses to changes in pasture composition and animal production (Ash *et al.* 1995). Extreme pressure on rangeland resources through over-grazing has the potential to have severe consequences for these resources and their future productivity, both economically and ecologically (MacLeod and McIvor 2008). Inappropriate grazing strategies, particularly in response to climatic variability, has resulted in the depletion of native grasses and decline in land condition (MacLeod and McIvor 2007).

Land condition has been defined by the Grazing Land Management framework (Chilcott *et al.* 2005) as the capacity of land to respond to rain and produce useful forage and is a measure of how well the grazing ecosystem is functioning. The ABCD land condition framework, developed by Meat and Livestock Australia (MLA) in partnership with the then Queensland Department of Primary Industries and Fisheries, classifies land condition through pasture species composition, weeds, woodland thickening, bare ground and soil condition. Table 2.3 provides a summary of the land classification.

Table 2.3 Land condition classification

Land condition classification	Perennial grasses. Described as palatable, productive and perennial (3P)	Bare ground	Weeds	Soil condition	Woodland thickening
A	Good coverage	Less than 30% in most years	Few weeds and no significant infestations	Good, no erosion, good surface condition	No sign, or only early signs
B	Some decline in 3P grasses and increase in other less-favoured species	More than 30% but less than 60% in most years	Increase in less-favoured grasses or weeds	Some decline, some signs of previous erosion and current signs of erosion	Some thickening in density of woody plants
C	General decline of 3P grasses, large amounts of less-favoured species	Greater than 60% in most years	Large amounts of less favoured species	Obvious signs of past erosion and/or susceptibility currently high	General thickening of woody plants
D	General lack of any perennial grasses or forbs			Severe erosion or scalding resulting in hostile environment for plant growth	Thickets of woody plants cover most of the area

Source: (Chilcott *et al.* 2005)

As grazing pressure increases and land condition declines the level of bare ground increases. This then makes the surface vulnerable to further degradation when summer dominant rainfall occurs and the soil is suspended as sediment into waterways (Karfs *et al.* 2007). McIvor (2001) explored regeneration of land in C and D condition at Hillgrove near Charters Towers for three years with the exclusion of stock as the method of regeneration. A criterion was developed to predict the capacity of over-grazed pastures to regenerate by relating pasture performance during the regeneration phase to initial pasture condition. The impacts

of regeneration on both native and sown pasture species resulted in the regeneration being dependant on growing conditions as well as the exclusion of stock (McIvor 2001). In the trial, the areas that consisted of fertile soil regenerated from C condition in two to three years and from D condition in three or more years through the exclusion of stock. This demonstrated that grazing management and soil fertility are important determinants of land regeneration capacity and recovery periods (McIvor 2001).

2.8 Summary

In this chapter the relevant literature regarding NRM policies and programs has been reviewed, framing the current issue for reducing sediment emissions onto the GBR. The chapter progresses through the various issues of land degradation, grazing management, and land condition. Attention now shifts to the underpinning economic theory regarding NRM issues.

CHAPTER THREE: ECONOMIC UNDERPINNING OF NATURAL RESOURCE MANAGEMENT ISSUES

The focus of this chapter is to understand why NRM issues occur, the available policy mechanisms and underpinning theories of utility and welfare. The chapter reviews past policies and frames why the approach of a cost–benefit analysis has been taken. It highlights the importance of bioeconomic modelling and choice modelling to estimate the costs and benefits to complete the analysis.

3.1 Utility

Sustainability has been defined as meeting the needs of the present without compromising the ability of future generations to meet their own needs. This would be satisfied by maintaining welfare into the future (Randall 2007). Welfare refers to the level of utility or satisfaction an individual consumer gains from a commodity or basket of commodities, and is described as the concept of satisfaction. Benefit is the notion of actual gain. However, utility is a subjective concept and it is not obvious how much satisfaction is gained from an individual's consumption. Furthermore, it is not obvious how to compare utility among individuals. To be able to express consumer choice and preference, Johansson (1991) suggests assumptions must be made:

- Completeness: when the consumer can decide which combinations of commodities are preferred and which yield equal satisfaction
- Consistency: the consumer's preferences are transitive or consistent. For example, an individual consumer prefers combination A over combination B and combination B over combination C. To be consistent, that consumer must also prefer combination A over combination C
- Non-satiation: the individual consumer always prefers more to less

- Convexity: one unit of good must be forgone to obtain one unit of another good.

The relationship between the level of consumption and the satisfaction gained forms a utility function. The Cobb–Douglas utility function (F(x)) for two commodities is:

$$U = F(x) = x_1^{\alpha} x_2^{1-\alpha} \qquad \text{(Equation 1)}$$

Where x_1 and x_2 are two different commodities, α is a positive parameter ($0<\alpha<1$) and U is the level of satisfaction obtained by the individual consumer. A utility function allows comparisons to be made between any number of combinations. An indifference curve represents combinations of two commodities between which the individual consumer is not concerned, or is indifferent, about the combination. It is downward sloping because of non-satiation; that is, more of one good must be obtained to compensate for a reduction in the availability of the other good if the individual consumer is to remain indifferent. Indifference curves are strictly convex to the origin; the assumption of convexity is synonymous with an assumption of marginal rate of substitution (MRS) between commodities. This is the number of x_2 that must be forgone per unit of x_1 gained if the individual consumer is to continue to obtain the same level of satisfaction. MRS can be measured by the slope of the indifference curve at a particular point. For example, in Figure 3.1 MRS can be calculated by the change in x_1 divided by the change in x_2.

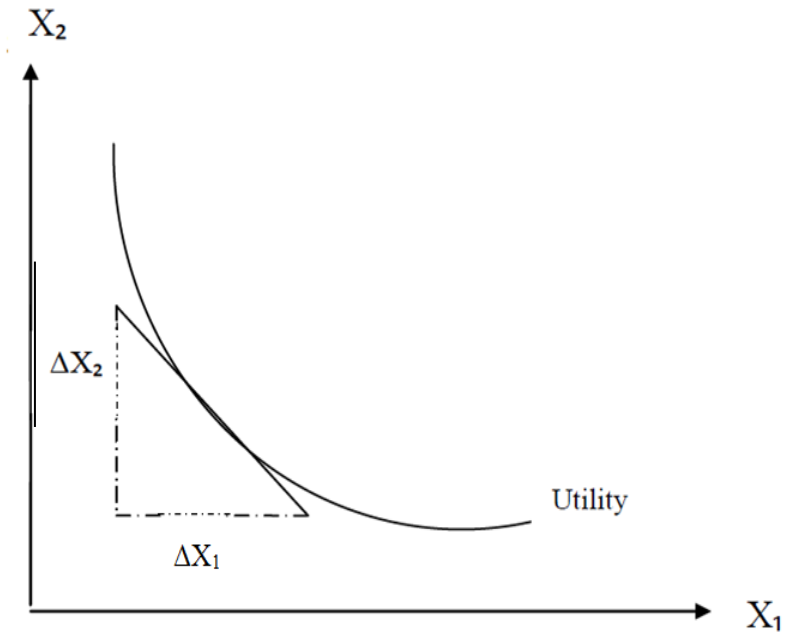

Figure 3.1 Marginal rate of substitution

Source: (Johansson 1991)

As the income of the individual consumer increases, the budget line will move out to the right as higher quantities of x_1 and x_2 can be obtained. Following the law of demand, consumers demand more at a lower price, and derive utility from consuming this good. This is the overarching concept for the price–quantity relationship (Pindyck and Rubinfeld 2005).

Marginal utility is the extra satisfaction from consuming one more unit of the good at a particular time, providing all other goods consumed remain the same. Following the theory of diminishing marginal returns, marginal utility usually decreases with each extra unit of consumption (Johansson 1991).

The extra utility of an additional unit of a good is measured by the unit price, with the measurement relative to other goods. The assumption that the utility from each dollar of expenditure is constant is known as the assumption of constant marginal utility of income. Rational consumers will purchase the good so long as utility gained exceeds the cost. It is this willingness to pay (WTP) for a good that is a pecuniary term to express the maximum benefit of obtaining a good. A consumer's WTP is influenced by the prices of other goods and income (Johansson 1991).

It can be assumed that a positive preference for a good will show up in the WTP; however, each individual's WTP will differ. To determine what is socially desirable, individual WTP can be aggregated to secure total WTP. While consumers would not pay for something they do not want, it cannot be assumed that WTP as measured by market prices measures the benefit to either individuals or society. This is because there may be individuals who are willing to pay more then the market price for a good. This extra benefit they obtain is consumer surplus.

In Figure 3.2, the market price of P_E is determined by the forces of demand and supply. P_E is the market price for everyone, where Q_E is supplied, since it is not possible to charge a different price to each individual buying the good. However, individual A is willing to pay a higher price, P_A, for the same good. As the market price is only P_E, individual A obtains the consumer surplus shaded above P_E and under the demand curve. The total area shaded represents the total benefit obtained, with the triangle representing consumer surplus received by individual A and the rectangle representing the total expenditure of all individuals on the particular good. These two areas together represent total benefit. In this particular case it is assumed that the demand curve is Mashallian and therefore income is held constant (Pearce and Turner 1990).

When utility is held constant and only price and compensation substitution effects are accounted this is termed a Hicksian demand curve. As income is not accounted for Hicksian demand curves are usually steeper then Mashallian, and biased estimates form WTP results from using Mashallian rather than Hickisan demand curves to measure consumer surplus (Pearce and Turner 1990). This bias estimate depends on the size of the income effect associated with the price change. When the income effect is small this bias can be ignored in Cost-benefit Analysis (CBA), particularly, when the price change of the good in question accounts for a relatively small component of total consumption.

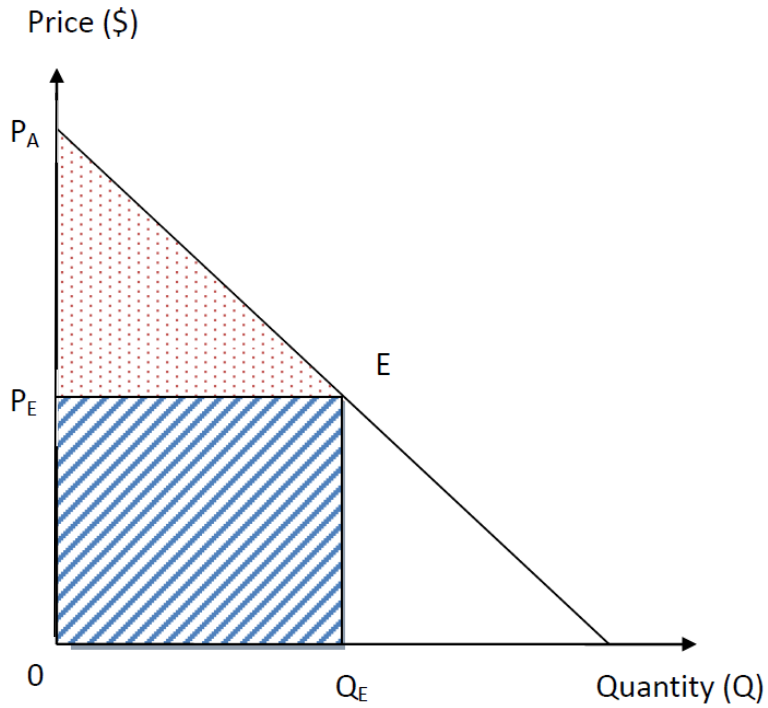

Figure 3.2 Demand curve for environmental goods

(Pearce and Turner 1990)

With environmental goods, the demand curve does not clearly exist as there is no market. Therefore, environmental economists have developed methods to estimate the economic values encompassed in the demand curve. Use values or benefits are derived from actual use of the environment, such as by anglers, bush walkers and beach-goers, who use the natural environment and derive benefit from it. Non-use values are slightly more complex, as these values are essentially an expression of preference or a WTP for the preservation of an environment even if it not used. The types of non-use values can be classified in terms of a) existence value, b) altruistic value, and c) bequest value. Existence refers to the WTP to keep a good in existence although there is no intention of using the good. Motivation for this may include a feeling of concern for the asset itself (e.g. a threatened species) or a 'stewardship' motive, where a responsibility is felt for the asset. Altruistic refers to when an individual is concerned that the good in question should be available to future generations. A bequest value is similar, but the concern is that future generations should have the option to make use of the good (Pearce *et al.* 2006).

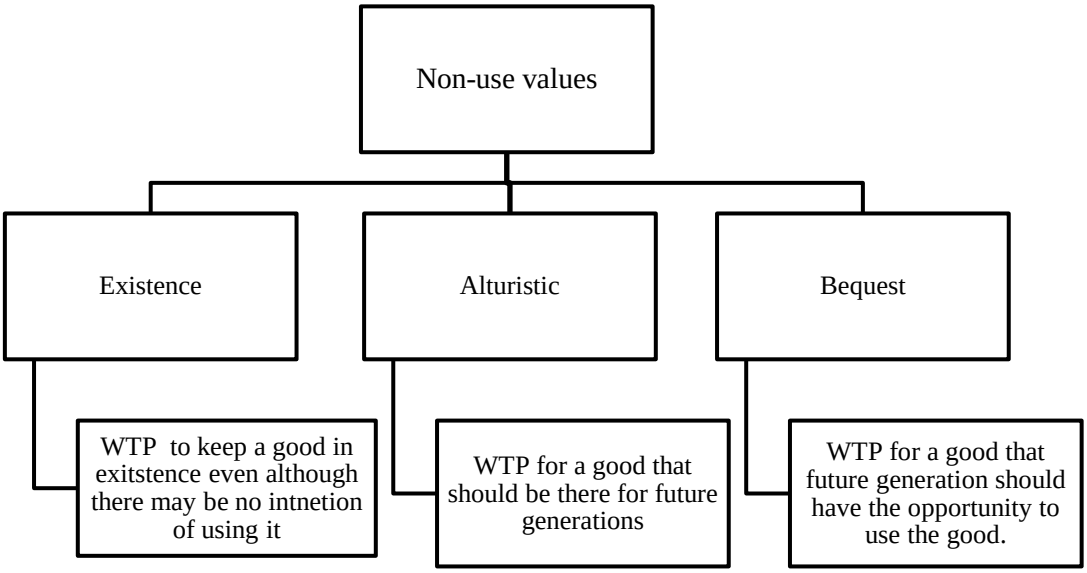

Figure 3.3 Non-use values

Total economic value decomposes these use and non-use values. This allows for an understanding of the changes in society's wellbeing due to a project or policy that has environmental impacts. Total economic value can be characterised differently according to the type of economic value arising. Techniques to values goods will be explored later in Chapter Seven.

3.2 Evaluating policy

The economic value of all goods, including environmental goods and services, is determined by the personal utility yielded. The preferences of individuals are revealed by the choices they make, and efficiency and consistency reflect rational behaviour. The 'basic theorem of welfare economics' aims to legitimise individual rational behaviour as socially desirable. This then justifies government intervention to improve the conditions under which individuals make choices. Government intervention is especially justified in cases of market failure and when the markets do not maximise collective welfare. Determining the value of a good held by society is pivotal in evaluating policy mechanisms.

3.2.1 Welfare theory

If there were only a single individual in the economy, the individual's utility could be used to rank different social states. For example, if the individual prefers state A

to state B, ‘society’ makes the same choice. However, reality is that society has millions of households with different tastes, preferences and income. Some individuals may gain from a policy decision such as a move from state A to state B, while others lose from the move. Therefore, to estimate the desirability of a policy for society as a whole, the preferences of all individuals must be aggregated to arrive at a meaningful welfare estimate (Johansson 1991).

Normative economic policy is based on three main principles: welfare, the Pareto principle and distributional equity. The goal of welfare theory is to allocate resources efficiently; this is termed the Pareto criterion (Bullock *et al.* 1999). By this criterion a policy change is socially desirable if everyone is made better off or at least some are made better off while no one is made worse off. However, when the possibilities of making such policy changes are exhausted, society is left with an allocation of commodities that cannot be altered without someone being made worse off. If there is a net improvement, this allocation is called Pareto-optimal. According to the Pareto criterion, all changes that improve or deteriorate welfare can be evaluated. However, to value the situation where some individuals are gainers and some are losers would involve comparing the increased satisfaction from the gainers and the decreased satisfaction from the losers, making the Pareto ranking of states incomplete as seen in Figure 3.4 (Johansson 1991).

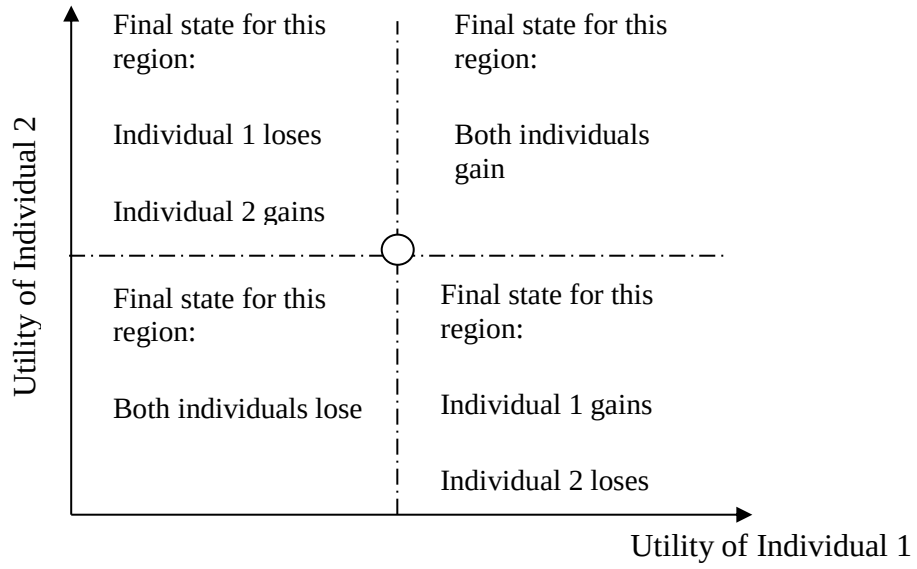

Figure 3.4 Pareto criterion

A change that makes at least one person better off and no-one worse off is a Pareto improvement. Consider Figure 3.5. The X and Y axes represents utility to Individuals 1 and 2 respectively. Shifting from 0 to F represents more utility to Individual 1 than 0, and shifting from 0 to N represents more utility than shifting from 0 to F for Individual 1. Similarly, shifting from 0 to G represents more utility for individual 2 than 0, and moving from 0 to M represents more than moving from 0 to G. The amounts of utility cannot be compared between individuals on these axes, so although the levels of utility at G and F look to represent the same amounts this is not necessarily true.

The line M to N represents all the utilities that combinations of two goods can produce given the current level of inputs and technology (Figure 3.4). At N, the inputs are allocated to produce goods and services that maximise the utility of Individual 1. At M, all inputs are allocated to maximise the utility of Individual 2. If it is assumed the economy is at A, with several policies and programs under consideration, any alternative that allows this two-person community to move within triangle ABC is a Pareto improvement. As the utility of at least one individual increases and of the others does not decrease. Any alternative that moves the economy into the shaded areas cannot be compared to A, as the changes in utility cannot be strictly compared, and one person is worse off. Moving into 0GAF makes both individuals worse off, so moving into ABC is the only Pareto improvement that can occur (Sinden and Thampapillai 1995).

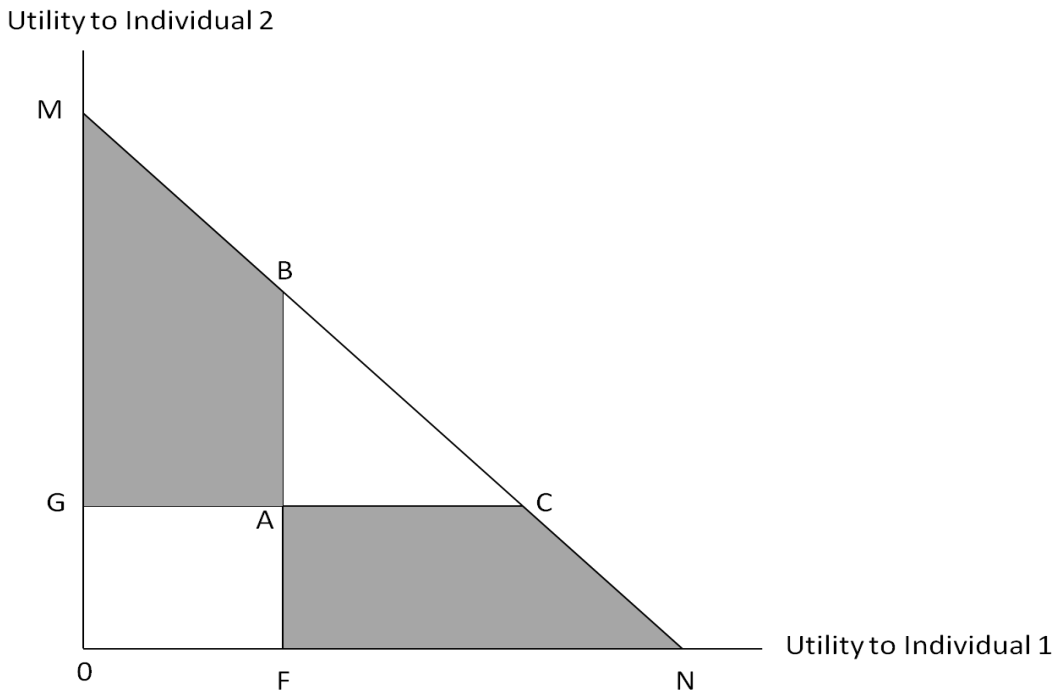

Figure 3.5 Pareto improvement in a two-individual choice

(Sinden and Thampapillai 1995)

Just about all resource allocations, programs and policies result in one individual being worse off in some regard, so the Pareto improvement principle does not apply. To overcome this issue, Kaldor and Hicks modified the principle. Kaldor created the principle that alternative *x* is preferable to the existing situation if the gainers could compensate the losers and still be better off. Hicks developed the criterion that alternative *x* is preferable to the existing situation if the gainers could bribe the losers and still be better off. These modifications are now known as the Kaldor–Hicks criterion, which says that economic welfare can be increased so long as the gainers can potentially compensate the losers. It is important to note that the payment of compensation is not required for the criterion to be satisfied: even if the result of a project makes somebody worse off but the overall project generates net benefits, then it should be undertaken (Johansson 1991). If compensation is actually paid, and the net sum of gains is greater than the sum of losses, this is a Pareto improvement.

Consider Figure 3.6 where Individual 1's utility is measured on the horizontal axis (U1) and Individual 2's utility is measured on the vertical axis (U2). The initial distribution of income enables Individual 1 to obtain utility level $U1_a$ and Individual

2 to reach utility level $U2_a$. By redistributing income from Individual 1 to Individual 2, a negatively sloped curve occurs (PP). PP is known as the utility possibilities frontier, and as we move along the curve from point A to point B the utility of Individual 2 decreases as the utility of Individual 1 increases.

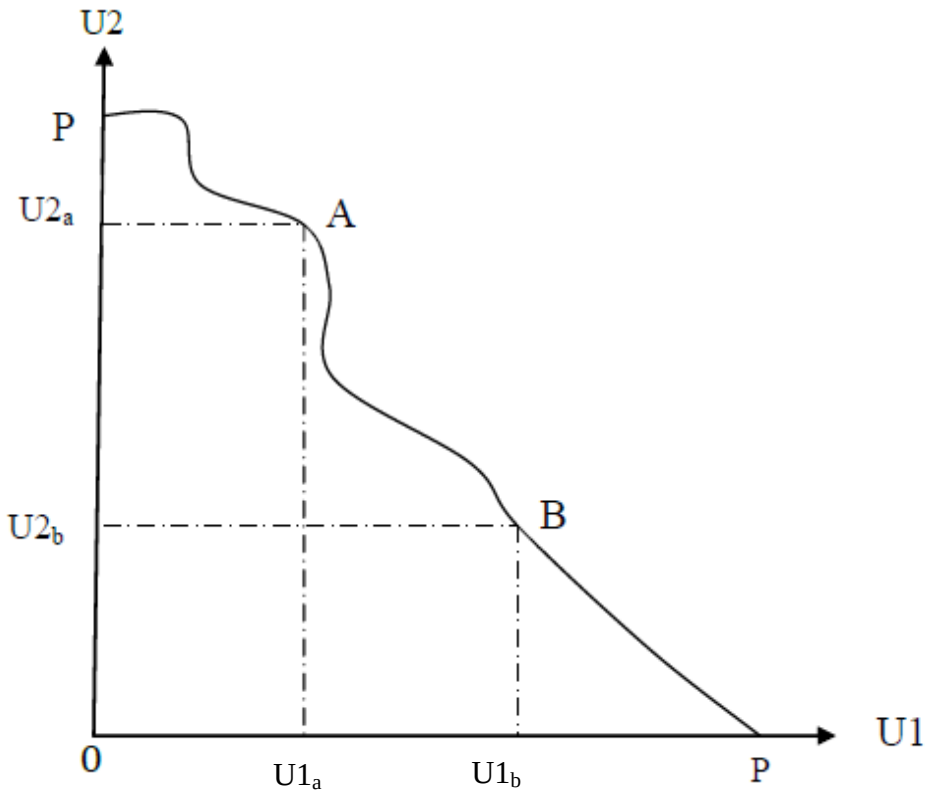

Figure 3.6 Utility distribution

The utility possibilities frontier is the general equilibrium, and redistributing purchasing power between individuals would possibly affect all prices; that is, prices must change to restore equilibrium in all markets. This is illustrated in Figure 3.7 where Individual 1 faces two different budget lines corresponding to different incomes and prices. Budget line AA is the initial budget line, and BB is the budget line after a particular redistribution. All points on the utility possibilities frontier satisfy the Pareto condition and any movements along the curve result in a decrease in one individual's welfare for an increase in another's individual welfare.

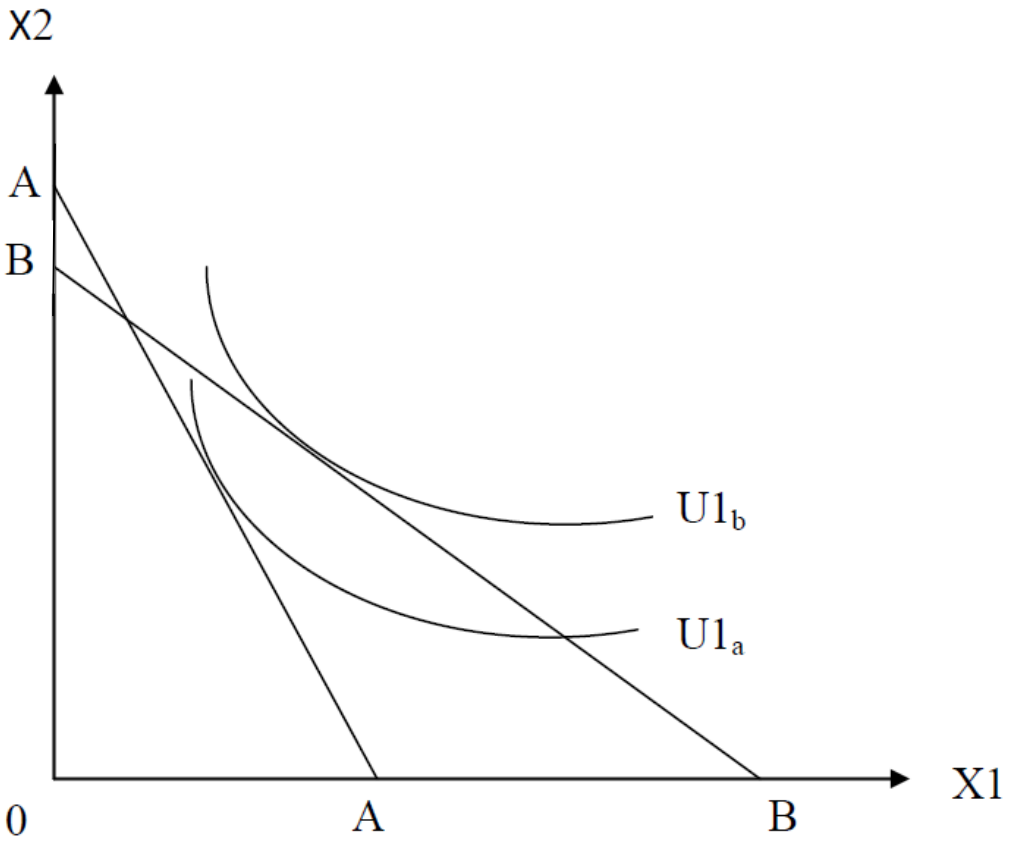

Figure 3.7 Redistribution of income and utility

To illustrate the Kaldor–Hicks criterion, consider Figure 3.8. Assume the economy is initially at 0, which is situated inside the utility possibilities frontier, therefore the economy is suffering from an inefficiency or distortion. A move to point A, B or C would be a welfare improvement for either Individual 1 or Individual 2 or both individuals. Any movement in the area of 0A and 0C and the production possibilities curve would improve welfare for both. Conversely, a movement from 0 to D would not represent a Pareto improvement as it would result in reduced welfare for Individual 2. However, when point D is reached the income in the economy could be redistributed to ensure that everyone is better off than at point 0. This meets the (Kaldor) compensation criteria and could be labelled as an improvement, if compensation is paid from Individual 1 to Individual 2.

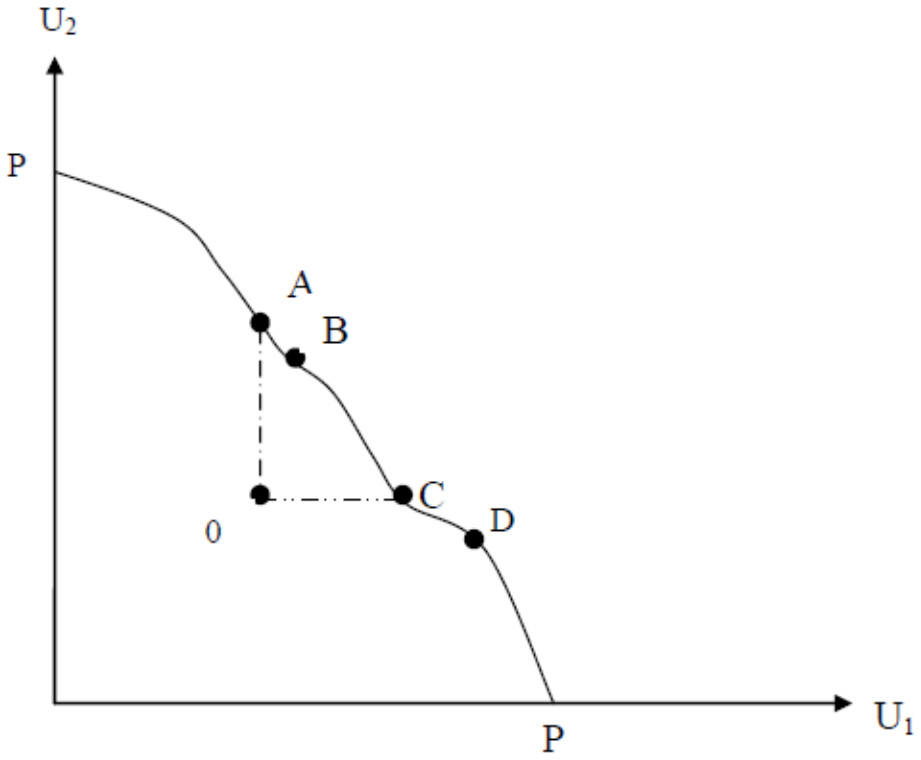

Figure 3.8 Kaldor-Hicks compensation principle

Although understanding in theory the impact of policy on welfare, Pareto principle and distributional equity to apply to a policy issue presents complexities and challenges. Bullock *et al*. (1999) assert there are five challenges to conducting a normative policy analysis. These include (i) modelling the economy, (ii) estimation of model parameters, (iii) estimation of welfare, (iv) choosing a set of policies to be examined, and (v) applying value judgements that rank the examined policies. These challenges give rise to the implementation of cost–benefit analysis, as explored in the following sections.

3.3 Cost–benefit analysis theory foundations

Cost–benefit analysis is a methodology derived from social welfare economics for determining appropriate policy actions and the social benefit of a decision (Grimble and Wellard 1997). CBA offers a process of evaluating costs and benefits of an investment to identify the option with greatest net social benefit.

In a competitive market, benefits are assessed through people's WTP for goods, and costs are the compensation to producers for generating the goods. It is assumed that

producers strive to maximise profits and, following the law of supply, are willing to increase output if the price rises. Individual producers define costs as the monetary amounts required to obtain their inputs, and are true resource costs to them. However, the cost to society of a particular quantity of input is the earnings next best use as these rescourses' are what is being foregone. This foregone return may accrue to a different firm or activity but would still accrue to society as a whole. Therefore, the true social cost of using an input in alternative *x* is the potential return of the input in the foregone alternative use of *y*. This return is opportunity cost and is defined as the potential earnings of the best alternative for that input (Sinden and Thampapillai 1995).

The theoretical foundations of CBA can be briefly summarised as:

- Preferences of individuals are determined by the benefit value. For an individual's wellbeing, welfare or utility to be higher in state A than B is to indicate that the individual prefers A to B.

- Preferences are measured by WTP for a benefit and willingness to accept compensation for a cost.

- It is assumed that individuals' preferences can be aggregated so that social benefit is simply the sum of all individuals' benefits. The cost is the sum of all individuals' costs.

- If a beneficiary from a change can hypothetically compensate the losers for a change, and have some net gains left over, then the basic test that benefits exceed costs is met. This is the Kaldor–Hicks compensation test explained previously in section 3.7.

The basic decision rule for accepting a project or policy is:

$$\{\sum WTP^{G}_{i,t} \cdot (1+S)^{-t} - \sum WTA^{L}_{i,t.} \ (1+S)^{-t}\} > 0 \qquad \text{(Equation 2)}$$

Where:

i = the individual

t = time

WTP = the measure of WTP to secure the benefit

G = the gainers

L = the losers

WTA = the willingness to accept or be compensated

S = the discount rate selected

(Pearce *et al.* 2006).

It is important to consider that a *private* CBA aims to maximise private net benefits, or determine where the private cost is outweighed by the private benefit. On the other hand the aim of *social* CBA is to maximise social net benefits, or determine where the net social benefit is greater than the net social cost. Therefore this must be understood before the process of undertaking a CBA occurs (Tisdell 1985).

3.4 Cost–benefit analysis (CBA) rationale

Although there are many techniques for appraising environmental policies and projects, CBA presents a robust methodology. The first rationale for selecting CBA as the method is that CBA forces the decision-maker to explore who are the winners and losers in pecuniary terms in regards to both spatial and temporal dimensions. It therefore allows rational decisions that are not 'lexical' or based only on a single goal or group of people. CBA achieves this by accounting for all gains and losses of utility or wellbeing (Pearce *et al.* 2006).

Secondly, CBA is clear in the requirement that any policy or project should be seen as one of a series of options. CBA also allows for the optimal scale of the project or policy, which would occur where the benefits are maximised. This allows the researcher to determine if the policy or programs should be undertaken at all. This is CBA's advantage over multi-criteria analysis (MCA) and cost-effectiveness

analysis (CEA), which only determine which alternative should be undertaken, and not if any alternative should be undertaken. This is possible as both the numerator and denominator are in the same units, unlike with MCA and CEA.

CBA allows for individuals' preferences to count. Some see this as a weakness rather than a strength as it implies that preferences should count, however badly informed these preferences are. It has also been argued that there are two categories of preference, those made out of an individual's self-interest (where the individual seeks to maximise private net benefit) and those made when the individual expresses a preference as a citizen (where the individual seeks to maximise social net benefit) (Pearce *et al.* 2006).

Finally, CBA accounts for time in a rigorous way. This is done through the process of discounting, which involves accounting for future benefits and or costs in today's dollar value. To do this, a discount rate which is expressed as a percentage, is discounted from the net benefit over each time step. The rate at which to do this is somewhat controversial. It is, however, impractical not to discount as it suggests that $1 AUD would have the same nominal value now as in 100 years' time (Pearce *et al.* 2006).

3.4.1 Discounting

Discounting allows future benefits to be expressed in today's dollar terms. There is significant debate regarding the correct discount rate because it has implications for the CBA result. Tisdell (1995) highlights the importance of ensuring the discount rate accounts for the time period in which the benefits are to accrue. The lower the rates of interest in the calculation, the greater the weight placed on future net benefits compared to current benefits. Consequently, there is more emphasis on the long-term sustainability of benefits (Tisdell 1995).

3.5 Undertaking a cost–benefit analysis

The overall CBA procedure to estimate and integrate public gains and private losses into net benefits involves undertaking five steps:

1. Identify the costs and benefits

2. Value each cost and benefit at each period of time
3. Convert this flow of gains and losses over time to their present day values by discounting them to give a present day value and then aggregating them together to give the net present value (NPV)
4. Undertake sensitivity analyses to assess the importance of uncertainties in the NPV
5. Prepare a final recommendation and draw policy conclusions.

(Rolfe 2002)

CBA is predominately used in partial equilibrium analysis, which assumes that one activity in one market has little or no effect on other markets. For market transactions, it is relatively easy to assess the different benefits and costs. The methodology selected to complete a CBA is generally marginal analysis or a partial budget approach.

To complete the CBA, firstly the demand and supply curves must be derived. For market goods, both the demand and supply functions and therefore curves are readily observable; however, with environmental issues both tend to be hidden.

This can be demonstrated in Figure 3.9 where price is on the horizontal axis, quantity is on the vertical axis, BS is the supply curve and CD is the demand curve. Net social benefit is calculated by subtracting WTP from opportunity cost: CBA calculates WTP as consumer surplus and opportunity cost as producer surplus. Consumer surplus is the total benefit from the consumption of a product (Figure 3.8 area 0CEQ), less the total cost of purchasing it (Figure 3.9 area 0PEQ); therefore, consumer surplus is area PCE (Pindyck and Rubinfeld 2005).

CBA in environmental analysis usually involves two different goods: one being valued by consumers, the other by producers. Producer surplus is defined as the differences between the producer's revenue minus the resource cost (Pindyck and Rubinfeld 2005). Opportunity cost is also represented by 0BEQ. Maximum net social benefit is achieved at the equilibrium E.

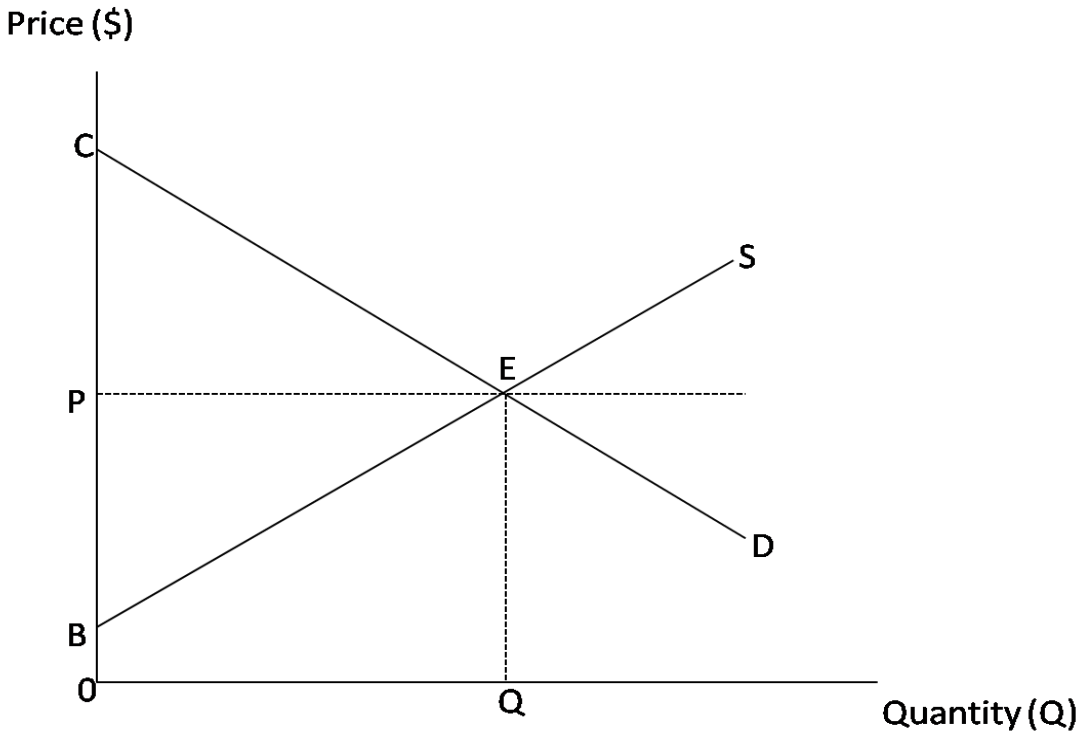

Figure 3.9 net social benefit

(Sinden and Thampapillai 1995)

Sinden (2004) explored the public benefits and private costs of vegetation protection in north-western New South Wales. The policy resulted in areas of native vegetation being protected, and the opportunity for clearing was possible if permits had been approved. The private loss or costs of the natural resource policy were measured through the loss of land value. The foregone cost of land clearing and the potential future benefit of decreased risk of land degradation, such as through salinity, were also accounted for. The public benefit was estimated by benefits from biodiversity protection, prevention of downstream river degradation, greenhouse gases and social factors, such as maintaining the viability of small towns. A non-market valuation methodology was implemented to determine the WTP of the community for the particular public benefits gained.

Similarly, Rolfe (2002) performed a CBA on vegetation clearing in central Queensland. The research explored the complexity of the issues and vested interests of different stakeholders. Marginal analysis was used to estimate the costs and benefits, with the costs estimated through changes in production and subsequent

income and the benefits estimated using choice modelling, a non-market valuation technique.

CBA provides the economic framework for debates about environmental management, with both the indirect and direct benefits and costs determined (Sinden 2004). In using the findings of a CBA, a decision-maker accepts measures of individual preferences expressed in WTP sums as valid measures of the welfare consequences of a given change in the provision of a public good.

3.6 Market failure

Inefficiencies and continued degradation of natural resources occur when the outcomes of an individual acting in their own interests are different from what would occur if individuals acted collectively to maximise the benefits and welfare to all members of society (Furubotn and Pejovich 1972). The decline in resource base occurs through aggregate short-term production or use levels that are too high and where long-term investment in the asset base is too low (Libecap 2009). Pearce and Turner (1990) explain that these different interests result in market failure, which occurs when property rights are inadequately specified or there are inadequate institutional arrangements. Natural resource problems can be caused by both market failure and government failure. The reasons for not having a market of environmental goods and services can include:

- large transaction costs
- externalities (spill-over effects)
- non-rival and non-exclusive impacts (public good effects)
- high uncertainty about the attributes of a good or service
- asymmetric information (sellers are much better informed than buyers, or vice versa)
- few buyers and sellers
- ownership cannot be defined and enforced, or it is very costly to do so.

(Murtough *et al.* 2002)

For example, asymmetric information has been the cause of many NRM issues, such as the introduction of invasive species, or salinity as a result of irrigation. Market failure in this instance may have occurred because the landholder was misinformed about the impact on their welfare of the alternative resource management strategies available to them (Godden 1997).

NRM issues have often been described as uncertain and 'wicked', due to the problems being not only complex, but also difficult to define and manage. Efforts to solve them create new, unexpected and possibly worse situations (Allan and Wilson 2009). Williams (1996) identifies the characteristics that a 'wicked' NRM issue exhibits:

- interconnectedness: strong connections link issues together
- a number of important dimensions: biological, geological, social, economic, and legal
- uncertainty: uncertain environments create risk where the outcomes may not be known and the risk is incalculable
- competing claims and interests
- social, organisational, and political constraints and capabilities as well as technological constraints, which are central to feasibility and desirability of solutions.

3.7 Issues of market failure

The existence of market failure is often used as justification for government intervention in a free market. NRM has three key types of market failure where government intervention is required; these include presence of externalities, existence of public goods and lack of adequate information.

3.7.1 Presence of externalities

Externalities refer to spill-over effects or impacts on any party not directly involved in a given economic transaction. This third party in the transaction then incurs these resulting costs or benefits. In 1920, Pigou argued that market failure resulted from the divergence between private and social costs that impact on third parties. There

are two situations which arise in this category: (i) the case where services are rendered without a due payment (i.e. a positive externality); or (ii) where disservices are rendered without compensation (i.e. a negative externality) (Coase 1960).

There are several types of externalities. Private externalities only affect an individual or firm, while public externalities affect a group of individual firms. There are also technological and pecuniary externalities. Pecuniary externalities occur when an impact is transmitted through the price mechanism, such as the advent of higher property prices resulting from more people moving into a region. Technological externalities occur when an impact is external to an individual or firm and is not transmitted through the price mechanism. These externalities affect production or others' utility directly. For example, a positive technological advance as a result of research and development of another firm is a positive externality (Tisdell 1982).

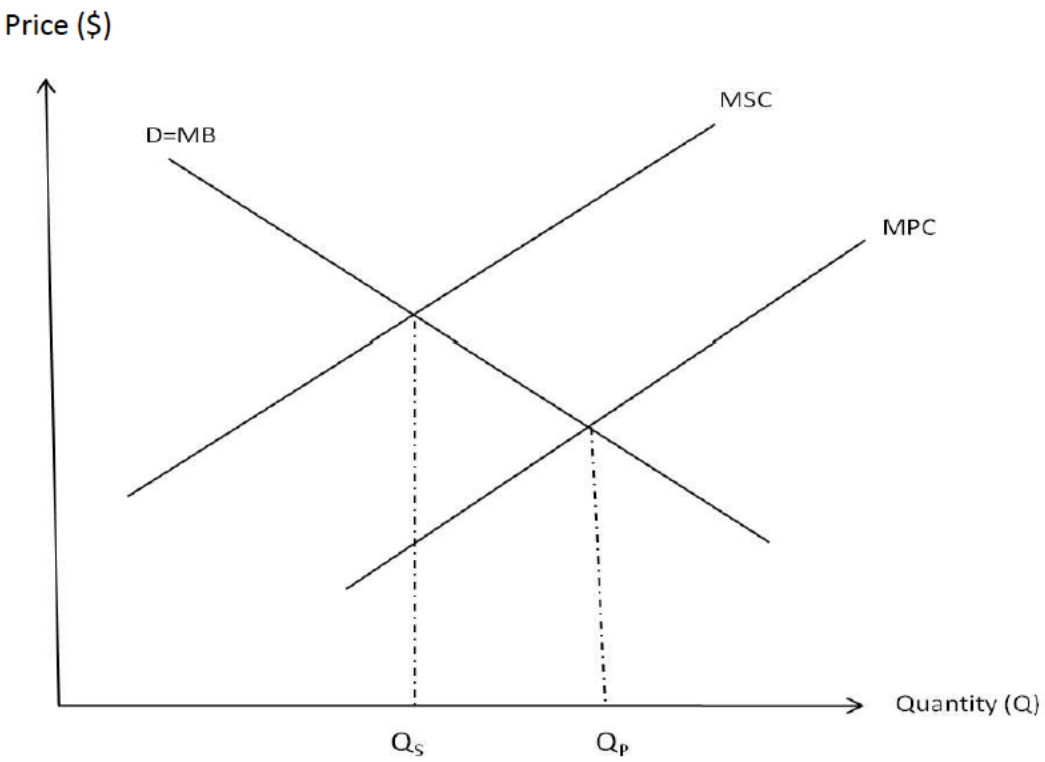

Figure 3.10 Negative externalities

(Tisdell 1982)

Many of the problems associated with NRM are negative externalities. One example is salinity, which sometimes has negative impacts on the water table for

the community. Figure 3.10 demonstrates how the presence of a negative externality affects the private and social costs of production. In this diagram, there is the demand curve which is equal to marginal benefit (D=MB), and there are two cost curves, the marginal private cost (MPC) and marginal social cost (MSC), which includes both private and public costs. As private firms only take into consideration their private costs in relation to what level of a good to produce, an equilibrium level would occur at Q_P. However, when taking into consideration the negative externalities resulting from this level of production, the more appropriate equilibrium level would occur at Q_S.

Positive externalities also occur in NRM issues. Positive externalities confer a benefit on the initiating agent, and there are spill-over effects on others. For example, a beekeeper receives the benefit of honey produced; however, there is also a benefit received by farmers through the pollination of crops. The divergence between private and social benefits helps to explain why biodiversity is declining around the world.

Figure 3.11 shows the incentive gap between private and social level of costs in relation to undertaking improved NRM activities. Individual landholders undertake NRM activities corresponding to point A. However, the maintenance and conservation of environmental and natural resources required to provide the desired social benefit is at point B. There is no incentive for private landholders to undertake the additional costs associated with extra NRM work, which they do not benefit from directly (i.e. the difference between C and D). As a result, despite public demand for environmental goods and services, the market for producing such goods and services is not well developed to facilitate such an outcome (Tisdell 1982).

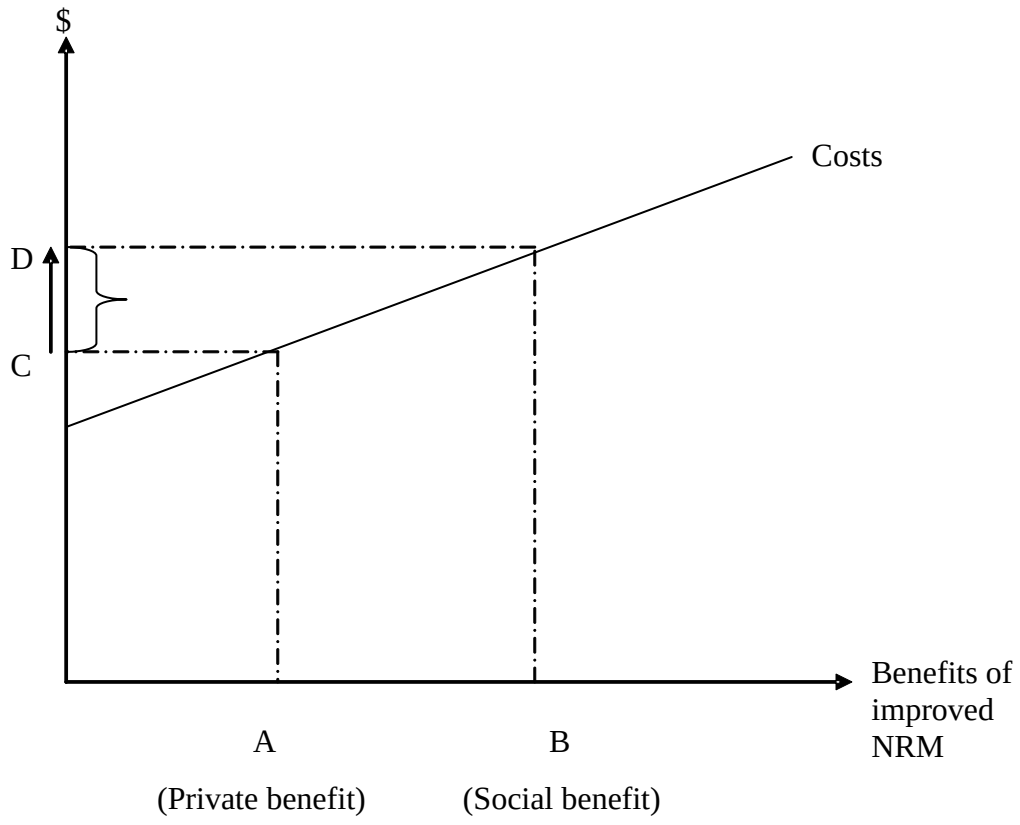

Figure 3.11 Gap between public and private benefits

(Tisdell 1982)

3.7.2 Existence of public goods

The attributes of public goods closely align with many NRM and environmental dilemmas. Pure public goods have attributes of non-excludability and non-rivalry in consumption. The former relates to the fact that no individual can be effectively prevented from consuming the good. However, the condition of non-excludability may change as the advent of new technology may give rise to the possibility of exclusion for certain types of goods. For example, city roads were non-excludable (and non-rival) before tolls came into operation. The introduction of tolls has meant that road companies are able to sell individual access to their particular road. Institutional and legal factors may also have an influence on the level of excludability of a good, hence excludability may not be inherent in the good itself (Chilsholm *et al.* 1974).

The second condition of non-rivalry in consumption refers to the notion that consumption of the good by an individual does not reduce the amount of the good available for consumption by others. This condition of non-rivalry in consumption is inherent to public goods such as biodiversity protection and clean air.

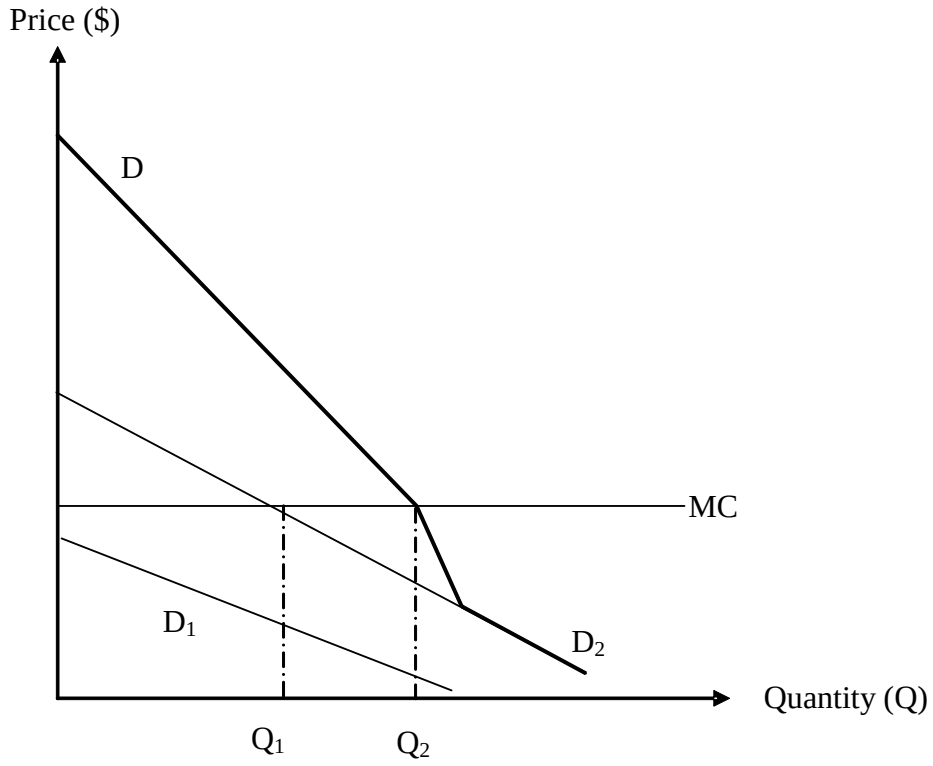

Figure 3.12 Optimal level of provision for a pure public good

(Tisdell 1982)

Figure 3.12 illustrates the optimal level of provision for a public good. Each individual has a separate demand for a public good. D_1 and D_2 represent individual demand, as there is non-rival consumption. D represents the sum of or total level of demand for the public good. The optimal provision occurs at Q_2 where total demand equals marginal cost (MC). The marginal cost of the provision of a non-rival good is greater than zero however the marginal cost of an additional unit of use is zero. D_2 represents the individual with the highest level of demand corresponding to the quantity Q_1, which is significantly less than Q_2 or the optimal

level of provision. There is no incentive for other individuals to provide the public good as they can take advantage of the provision at Q_1. However, it is noted that no amount of public good will exist if the individual demand curve exceeds the marginal cost.

One of the main reasons for the market failure outcomes concerning the provision of public goods relates to the inadequate incentives for individuals to voluntarily pay as they are able to enjoy the benefits from the good despite not contributing towards its provision (Pearce and Turner 1990). This is called the 'free rider' phenomenon. Pearce and Turner (1990) identified that this response stems from individuals who view their contribution to be too insignificant to make any difference to the overall outcome.

In reality, pure public goods are rarely encountered, with most public goods known as 'impure' public goods, sharing attributes of both public and private goods. Goods that exhibit the attribute of non-excludability but can be subject to congestion as use increases, are known as 'common pool resource' rather than a public good. For example, an unregulated fishery resource cannot exclude anyone, yet overuse tends to decrease the fish stock numbers (Pearce and Turner 1990).

Other goods known as 'mixed' goods provide both public and private benefits, and are often a common feature of environmental policy. For example, increased ground cover may result in both decreased soil run-off (a public benefit) and increased perennial pasture species for livestock grazing (a private benefit) (Ludwig *et al.* 2001).

3.7.3 Lack of adequate information

The efficient and effective operation of the market mechanism is premised on full access to information by all economic participants (Pearce and Turner 1990). This assumption is rarely satisfied in practice, as it is not always possible to obtain full knowledge before making an economic decision. This results in information failure, where decisions and actions may be misinformed. For example, lack of adequate information relating to the broader landscape impacts of specific land management actions may result in individuals undervaluing environmental goods and services.

When there is inadequate knowledge and understanding of environmental and biophysical processes, NRM practitioners face a number of dilemmas. This knowledge gap disguises the casual links between various land management actions and their impact on the environment. The consequence of time lags in management actions add to the complexity, as the impact of actions may not be observable for many years (Kirby and Blyth 1987). For example, land management actions targeting improved water quality may take years before a measurable impact is observed in the rivers and catchment areas.

Finally, another form of market failure occurs when there is asymmetric information. This occurs when one agent in the transaction is much more informed than the other. For example, land managers may understand and know the costs of land conservation better than governments (Fraser, 2013). However, governments may have a better understanding of the benefits to the community. This situation results in the task of NRM policy and programs responses to market failure being all the more challenging especially since the costs and benefits to the community are not easily discernible (Pearce and Turner 1990).

3.8 Policy and program responses to market failure

NRM policy is concerned with managing the agricultural–environmental components of land, water, marine and biodiversity impacts on the environment (Morrison *et al.* 2010). Morrison *et al*. (2010) explain it is a policy area characterised by politics (environmental versus agricultural), administrative complexity and scientific uncertainty. Complexity in NRM also comes from the interaction between facets of sustainability such as social, economic and biophysical, all of which have specialised objectives requiring specialised training (Ludwig *et al.* 2001).

To determine the appropriate policy mechanism, consideration of the costs and benefits of all interventions must be made, including the following factors:

- effectiveness of the policy achieving its desired outcomes
- efficiency regarding administration, monitoring and enforcement costs and the level of information required

- flexibility of the policy to deliver outcomes in the face of changing conditions
- acceptability of the policy to stakeholders
- equity concerns about the impact of the policy on different stakeholders.

(ABARE 2001)

It is important to consider that it is not just necessary to identify one policy instrument to respond to individual environmental problems. Depending on the nature of the problem, a policy response should consider a suite of responses that complement each other. Selection of these instruments has the resilience to ensure that the weakness of each is outweighed by the strengths of others. Similarly, it also needs to be acknowledged that multiple policy responses also increase transaction costs (Bennett 2005a; Randall 2007).

Precaution should also be included in any evaluation of policy interventions, to ensure that the policy includes methods to prevent the occurrence of serious or irreversible consequences in light of scientific uncertainty of the outcome. Equity is also often a priority of governments. Measuring equity is difficult because there are so many potential aspects and there is no agreement on the correct approach.

In Australia a variety of policy mechanisms have been implemented to address natural resource issues. These can be classified into six large groups:

1. direct provision: through the acquisition of land by government and non-government organisations to use as conservation reserves
2. regulation and legislation (Gunningham and Sinclair 2005)
3. improved property rights
4. voluntary approaches, supported by incentives and grants to effect behavioural change and enhance the likelihood of ongoing adoption of management practices
5. education and technical support programs
6. economic approaches, including property-based instruments such as leasehold agreements attached to the title, revolving funds for land purchase and resale, market-based instruments such as auctions, and off-set schemes with tradeable rights (Bennett 2003; Earl *et al*. 2010).

3.8.1 Direct provision

Governments may decide to address information failure and public good issues of market failure and directly intervene to provide the good or service (Rolfe *et al.* 2007). This may include the protection or conservation of a good such as biodiversity; however, typically the government aims to address this lack of information through providing more information on a range of issues.

Through a direct provision to address market failure, the government has to be cautious not to 'crowd out' (Pearce and Turner 1990). Individuals may voluntarily contribute to a public good for reasons such as ethical considerations and social responsibility. If government takes over supply of the good, individuals previously contributing to its provision may no longer to do so as the government provision may decrease their 'intrinsic motivation' or utility to do so (Pearce *et al.* 2006).

3.8.2 Regulation and legislation

Regulation has been a traditional response to address market failure. Implementing regulation directly influences environmental performance by identifying limits on pollution levels, regulating products or by limiting activities to certain conditions. Through regulation the government intends to have producers internalise the cost of their externalities. For example, governments may introduce licensing to reduce information asymmetries that may occur between buyers and sellers (ABARE 2001).

There are, however, some pitfalls for regulation as the main tool for environmental policy. The result of uniform requirements for compliance is that all firms have to meet the same standards, although the costs of modifying behaviour or production process vary between individuals or firms. As well there is no incentive to encourage improvement of environmental performance beyond the prescribed minimum standard, and no incentive for innovative environmental performance improvements. Regulations outlining a method of compliance can be costly for firms as it removes the discretion they could exercise to adopt the least-cost approach. To be effective, regulations are required to be enforceable and linked to penalties for non-compliance, requiring resources for ongoing monitoring of individual agents' actions (Bellamy *et al.* 1999; Pannell *et al.* 2008).

Despite potential problems and inefficiencies of regulation, there is still a role for regulation in a policy mix for achieving outcomes. This is particularly the case in situations where there is a high degree of risk for irreversible environmental damage and in situations where there is higher scientific uncertainty. Regulation is also an important component to support other policy mechanisms and instruments. Regulation also provides support to the other forms of policy mechanisms, including the smooth function of markets for environmental services and the enforcement of property rights (Pearce and Moran 1994).

3.8.3 Improving property rights

Property rights refer to the exclusive authority to determine how a resource can be accessed and used. Well-defined property rights supported by regulatory and institutional frameworks can assist with addressing problems associated with market failure and public goods. The issue of externalities involves more than one party, and it is often not clear who has to modify their behaviour. As long as clearly identified property rights exist, and there are benefits to be gained from changes, bargaining and negotiation will result in an improved outcome. However, the existence of transaction costs can have an impact on negotiation, especially where there are a number of stakeholders involved. Governments can intervene to modify property rights by:

- establishing property rights where previously none existed
- modify existing property rights with the introduction of restrictions governing use and access over the resource
- facilitation bargaining and negotiations between property rights holders and beneficiaries.

It is important to consider equity and fairness in the design of policy to modify existing property rights.

3.8.4 Suasive instruments

Suasive instruments are aimed at changing an individual or firm's perception and priorities regarding the environment through information provision, education programs and social recognition and pressure schemes. Suasive instruments attempt

to internalise environmental awareness and responsibility onto individual decision-making (ABARE 2001). In comparison to other policy options, suasive instruments present a comparatively low cost option if the private benefits are available and well understood but are likely to be ineffective if used in isolation from other policy options. For example, suasion measures work well with market-based instruments to try to achieve NRM outcomes through the provision of information and implementation of education campaigns to raise awareness and improve participation rates.

3.8.5 Market-based instruments

Market-based instruments aim to influence the behaviour of individuals to ensure that changes occur at least cost, and hence the resource is used more efficiently (Windle and Rolfe 2008; Rolfe and Windle 2011a). To ensure this occurs, market-based instruments affect the relative prices of alternative behaviour available to firms and individuals. Market-based instruments allow the opportunity of market signals to incorporate all or some of the costs that the externality imposes on society through (in the case of NRM) environmental degradation (ABARE 2001).

Market-based instruments include tradeable permit schemes, pollution off-set arrangements and conservation auctions (Rolfe and Windle 2011b). The use of market-based instruments has the advantage that resource users can determine their best response to new price signals or quantity constraints. By achieving this, the environmental outcomes are achieved in the most efficient manner (ABARE 2001).

3.9 Selecting policy mechanisms

Depending on the complexity of the NRM issue, an astute policy response will include a suite of policy responses, where many instruments can complement each other. This approach is therefore more likely to achieve outcomes and provide more flexibility in responding to changed conditions. Some instruments are more likely to function more efficiently when used in conjunction with others. However, in practice, the selection of mechanisms is often not selected through a rigorous process, so the policy response and tends to rely on a small number of policy mechanisms. This highlights the importance of understanding the public and private benefits in selecting an appropriate suite of mechanisms (Pannell 2009).

Key considerations include the understanding of the public and private costs and benefits to evaluate which policy mechanism is most effective at achieving environmental outcomes (Pannell 2009). Similarly, information regarding the importance of the environmental asset to the community, the project risks, adoption of new practices, time lags and costs are all required to fully understand how to prioritise investments and the subsequent policy recommendations.

'Public net benefits' are defined as benefits minus costs accruing to society other than to the person whose land management is to be altered. 'Private net benefits' on the other hand are defined as the benefits minus the costs accruing to the private land manager as a result of the proposed changes in land management excluding transaction costs that are a part of the policy intervention (Pannell 2009b). Landholders will adopt land management practices with positive private net benefits, provided that they are able to learn about those practices. Positive incentives refer to land use change being encouraged through the use of regulation or financial instruments. Negative incentives are regulatory or financial incentives that are used to inhibit change.

Pannell (2009b) articulates the following rules for selecting policy mechanisms:

1. Do not use positive incentives for land-use change unless the net benefits of change are positive

2. Do not use positive incentives if landholders would adopt land-use changes without those incentives

3. Do not use positive incentives if private net costs outweigh public net benefits.

4. If private net benefits outweigh public net costs, the land-use changes should be accepted if they occur, implying no action. Alternatively, if it is not known whether private net benefits are sufficient to outweigh public net costs, a relatively flexible negative incentive instrument may be used to communicate the public net costs to landholders (e.g. pollution tax) leaving the decision to landholders. Inflexible negative incentives, such as regulations, should not be used.

5. If public net costs outweigh private net benefits from a set of land-use changes, use negative incentives to discourage uptake of land used in this case.

6. If public net benefits and private net benefits from a set of land-use change are both negative and landholders accurately perceive this, then no action is required. Adverse practices are unlikely to be adopted. If there is concern that landholders have misperceptions about relevant land uses, adoption of environmentally adverse practices could be discouraged by extension, or more strongly by negative incentives. Pannell (2009b) illustrates this concept further in Figure 3.13.

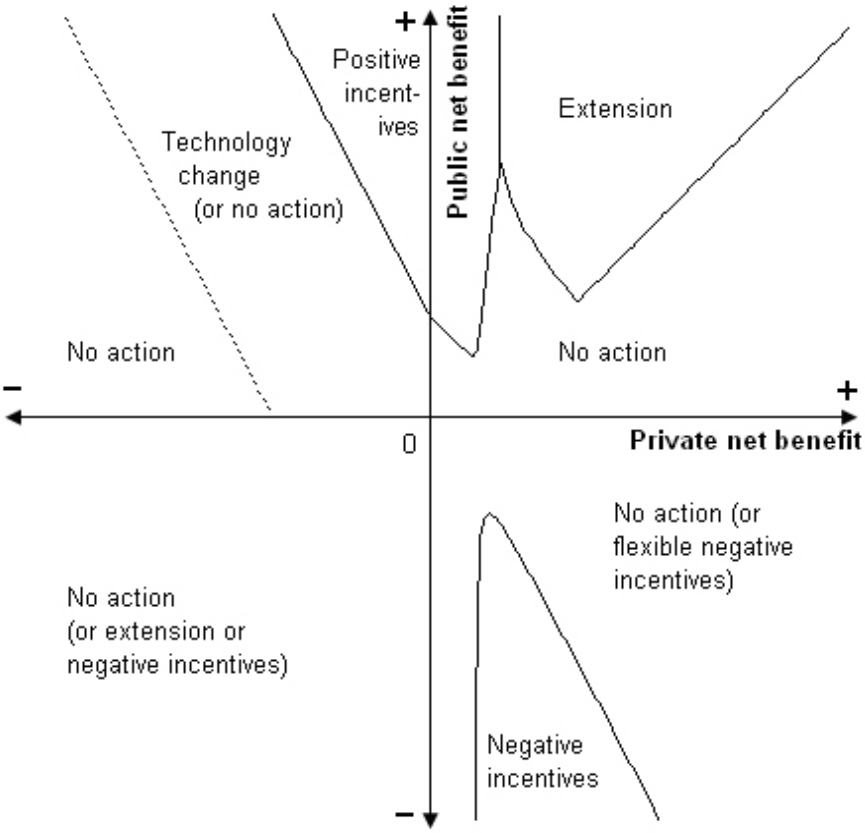

Figure 3.12 Selecting policy mechanisms

3.10 Summary

This chapter highlights the complexity of natural resource management, reasons for market failures and the role of cost benefit analysis. The difficulties in applying

effective policies for improved outcomes underpins the understanding of the complex interaction between utility, welfare and natural resources. The following two chapters will explore the bioeconomic methodology and results to further understand the cost of sediment reductions.

CHAPTER FOUR: ESTIMATING THE COSTS AND DETERMINING THE SUPPLY CURVE OF CHANGED MANAGEMENT PRACTICES

Understanding cost-effective policies and programs in rangelands productions systems requires an understanding of how the various biophysical attributes and production attributes interact. This chapter describes different approaches that have been taken to estimate rangelands productions systems and then describes bioeconomic modelling, its useful applications and the components required to be integrated. Finally, the chapter highlights the use of pasture simulation models in rangelands, more specifically, in northern Australian rangelands.

To understand the grazing production system a production economics approach is required. There are several methodologies exploring inputs and outputs of the production system; these include data envelopment analysis; stochastic frontier analysis; and econometric cost, profit and production functions (Coelli *et al.* 2005).

Data Envelopment Analysis (DEA) is a measure of efficiency in producing outputs with a given number of inputs. It is a type of frontier analysis of inputs and outputs and classifies decision-making units according to whether they are, relative to each other, (using linear programming methods) on the efficiency frontier or not. DEA assigns an efficiency score between zero and one for each decision-making unit. When a decision-making unit does not provide evidence of inefficiency in the use of inputs to generate outputs compared to the other decision-making units, a score of 1 is assigned. There are two basic forms of DEA with each one the dual of the other. They provide similar information, with the solution of one being the shadow price of the other (Coelli *et al.* 2005).

DEA requires large amounts of observations as well as variables. For example, Hof *et al*. (2004) used 12 indicator variables to identify areas in the USA where there is

a maximum potential for improvement in forest and rangeland condition. A number of biophysical variables were included, such as per cent mortality in growing stock on timber lands and total nitrogen measured in surface waters. A number of human activity variables were also included, such as beef cow density (stocking rate) and total farm production expenses. This amount of data variables and points limits the use of the DEA methodology in Australian rangelands.

There have been a number of optimisation studies using methods such as stochastic dynamic programming, or dynamic programming. The approach implicitly incorporates user costs in determining an intertemporal optimum resource allocation. The approach is appropriate for sustainability issues, as it explores the environment at a particular time (Cacho 1997).

Productivity is defined as a measure of the amount of output produced by a given bundle of inputs. There is a series of components to total factor productivity, including scale efficiency, technical efficiency and productive efficiency. Productivity growth through technical and scale efficiency has been explored in the northern beef industry; the research found that total factor productivity growth was increasing at a decreasing rate (Gregg and Rolfe 2011).

These production economics approaches, although robust when applied appropriately, do not always involve the inputs of biophysical inputs such as pasture growth, which depends on rainfall, soil type and species composition and hence do not allow relationships between economic and environmental systems to be modelled. To account for such processes and interactions, bioeconomic modelling is used. This comprises a biophysical model (to account for biophysical processes) integrated with an economic model (to explore the production function). The bioeconomic model allows an estimation of the supply side of the CBA to be estimated.

This chapter provides a review of the application of bioeconomic modelling to rangelands and grazing, giving insight into the strengths and challenges with applying the methodology. It also provides a basis for understanding the components of a bioeconomic model and raise awareness of issues that require

attention. This chapter aims to provide a foundation to develop and design the bioeconomic model for the case study used in this research.

4.1 Bioeconomic modelling

To estimate the costs of changing production systems for environmental benefits, a bioeconomic model integrates biophysical and economic components. This economic link makes available the connection between the production system and its environmental impact (Cacho 1997). Bioeconomic modelling was devised to explain and predict cause and effect relationships in ecosystems and then determine associated economic effects (Bennett 2005a). The challenge of linking the cost of changing production systems with environmental changes which are encompassed in different biological processes and ecological systems is complex. The use of bioeconomic modelling in environmental economics has been growing in Australia, with an increase in public interest and political debate (Bennett 2005a).

The biological system can be represented in different types of economic models. Bennett (2005a, p. 248) describes the contribution of bioeconomic modelling as "varied from straightforward considerations of the costs of alternative resource use strategies to complex integrations of biophysical models of ecological farming systems with social cost–benefit analyses and policy advice".

Bioeconomic modelling is increasingly being implemented for use in environmental economics. It is flexible in that it can be applied in a wide variety of contexts. Researchers who have developed bioeconomic models have often used them to draw policy conclusions or make statements regarding the incentives that stakeholders face (Bennett 2005a).

The economic model provides the link between market information and the production system, and can be designed to interact with the biological model at a given level of aggregation, depending on the problem being analysed. It forms the overarching model, which controls the biophysical model through the model's inputs, and feedback is acquired as outputs.

Bioeconomic modelling allows complex interactions between agro-ecological and socio-economic issues to be transparent in the policy debate (Kokic *et al.* 2007). Bioeconomic models contribute to the policy debate by identifying the net social costs of alternative courses of action. Such models can then be incorporated into a CBA. The ability of bioeconomic models to effectively compare different management scenarios with the business-as-usual scenario allows a NPV to be estimated. A comparison of different management systems ensures that investments can be targeted and policy options considered further. Bioeconomic modelling specifically has the ability to provide information regarding the scale, nature and distribution of the potential cost and benefits of a particular management practices (Whitten and Bennett 2005a).

Brown (2000) describes a number of purposes for agricultural systems models. Kruseman (2000) classified models by the specific characteristics of the system being modelled. The classifications developed included descriptive models, which are qualitative and explain the interactions within a system using a particular set of terms and definitions; explanatory models, which provide interpretative insights into past relationships between measurable indicators or variables; predictive models, which are used to forecast future activities, often making use of data from past performance; and prescriptive models, which are intended to offer guidance on management of a system in light of some normatively specified goals.

Descriptive models demonstrate what biophysical systems or decision-makers would actually do in a current situation and inform the potential impacts or consequences of future projects. For example, a bioeconomic model has also been applied to rangelands in NSW to explore a robust grazing strategy while being conscious of rainfall variability for improved profit (Janssen *et al.* 2004). The model explores the use of a lower sheep stocking rate to limit the amount of 'trading strategies' in particularly dry or wet years. The model used a 100-year time horizon with two scenarios. Profitability was dependant on wool production, mustering costs, adjustment of stock costs, and loss of revenue due to fire. The first scenario modelled the manager having perfect knowledge of rainfall events and perfect foresight, and the second scenario focused on the manager ignoring rainfall variability. The research explored the certainty of variability in rainfall and the

manager's ability to maintain productive capacity. The research found that rainfall and stocking rates were key factors to maximising profit (Janssen *et al.* 2004).

Predictive models describe a decision that should be made if certain objectives are achieved either at a society level or at an individual level. They provide interpretive insights into the future relationships between measurable indicators. They optimise a measure of welfare or utility or assess consequences of an optimum being achieved. These types of models are often used to optimise social welfare for a catchment or community, and they provide thought or decision on future directions to achieve an improved level of wellbeing, such as shown in (Bhattarai *et al.* 2008), where the objective was improved water quality outcomes at a community level.

Models focused on econometric solutions to production issues such as DEA or optimisation, biophysical models and bioeconomic models also can inform policy by providing meaningful insights into various systems and trade-offs (Kokic *et al.* 2007).

Bioeconomic models can be implemented for a number of purposes in research and management of production systems. Possible uses are:

- integration of existing data and concepts
- identification of gaps in the research
- screening of potential experiments
- generation and testing of hypotheses
- deduction of unmeasurable parameters
- interpretation and evaluation of experimental results
- design of efficient production systems
- determination of best operating conditions for a given production system
- evaluation of policy.

(Doole *et al*. 2013)

The process of developing a bioeconomic model is not a linear one, and generally involves:

1. defining the objectives of the goals to be achieved and the system boundaries
2. reviewing the existing knowledge and developing a conceptual model to determine the key relationships and variables; this also can allow a simple mathematical process to be developed
3. formulating the model equations and interactions between variables
4. estimating parameters; often, computer statistical packages will allow a best fit of non-linear differential equations into a dataset
5. model implementation
6. sensitivity testing of the key parameters
7. model validation.

(Doole *et al*. 2013)

4.3 Application of Bioeconomic models

As bioeconomic models are able to be applied to a wide range of production systems and explore varying policy contexts, there is a variety of different classifications of models (Cacho 1997). Models can be classified by the objective they solve or by the system they describe, including:

- Simplistic or holistic: Simplistic models have limiting assumptions and are designed to be solved analytically. Conversely, holistic models attempt to describe a system in detail and often cannot be solved analytically. Simplistic models are 'open' scientific investigations with the assumptions, deductions and conclusions available to critical analysis. In contrast, a holistic model requires IT capability, and equations are solved numerically.

- Empirical or mechanistic: Empirical models aim to predict, based on empirical data collected over a period of time, and mechanistic attempts to describe a process, which may include a series of assumptions. Empirical models are often a 'black box', where the objective is to obtain an output

based on a series of inputs. Mechanistic models, on the other hand, represent a mathematical description of the actual process being modelled.

- Static or dynamic: Static models do not contain time as a variable and generally simulate equilibrium conditions. When the system is disturbed, a static model will predict what the resulting outcome is but will not provide any information on the path to getting there. In contrast, a dynamic model includes time explicitly, and differential equations are used to explain the change in the system variables through time.

- Deterministic or stochastic: Deterministic models can be used to predict expected values for the system and ignore the possibility of variability. Stochastic models contain random elements or probability distributions reflecting the uncertainty or riskiness of the real world. Frequently, models are built that are initially deterministic until confidence in the model is achieved, and then a random element is added, changing it to a stochastic model.

- Positive or normative: Positive models (also called descriptive models) indicate what outcome will be achieved from alternative decisions; normative models (also called optimising models) provide information to describe the course of action that will lead to an optimised outcome with particular criteria (Cacho 1997).

Bioeconomic models may contain elements from a number of these classifications, depending on the overall purpose of the model.

4.4 Important features and considerations of bioeconomic modelling

This section aims to explore the issues and features of bioeconomic modelling to provide insights and ensure a robust model is developed.

In deciding to use a bioeconomic model to solve a problem, the problem must first be defined. This allows an assessment of the goals to be achieved, such as profit

maximisation subject to environmental regulations. Then an assessment is needed of the system boundaries, such as paddock level, whole farm or catchment level. The modular structure of the modelling framework easily enables the incorporation of information from various disciplines. Relevant biophysical and economic process can thus be analysed separately before their full interaction is taken into account. This allows a degree of transparency, and data requirements can be better controlled (Kruseman and Bade 1998). Once the problem has been defined, there are five aspects required to be integrated into a bioeconomic model; depending on the complexity of the issue, each aspect requires varying degrees of attention (Brown 2000).

1. dynamic and recursive process modelling
2. temporal and spatial scale
3. unit of analysis and decision-making level
4. integration and linkages
5. uncertainty and risk management.

These components are explored as the chapter progresses, and an applied case study is reviewed to demonstrate the application of these components.

4.4.1 Dynamic and recursive process modelling

To develop a dynamic or recursive model the interaction between the economic and the agro-ecological sub-models are required to be dynamic, to ensure that they are responsive to changes in the environment. These responses and the subsequent decision rule from the agricultural side of the model is then required to feed back into the biological process and environmental impact on a given time scale. The ability to have sequential decisions and outcomes that link into the next period's decisions and outcomes is an important aspect (Brown 2000). Models that predict the outputs of biological processes based only on a set of empirical observations are limited in the range of input combinations that can be considered. Process models

that are developed in an attempt to describe biological processes are more flexible and adaptable than empirical or mechanistic models (King *et al.* 1993).

For example, MacLeod *et al*. (2010) applied bioeconomic modelling to determine the profitability of best grazing practices in the beef industry. The model was responsive to stocking rate, wet season spelling and prescribed fire over a 26-year timeframe based on 18 paddocks in the Duaringa area in central Queensland. These variables were modelled with the point-based pasture modelling system GRASP, using yearly time-step feedback mechanisms. This annual dynamic feedback allowed a range of input combinations to be considered.

Teague *et al.* (2009) developed a bioeconomic model to determine the ecological consequences of a grazing system with continuously stocked livestock on maintaining range condition, maximising profit or improving range condition over a 30-year time frame. Range condition was programmed to decline in response to excessive grazing pressure, below average precipitation and increase in woody plants. Ensuring that the model was responsive to environmental changes allowed feedback processes to be realised over the 30-year period.

4.4.2 Temporal and spatial scale

It is important to use the most relevant scales, both temporal and spatial, to model decision-making. Causes and effects must be able to be linked spatially to allow the full production impact on environmental process to be realised. Spatially explicit modelling accounts for the dynamic interaction between sub-areas of the area being modelled, whether linked to GIS data or not. Similarly, it is important to assess the situation at more than one point in time. For example, food security and NRM have a linked effect and are intertemporal. Their spatial dimension can only be understood as the time element is incorporated into the model (Brown 2000).

In Alabama, in the Wiregrass watershed (catchment) a bioeconomic model was implemented to assess the change in nitrogen and phosphorus run-off and sediment deposition in the main channel, which has altered as a result of land-use change (Bhattarai *et al.* 2008). The focus time scale for the study was 1992–2001, and the land-use change was predominately from forests to cropland, pasture and rangelands (Bhattarai *et al.* 2008).

A model simulation called SWAT was run, which required weather inputs (daily record of precipitation, wind and minimum and maximum temperatures) and management inputs (irrigation, tillage, chemical and fertiliser application). These inputs were then converted into standard SWAT input files within the model. The model then simulates the run-off levels of nutrient sediments and chemicals under a particular combination of land-use and land management scenarios. The output from SWAT is crop yields, stream flows and sedimentation and nutrient run-off levels, which are traced through the catchment for both short and long periods of time. The economic model used explored the change in total profit as the land use changes. The research did not, however, have detailed information on the type of crop or production system being used, and this was a limitation of the study (Bhattarai *et al.* 2008).

Bioeconomic modelling has been used in Californian rangelands to determine the most profitable enterprise operation (Standiford and Howitt 1992). The research was prompted by the decrease in hardwood rangelands through the clearing for improved rangelands and for firewood. The increased demand for recreational hunting influenced the stock of oak trees, with the game species in demand being quail, deer, turkey and feral pig, which have an enhanced habitat with an increase in trees (Standiford and Howitt 1992). A number of control and state variables, to account for the time steps were required for the different game species. To overcome these challenges, a non-linear optimisation approach was used (Standiford and Howitt 1992). This model is complex, accounting for three production models of firewood, grazing and game hunting and the different time effects. It does, however, highlight the complexity of problems that bioeconomic modelling can be used for. The model simulation were conducted over 15 years, which is relatively short compared to some similar grazing bioeconomic models. The stochastic elements that were accounted for include rainfall, prices and a chance constraint.

There were a number of key factors that impacted the economic outcomes of the model. The time periods of consecutive high rainfall, which allowed for opportunistically high production. This highlights the importance of accounting for temporal scale, particularly for stochastic variables. Rainfall is a key variable to account for over different temporal scales as it has a high impact on the agricultural

production system (Standiford and Howitt 1992). Heterogeneity in the landscape affects the model outcomes significantly, and therefore the spatial scale on which NRM problems are considered is critical. For effective use in a CBA, the spatial scale that is being explored must be identified and the degree of heterogeneity understood. This then potentially allows various variables and the impact they pose on both the biophysical and economic outcomes to be understood in a broader welfare sense (Kokic *et al.* 2007).

4.4.3 Unit of analysis and decision-making level

The level of analysis needs to capture who is being affected at the policy level or whose welfare is potentially altered. The decision-making level (individual or the catchment level) will determine the design and the parameters included in the model. This would allow for welfare issues to determine who are the winners or losers of the system being modelled (Brown 2000).

Zivin *et al*. (2000) analysed the feral pig as a multi-use resource and considered the potential for recreational hunting as a policy for population control in the Californian rangelands, affecting both individuals' and the communities' utility. To account for both individuals' and the communities' utility, the key design parameters which captured the relationships of the damage to agriculture, as a function of total animal population and animal population dynamics, were critical. The model then introduced the landholder optimisation problem. Three management practices were explored: hunting and trapping, just hunting and just trapping. The results were dependent on the steady-state size of the herd, with the cost of trapping only viable under a large herd size and the fee for hunting creating an incentive for the landholder to have a large steady-state herd.

Having the landholder as the decision-making unit resulted in the parameters and model being designed for optimisation for their decision making. It did not, however, consider the community-level impacts on welfare nor the NRM implications, which would increase with a larger population of feral pigs. This outcome reflects the result of increased feral pig populations in California after hunting was legalised in the 1960s as there is greater private benefit for the landholder to have people come to pay for a hunting experience than the cost to the

landholder from damage to agriculture. However, the community cost of loss of biodiversity from the increased populations was not considered (Zivin *et al*. 2000).

4.4.4 Integration and linkages, and uncertainty and risk management

The integration between economic and biophysical parameters is important to allow feedback among the processes and to achieve dynamic modelling. Uncertainty and risk are important aspects to cover, with bioeconomic modelling allowing potential risk analysis of factors neglected in production economics. Simulated time series reflects the effects of stochastic effects such as rainfall under constrained production parameters; however, the simulated output usually will not reflect all the stochastic influences affecting output (Musser and Tew 1984).

Cooper and Huffacker (1997) explored the interaction between stocking rates and an invasive weed species known as cheatgrass on the productive pasture species in the intermountain regions of the United States. To account for uncertainty and risk, a 100-year time frame was used to explore the dynamics of pasture competition, and a resilience threshold was inbuilt to explore the recovery of the rangelands after historic over-grazing (Cooper and Huffacker 1997).

This research highlights the effect of different time periods and starting points. The timeframe selected allowed uncertainty to be accounted for and the various impacts of selective seasons to be understood, avoiding skewed results. However, having variables in the biophysical model that cannot be manipulated or teased out further such as pasture recovery thresholds provides some limitations, as explained by Musser and Tew (1984).

Musser & Tew (1984) also identify some potential problems with biophysical simulation in that many decisions and uncontrollable inputs are predetermined. While reason to support the approach of predetermined outcomes may be provided, it may result in agricultural economists being frustrated and disadvantaged. In the review of bioeconomic models for beef production systems, Chudleigh and Cezar (1982) recommended that future models avoid the ‘black box’ approach and enable components to be transferred much more readily to other models if required. They also recommended that there are clear guidelines on model construction to enable use from a wider audience.

Bennett (2005b) describes bioeconomic modelling as the least controversial method used in environmental economics. However, he does note that failure to consider the benefits side of the equation is sometimes an inadequacy. Bioeconomic modelling focuses on assessing real opportunity costs of making management changes, which can be challenging when there is a lack of science linking alternative management actions with environmental attributes (Bennett 2005b).

4.5 Applications of bioeconomic modelling to rangelands

Bioeconomic modelling is increasingly being used in NRM policy contexts to gain a better understanding of the economic consequences of changed resource conditions.

Kruseman and Blade (1998) highlight the importance of bioeconomic models and suggest:

> The complexity of household behaviour in combination with changes in the state of natural resources (e.g. soil degradation), make it necessary to use bio-economic models to evaluate if changes in technology and policy induce farm households towards more sustainable land use while maintaining or improving their welfare. The aggregate effects of such farm household decisions on environmental and welfare indicators are a benchmark for evaluating different policy options. (Kruseman and Bade 1998, p. 466)

Kokic *et al*. (2007) implemented bioeconomic modelling to gain further understanding of the impact of climate change on farm income. Using a farm income model and probabilistic forecasts of crop and pasture growth for the coming season, the model represented the changes in cropping incomes with changes in climate.

Bioeconomic modelling has been implemented to predict natural resource outcomes and costs in forestry production systems. An example of this is the work completed by Cacho *et al.* (2001) that explored the use of forestry as a means to control dryland salinity in the Liverpool plains in NSW. Cacho *et al.* (2001) implemented a model to assess forestry both as a means of income and to prevent the negative impacts on crop production from the water table from rising further. The model examined the production system in relation to the growth of the trees and crops and

the other options available to reduce salinity and ensure that future crop production would continue, allowing the cost of reducing salinity to be estimated. The research found that although forestry did not represent a viable means of income alone, it did ensure that the water table would not increase further and crop production could be continued (Cacho *et al.* 2001). The model allowed the additional benefits of forestry establishment to be evaluated.

In the past, bioeconomic modelling has been used to examine the effect of land degradation and stocking rates in rangelands, and to examine the impacts of higher wool prices, increased discount rates and lower property size on land degradation (Epps and Crittenden 1992). The results demonstrate that producers would risk the degradation that occurs with higher stocking rates in response to these variables (Epps and Crittenden 1992).

Passmore and Brown (1991) highlight the importance of stochastic variables such as rainfall in rangeland production systems and optimal stocking rates for sheep. Using the variables of stocking rates, property size, wool prices, discount rates and risk aversion, the trade-off between degradation and profit was investigated. The research highlighted the lack of technical data as a key constraint of the models (Passmore and Brown 1991).

Conservation bioeconomic models in the semi-arid regions of Africa have previously also been used, and the optimisation function can be applied to various uses of the rangeland depending on the point in time and relative to prices. In one study, the model explored the resilience of the rangeland and incorporated variables such as wildlife growth rate, grass growth rate and woody biomass to optimise the use of the ecosystem through revenues from harvest and costs of maintaining the system (Perrings and Walker 2004). The research suggests that ecological–economic systems are hierarchical, with separate subsystems each operating at different spatial and temporal scales and both interacting with each other. They argue that biological variables are characterised by flexibility or rigidity and by resilience or adaptive capacity (Perrings and Walker 2004).

The development of bioeconomic models in the United States that combine biophysical and economic data to explore the trade-offs between industry and NRM policy has been rapidly expanding, particularly for rangeland issues. Research

subjects have included the impact of over-grazing on species composition and the increase of less productive pasture species when over-grazing occurs (Finnoff *et al.* 2008). Cooper and Huffaker (1997) also examined the resilience of pasture species in rangelands, creating three separate sub-models. The first sub-model was implemented to explore the dynamics of an optimisation-based grazing decision model. Secondly, a sub-model capturing the predator–prey relationship between the livestock and vegetation was developed. Thirdly, a long-term (decade) dynamics of species-competition sub-model was developed. This sub-model investigated the grazing-induced succession from perennial grasses to less environmentally desirable annual species. The model allowed an estimation of a 'successional threshold', which measures the resilience of the rangeland ecosystem in recovering from historic over-grazing. The conclusions highlighted the environmental efficiency of programs that promote the recovery of private rangelands by offering financial incentives to reduce grazing pressure (Cooper and Huffacker 1997).

The evaluation of natural resource policies and mechanisms has been explored by Haffaker *et al*. (1990), who determined the trade-offs between different policy options for controlling wild horse populations and the impact on the western US livestock industry. A model was created to reflect the institutional framework the range landholder was to operate in. The results concluded that the policy option of legislation for control is possibly economically inefficient (Huffaker *et al.* 1990).

4.5.1 Key considerations in rangelands bioeconomic modelling

With few Australian bioeconomic models (Passmore and Brown 1991; Epps and Crittenden 1992; MacLeod *et al.* 2010) but a large body of specific rangelands biophysical literature (Ash *et al.* 1995; Scanlan *et al.* 1996; Orr *et al*. 2006; O'Reagain *et al.* 2009; Silburn 2011), there is opportunity to create a bioeconomic model regarding the economics of rangelands grazing and its environmental and economic trade-offs.

This model could test the impact of stochastic variables, such as rainfall, to allow further understanding of management practices. Temporal and spatial scales are also methodological components for further consideration and testing. Temporal scale may provide justification to explore further if there is a starting point bias, and

if the choice of time period affects results (Musser and Tew 1984). Spatial consideration of the heterogeneity of the landscape and the impact of the selected parameters for the property scale and location in the catchment could also be further explored.

4.5.2 Rangelands simulation models

Simulation models provide insights into rangelands management and changes to the system. However, rangelands are complex ecosystems that experience considerable rainfall variability; it is therefore challenging to use empirical studies to forecast the impact of factors such as climate, fire, grazing, invasive species and management.

Many trials and research studies are constrained to specific locations and short time frames (5–10 years or less) (CSIRO *et al.* 2004). To capture the complexity of rangelands, models are often developed to understand research questions. Modelling pasture and animal production and ensuring the integrity of the model is critical. Model developers must ensure that the mathematical expressions are a true and correct representation of the biophysical processes. It is also critical that the underlying input data is accurate under the model design specifications (Rickert *et al.* 2000). This section provides an elementary explanation of biophysical modelling and a review of grazing models and the key variables implemented; it also provides further understanding of the importance of obtaining the correct biophysical rangeland simulation model.

Simulation models can be developed through the use of empirical data, accumulated through research trials, and mechanistic models, which use equations that reflect a theoretical understanding of factors that control the relevant process: for example, the factor ‘intake’ as a function of rumen size, rate of digestion and energy metabolism (Rickert *et al.* 2000). These components form research models that allow analysis of complex interactions. Interrelationships between subsystems are often found in pasture and animal production models (Figure 4.1). Each subsystem consists of processes that control the state variables in the subsystems. The key processes are the interfaces between the subsystems, which are influenced by factors in two or more adjoining subsystems (Rickert *et al.* 2000). This is illustrated in Figure 4.1.

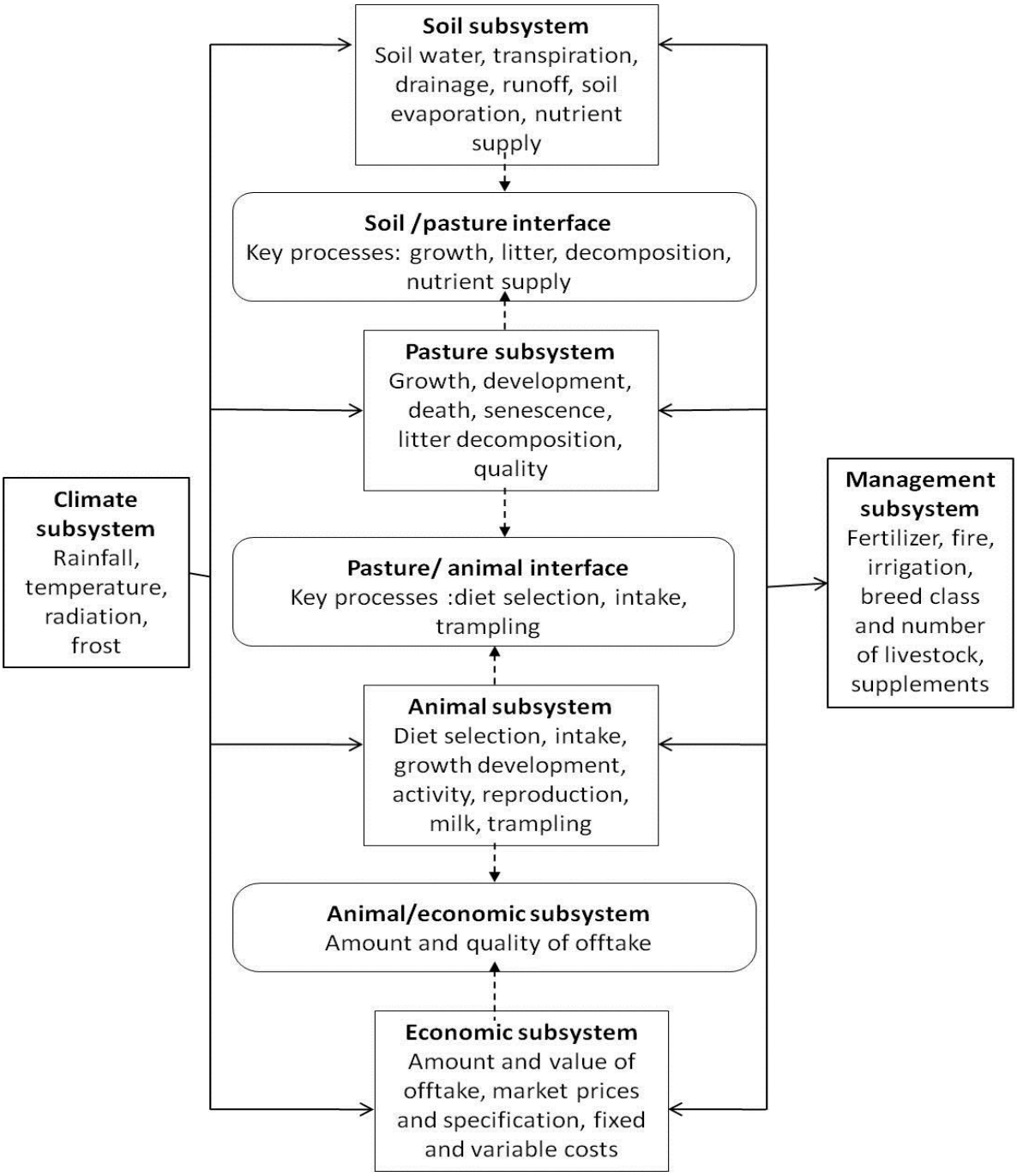

Figure 4.1 Pasture and animal production simulation models

(Rickert *et al.* 2000)

Simulation models provide an opportunity for land managers to explore a range of scenarios regarding climatic, edaphic, biotic, economic and management factors. The simulation model is captured in computer code, with the model validated against previous trials or research projects. Some simulation models have a specific

focus limiting their use, or interpretation of the output may require in-depth knowledge of the model (CSIRO *et al.* May 2004).

For example, the CENTURY model was built in the US to explore the interactions between soil carbon and nitrogen dynamics in grasslands and cropping systems. It incorporates land management practices such as irrigation, fertilisation, harvesting, fire and grazing. Land-use change or management practice change can be simulated by changing the plant community type during model runs, that is, starting with woodlands, clearing to pasture and then running a grazing system (Kirschbaum *et al.* 2001). Some of the major input variables include:

- monthly average maximum and minimum air temperature
- monthly precipitation
- lignin content of plant material
- plant maximum and minimum nitrogen, phosphorus and sulphur content
- soil texture.

CENTURY has widespread application on an international level, with comprehensive carbon and nitrogen models and common land management practices. The model's plant production components are probably not as strong as its specialist forest production, crop and pasture growth components. It also does not have the capabilities to model animal production interactions, as it was not designed for this purpose (Kirschbaum *et al.* 2001).

With grazing management and long-term sustainability highly correlated, Carrying Capacity Estimator (CCE) was a model developed for south-west Queensland to provide further informed decisions on assessing safe long-term carrying capacities. The model is at a property or paddock scale and is underpinned by empirical trials completed in the region. The estimates are produced by considering average annual rainfall and average long-term rainfall for the property. Actual forage growth is estimated after accounting for the negative impact of trees and shrubs, and the 'safe' utilisation rate is used as the proxy for carrying capacity, which is the portion of the forage growth that is calculated. There is a number of methods by which the mathematical relationships can be calculated, and CCE is just one approach. This is

not a stand-alone model, as it requires detailed land-type maps and property survey data (CSIRO *et al.* 2004).

GRAZPLAN is a suite of simulation models designed for temperate areas in Australia. The sub-models are the ruminant biology model, the soil moisture budget model, the pasture growth model, and the non-commercial soil nutrient cycling model. The nutrient cycling and pasture growth models use a daily time step, with the changes in digestibility accounted for. GRAZPLAN has taken a mechanistic approach to estimating pasture growth, which has been designed to integrate with an existing feed intake and animal production sub-model (Moore *et al.* 1997). GRAZPLAN can be applied to any breed of cattle, with potential intake calculated as a function of size and actual intake estimated as a fraction of the required intake (Kirschbaum *et al.* 2001). However, this model simulation is designed for temperate areas – particularly the pasture growth and climate sub-models – therefore, issues arise with application in rangelands (CSIRO *et al.* 2004).

4.5.3 GRASP

GRASP is an Australian rangelands empirical pasture growth model which links known empirical relationships between plant growth and water use. Initial development started in 1978, in parallel with application and field trials. The model has been applied to simulation studies since 1992 and remains under continual development (CSIRO *et al.* 2004). It was designed to meet specific objectives relating to grazing management of Australian rangelands. These were:

- objective assessment of drought and degradation risk in near-real time (Carter *et al.* 2000)
- simulating grazing options, including seasonal forecasting (Ash *et al.* 2000; McKeon *et al.* 2000; Stafford Smith *et al.* 2000)
- assessment of safe carrying capacity (Johnston *et al.* 1996; Hall *et al.* 1998)
- evaluation of the impact of climate change and CO_2 increase (Hall *et al.* 1998)
- reconstruction of historical degradation episodes (Carter *et al.* 2000).

GRASP has been used for a number of studies to explore relationships between pasture and animal variables. It has often been incorporated with other components to focus on particular areas of research. MacLeod *et al*. (2004) used GRASP to explore the interaction between stocking rates, animal production and economic outcomes at a Charters Towers property. The study estimated the impact of land condition on livestock performance and the economic implications. GRASP was used in conjunction with a beef enterprise economic model to determine the economic outcome. Figure 4.2 below demonstrates how GRASP was integrated for the research outcomes to be achieved.

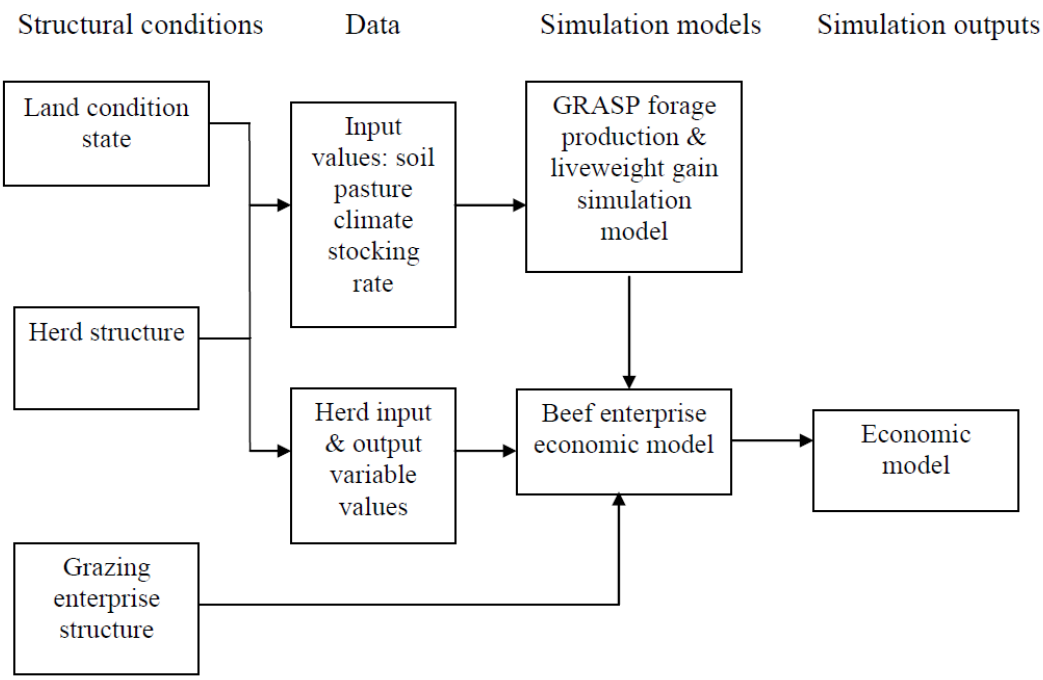

Figure 4.2 GRASP Integration

(MacLeod *et al.* 2004)

Chilcot *et al.* (2004) implemented GRASP to explore the interaction between 'safe' carrying capacity and loss of soil, which reduced productivity in the long term. The first stage was participatory research, where landholders estimated the 'safe' long-term carrying capacity in terms of pasture utilisation. This was then compared to the GRASP simulation 'safe' carrying capacity for a number of land types; the results indicated that the landholder pasture utilisation rate and the GRASP modelling were highly correlated. It showed that a loss of 100 mm of soil was

likely to lead to a 7 per cent to 31 per cent reduction in productivity of these pasture communities.

GRASP was also integrated into the ECOGRAZE project (Ash *et al.* 2002) to determine what the annual pasture and animal production for land in various conditions would be over a 100-year period; the farm economics was also linked in. To account for the complexity of the economic model accounting, production data was used to drive branding, mortality rates and sales in a spreadsheet model of enterprise economics (Ash *et al.* 2002).

GRASP is currently undergoing redevelopment as a part of the Department of Environment and Heritage Protection research and development program, however a previous version was used for integrating into the bioeconomic model for this research

4.6 Summary

Bioeconomic models have been used extensively in rangeland systems and have allowed a number of aspects of rangeland management and water quality outcomes to be explored. The research using these models has varied in scope from landholder- to catchment-based levels. The optimisation of these models and components of the various models have been designed to encompass the important aspects of the NRM issue or production trade-off. The method of optimisation has featured prevalently, and accounts for certain aspects of bioeconomic models with stochastic elements. Reviewing the important features and considerations allows this research to clearly define the biophysical and economic parameters. The following chapter describes the methodology of the bioeconomic model used in this research and its various components.

CHAPTER FIVE: BIOECONOMIC MODELLING METHODOLOGY

The development of a bioeconomic model requires a number of steps to ensure the required components are accounted for. This chapter focuses on designing a bioeconomic modelling to assess the costs of water quality improvements in rangelands grazing systems. The problem to be addressed and the key hypothesis is outlined, then the subsequent sections consider the design and details of the biophysical and economic sub-models of the bioeconomic model.

The review of the literature has provided the theoretical underpinnings of the bioeconomic modelling methodology. It has also described the case study application, with each problem having its own unique challenges and solutions. Bioeconomic modelling has a number of variables that need to be taken into account to ensure the problem being studied is addressed.

Modelling enables complex relationships to be shown in a way that allows predictions to be made about outputs. Often, a multidisciplinary team needs to be involved in the modelling process to ensure the correct interpretation of the problem and the objectives of the decision-makers and to allow increased understanding of the goals of the different parties involved.

To ensure the model represented the biophysical data correctly, a representative mix of land types was selected. Expert opinion was sourced about the case study areas so that meaningful results could be simulated. These experts included rangelands pasture scientists, agronomists, NRM officers, extension officers, landholders and catchment modellers (Appendix A).

The steps to the applied model development were as follows:

- Develop an understanding of the system
- Formulate the problem and objectives
- Define and collate resources
- Formulate conceptual model

- select model type
- construct model
- validate model
- generate model output to address objectives
- evaluate the modelling process.

(Doole and Pannell 2013)

The first four steps are used to provide the framework for this chapter with each subsequent section addressing a separate step. The last two sub-steps will be presented in the results in Chapter Six.

5.1 Develop an understanding of the system

This step involves understanding the different components and helps to differentiate between the symptoms of a problem and the problem itself, allowing a more appropriate problem formulation and identifying the system in question. To develop this understanding, a literature review is essential. Another critical component is to seek advice from people who have an in-depth understanding of a particular component of the problem or model. To achieve this and obtain a better understanding of the different components of the production system, various meetings were held with pasture scientists, beef extension officers, catchment modellers and landholders. These meetings resulted in the determination that the model needed two sub-models: a biophysical sub-model and an economic sub-model.

5.2 Formulate the problem and objectives

Before integrating the biophysical and economic models, the goals and objectives of the model need to be established.

The focus of the bioeconomic model is on the cost of sediment reductions from grazing systems in the Burdekin and Fitzroy Basins. As discussed in Chapter 2, the sediment exported is due to low ground cover at the end of the dry season due to

high stocking rates or grazing pressure; therefore, reduced grazing pressure can reduce the quantity of sediment exported (Brodie *et al.* 2003; McKergow *et al.* 2005; Packett *et al.* 2009). Given that bioeconomic modelling will effectively compare the net present value of alternative management scenarios, the opportunity cost of reduced grazing pressure will provide the social cost of sediment reductions (Bennett 2003; Whitten and Bennett 2005a).

The key task for the bioeconomic modelling is therefore to identify the costs of reducing sediment from rangelands grazing in the Fitzroy and Burdekin basins. The grazing system generates both profits and sediments, which both vary with grazing pressure. Secondary to this key task is to understand the interactions between the pasture production, animal production and subsequent economic outcomes.

It was assumed that landholders are profit maximisers and have perfect knowledge of seasons and pasture utilisation rates (per cent of total standing dry matter: % TSDM). These relationships lead to the mathematical process developed to further understand the objective of the model.

Economic results are expressed through a profit function or net present value that was generated using the following parameters:

P = Profit (Net Present Value)

F = Vector of biophysical factors that affect and influence stocking rate and animal performance

F_j = Level of pasture utilisation (% TSDM) j such that j = 10, 15, 20, 25…70

J_t = Tonnes of sediment exported given the level of pasture utilisation

S = Vector of economic implications

S_k = Variable income from production

C = Cost of economic implications

i = Discount rate

T = A block of 20 consecutive years

t = Years 1, 2, 3…20

The following describes the profit maximisation (P) for each level of pasture utilisation (Fj) and the associated level of sediment exports:

$$\text{Max P} = \sum_{t}^{T} F(S) \qquad \text{(Equation 3)}$$

$$S = \frac{S_k - C}{(1-i)^t} \qquad \text{(Equation 4)}$$

$$J_t = F_j\ (F) \qquad \text{(Equation 5)}$$

The formulation of a conceptual model and the development of the model then considered the economic problem in the integration of critical elements.

5.3 Formulation of a conceptual model

Given the complexity of rangeland grazing systems, there are a number of contributing factors along with grazing pressure that may affect the cost of sediment reductions. Broadly, these can be grouped into the interrelated characteristics of land type, location and starting condition.

Land type is a combination of soil type, landscape and vegetation. Productivity is related to soil type – and its ability to make nutrients available for pasture species and utilise water effectively – and to the pasture communities that subsequently occur in the particular soils. The types of soils follow landscape formations, which also affect the productivity of the soil. For example, floodplain soils have higher productivity than ranges soils due to the ability for the soil to get more organic matter in flood events. These soil types and locations in the landscape also affect the vegetation types.

Location is the key influence over climate, and rainfall. Climate is also a key driver of productivity and soil movement, with the two components being rainfall and growing days. Rainfall is essential to pasture establishment and growth, influencing the available TSDM. Growing days refers to days with the ideal conditions of warmth, sunlight and rainfall for the pasture species to grow. This means that the seasonal time of the year when rainfall events occur, how much rain falls over a particular time period, sunlight and warmth are fundamental to the productivity of the location.

The starting condition of a land type is a key driver of its long-term resilience and productivity. Starting condition is also a key driver to the long-term carrying capacity (management) and productivity of the land type. Land that is in C condition (see Table 2.3) does not have the resilience to a higher stocking rate compared to land which is in relatively better condition, such as B condition. This means that there may not be a high existing seed bank or established pastures to achieve a high level of productivity. It may also be that there are signs of scalding and water is not able to penetrate the surface easily, therefore running off.

These three characteristics then determine the enterprise that can operate and the grazing pressure that is economic optimum. To capture these key biophysical parameters, GRASP modelling was used. The characteristics also determine the variable costs and sales associated with the particular grazing pressure. Finally, a combination of the biophysical and stocking rates impacts the sediment that is exported from the property.

These relationships therefore lead to the concept model below:

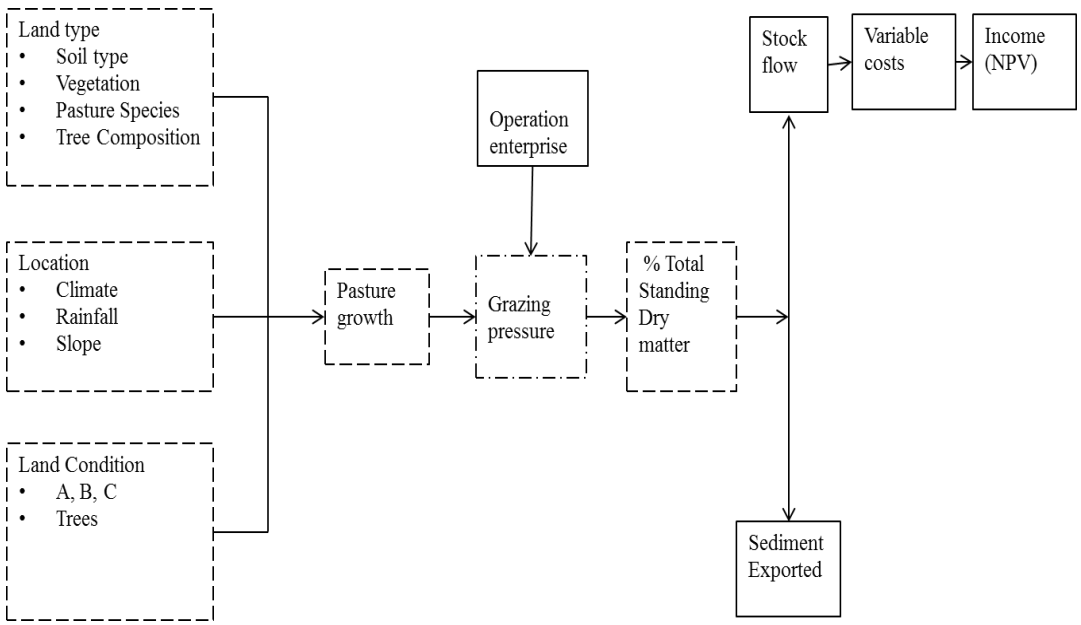

Figure 5.1 Bioeconomic concept model

5.4 Select model type

The construction of the model allowed numerous methodological issues to be tested and further understood. To develop the model and resolve the economic problem, the following characteristics for the model were selected:

Holistic: the model required a whole system to be described, encompassing the pasture species, the growth of pastures, rainfall, consumption, live weight gain, stocking rate, variable costs and the price achieved for cattle sales. For this component of the GRASP model, a pasture simulation model was selected.

GRASP operates on a daily time step model, integrating the land types soil characteristics, with rainfall, climate, pasture growth days, types of pasture species, and soil water holding capacity. These then allow live weight gain to be estimated and the stocking rate calculated for each of the pasture utilisation rates.

Empirical: as the economic problem involves estimating the opportunity cost and identifying where maximum profit can be achieved, the model is required to be predictive and therefore empirical. The ability to have a strong dataset allows for an empirical model to be developed to explain causal relationships.

Deterministic: Although a large number of years were selected, the model is still largely deterministic with no random elements, apart from rainfall. The model does not account for random elements such as fire or biosecurity outbreaks to be integrated into the bioeconomic model.

Normative: To solve the economic problem, an optimisation of the pasture utilisation is required; therefore, criteria for this to occur are also integrated into the model.

5.4.1 Construct model: GRASP

The model includes several components to allow for interactions between various physical processes. For example, the model had two aspects of degradation: land and pasture. Pasture degradation was defined as changes in pasture composition, which were determined by pasture utilisation; in other words, pasture eaten divided by pasture grown. Land degradation (soil loss) was defined as the surface cover, change in pasture species and erosion susceptibility, which was correlated with the standing pasture dry matter. Standing pasture dry matter can be defined as the net result of the addition of pasture growth, losses by animal consumption and losses by natural detachment. Stocking rate affects the processes of pasture growth, animal consumption and detachment (Day *et al.* 1997)

There are 46 major types of pasture communities in Queensland. The model over time has integrated the available data from enclosure and defoliation trials; these data were then integrated with pasture yields, nutrient concentrations and soil water measurements. Field studies were completed for the pasture communities not already modelled, making use of data from previous projects. Data were then used in the pasture growth sub-model. The model parameters included the plant-available water capacity, transpiration–use–efficiency and potential nitrogen uptake. For all available data (179 site by year combinations) the calibrated model simulated soil water accurately, and the absolute error for over the 700 field measurements in soil water (mm) was similar to measurement variation (Day *et al.* 1997).

The modelling of grazing effects using pasture utilisation requires the model to determine the effect of grazing growth, trampling and detachment. Data from

historical trials that have taken place were analysed using the model, with the objective being to examine how the model parameters change with grazing utilisation. Day *et al.* (1997) explains that the pasture communities differed with the parameters of grass basal area for Mulga, detachment rates in Mitchell grass, soil water range, nutrient uptake, and root/shoot partitioning in the black speargrass zone.The field trials were used to inform the GRASP relationships and empirical formulas occurred in Eastern Queensland (Table 5.1) and Western Queensland (Table 5.2):

Table 5.1 Trials used to inform GRASP in Eastern Queensland

Grazing trial	Nearest town	Pasture community	Years	Treatments
Brian Pastures P55-1	Gayndah	Southern black speargrass (cleared)	1961–1970	Three stocking rates in summer–autumn
Brian Pastures P55-2	Gayndah	Southern black speargrass (cleared)	1970–1979	Three stocking rates in either summer–autumn or winter–spring
Brian Pastures P55-3	Gayndah	Southern black speargrass (cleared)	1980–1984	As above in P55-2 with/or without burning in spring
Ladies Mile	Gayndah	Southern black speargrass (cleared)	1989–1994	Five periods of deferring grazing
Galloway plains	Calliope	Central black speargrass cleared	1988–1995	Six stocking rates
Kangaroo Hills	Seaview Range (West of Ingham)	Northern blackspeargrass (oversown with Townsville stylo)	1965–1975	Two stocking rates with or without clearing and phosphorus

(Day *et al.* 1997)

Table 5.2 Trials used to inform GRASP in Western Queensland

Location	Nearest Town	Pasture Community	Years	Treatments
Toorak	Julia Creek	Mitchell grass	1985–1995	Five levels of pasture utilisation
Eastwood	Blackall	Buffel grass	1967–1983	Three to four constant stocking rates
Burenda	Augathella	Mitchell grass	1967–1989	Five levels of utilisation

Arabella	Charleville	Mulga pastures	1977–1986	Four levels of utilisation
Gilruth Plains	Cunnamulla	Mitchell grass flood plains	1941–1954	Three constant stocking rates

(Day *et al.* 1997)

The four major studies which have been integrated into GRASP were the mulga lands, southern black speargrass, central black speargrass and northern black speargrass. Pasture growth was estimated using the pasture growth sub-model. All of the studies estimated that the 'safe' pasture utilisation rates were in the range of 15 per cent to 25 per cent of average annual pasture growth. However, at the same time the ABS data suggested that the utilisation rate can be over 15 per cent, and grazing trial data showed that short-term animal performance increased with a marginally higher utilisation rate (Day *et al.* 1997).

The model also includes an animal intake and live weight gain sub-model, which depends on a number of variables. There are two feed-quality restrictions on the utilisation of feed. The first is the restriction of limiting intake by the proportion of growth eaten by the animal, and the second occurs at low levels of TSDM. The animal intake is then calculated by (Day *et al.* 1997) as:

Animal intake = *intake restr* * [(*pot lwg/Days per season* +1.058)] / 0.304

(Equation 6)

Where:

Animal intake = intake of biomass (kg beast^{-1})

intake restr = restriction of intake by the animal

pot lwg = user-defined potential live weight gain for the season (kg beast^{-1} per 91 days)

Days per season = number of days in the season (default = 91.25)

Finally, the animal intake equation is substituted to determine the live weight gain.

Daily lwg =0.304 Animal intake*

Where:

daily lwg = live weight gain per weaner steer (kg per head per day)

animal intake = intake of biomass

A weaner steer is calculated as a 200 kg live weight beast. The total amount of biomass eaten by all animals is then calculated by:

*Eaten = Animal intake * stock equivalent*

Where:

eaten = total biomass eaten by animals (kg)

animal intake = intake of biomass (kg beast^{-1})

stock equivalent = number of beasts equivalent to weaner steers at 200 kg live weight

The model also has a sub-model for sediment run-off. The data for this function was collected on red duplex soils near Charters Towers and helped to create an algorithm for soil run-off. Daily run-off volume is calculated from ground cover, daily rainfall, rainfall intensity and soil water deficit. The algorithm to calculate run-off is explained by (Day *et al.* 1997) as:

Run-off = *cover term* * [*rain* – (1-(*rain intensity*)/110) * *sw deficit]*

(Equation 7)

Where:

cover term = cover index

rain = daily rainfall (mm)

rain intensity = maximum rainfall intensity in a 15-minute period limited to 110mm hr^{-1}

sw deficit = soil water deficit of the top two profile layers (mm)

Any rainfall that does not run off is assumed to infiltrate the top profile layer of the soil.

The model does have some deficiencies that require further field work and model development. Day *et al.* (1997) explain that the majority of the model errors occur under conditions of above average rainfall, when nutrient limitations and or phonological developments are the major processes affecting plant growth. This, however, is not regarded as a major limitation for the purposes of the present study.

5.5 Validate model: Economic Model

The economics model was designed to use the GRASP output and put it into an economic framework. The economic model consisted of three separate sub-models: herd dynamics, variable costs and sales, and a discounted cash flow. The stock flow sub-model was dynamic, with the stock flow changing annually to match the pasture utilisation requirements set in GRASP. The variable costs of animal health, transport for sales and other associated operational costs were accounted for on a per head basis. The economics model was based on a representative property, and one of the possible enterprise options for the land type was selected (the economics model captured the boxes in the Figure 5.2 concept diagram that do not have broken lines).

The construction of the model comprised various components that needed to be integrated. In this phase of development, technical experts played a key role in developing parameters. This section will step through the construction of the model. Firstly, an overview of the case study will be discussed, followed by the components of the biophysical model and economic model.

5.5.1 Case study characteristics

This section focuses on the characteristics of the Fitzroy and Burdekin catchments to understand the implications and important aspects and how to best inform the

biophysical components of the model. The section then justifies how these sub-models have considered the relevant methodology components.

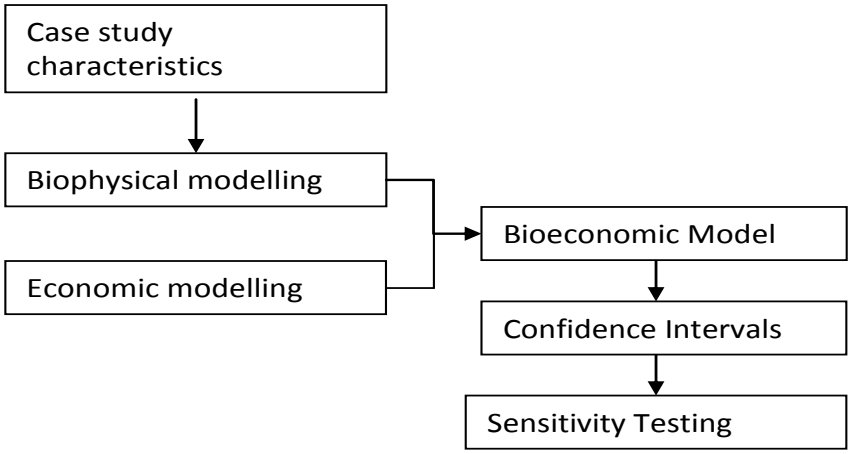

Figure 5.2 Case study methodology

The bioeconomic model is designed at the catchment level to encompass the two largest catchments for the GBR (Fitzroy and Burdekin). These catchments both exhibit large amounts of heterogeneity in land type characteristics; representative land types were selected for separate model simulations based on various characteristics.

5.5.2 Fitzroy Basin land types selected

The land types were selected based on geographical location, percentage of the catchment that consisted of particular land types, decreasing ground cover over the past four years, erosion susceptibility and sediment run-off. The land types for each catchment can be grouped into the larger classifications of alluvial land types, bluegrass downs land types, brigalow scrubs land types, coastal land types, eucalypt woodland land types, mountains and ranges land types, and sandy land types. This reflects the additional aim of having diversity from the broad land type groups.

As there is high climatic variability in rainfall, temperature and slope, the geographical location was also important to ensure that different aspects of the catchment were represented. Dougall *et al*. (2008) report that in the Fitzroy Basin, approximately 50 per cent of the total flow from the Isaac sub-catchment discharges into the GBR. The eastern part of this sub-catchment is the Connors region, which has relatively high average annual rainfall (Dougall, Carrol et al. 2008). Due to

these factors, it was assigned as an area of interest. The predominant land type in this area is narrow-leaved ironbark ranges, comprising 6.61 per cent of the catchment. Narrow-leaved ironbark was therefore selected to be modelled using the climatic data from the Balaclava Mountains.

Brigalow blackbutt was the land type selected to represent the land grouping of brigalow scrubs. The Fitzroy Basin catchment comprises 7.94 per cent of Brigalow blackbutt, with a high percentage of the land type located in the central area of the catchment. Abbott (2008) and Karfs, Abbott *et al.* (2009) have also defined areas within the two catchments that have increasing bare ground, and these locations were also taken into consideration while selecting the climate location. Bare ground index mapping demonstrated a downward trend in mean bare ground cover around the Dauringa area, with the average cover less than 60 per cent. Based on this, the climate station selected was Blackwater, the largest town in proximity to Dauringa. The mean bare ground index for the catchment can be seen in Figure 5.3.

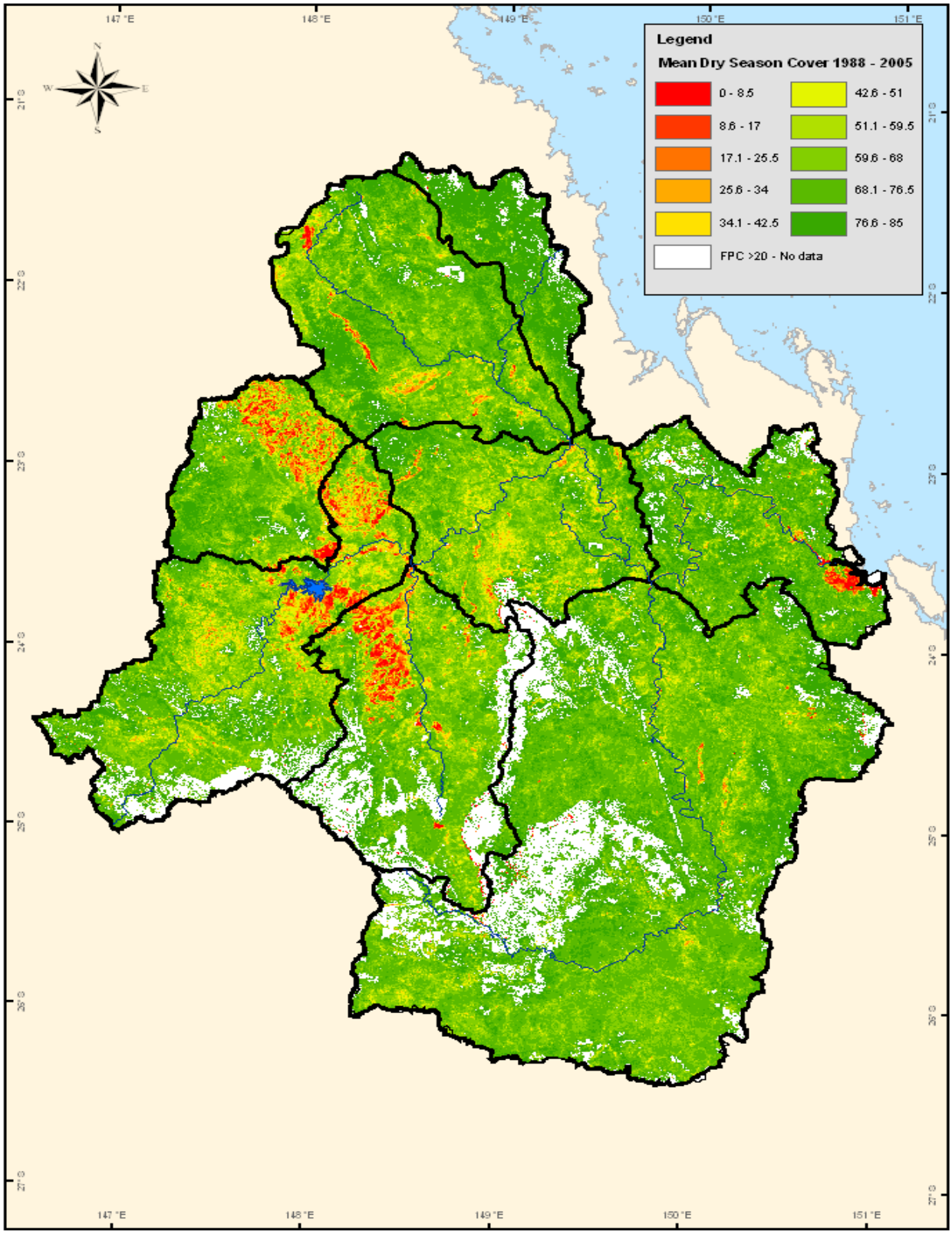

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Figure 5.3 Bare Ground Index map of the Fitzroy Basin

Narrow-leaved ironbark woodlands was the third land type chosen. It was selected based on the high percentage of this land type found in the southern part of the region. In total 4.78 per cent of the Fitzroy Basin is narrow-leaved ironbark woodlands. The climate station initially used to undertake the modelling was Galloway Plains, due to a long-term grazing trial taking place. However, this location was changed to Duaringa as it was deemed to be more representative climate and location for the land type.

The fourth land type was selected based on an alluvial land group in the western part of the catchment. To represent the group, coolibah floodplains was selected using the climate station of Mantuan Downs in the Springsure area. Coolibah floodplains occur in alluvial plains and consist of coolibah woodland with an understorey of scattered clumps of brigalow and bauhinia. The preferred pasture species is Queensland bluegass, forest bluegrass, silky browntop, bull and curly Mitchell grass and couch grass. The soil is described as black cracking clay, which has variable sodic areas (Queensland Government 2010).

Spotted gum on ranges, the fifth land type selected, was chosen for its high percentage of land in C condition. Duaringa was the selected climate station to undertake the modelling. This land type can be best described as occurring on mountains and ranges, with rocky and shallow soils; an understorey of wattles, zamia and red ash may also be present. The preferred pasture species is black speargrass, kangaroo grass, hairy panic and desert blue grass. Spotted gum is a commercial timber species and the land type occurs on steep slopes with rocky and shallow soils (Queensland Government 2010).

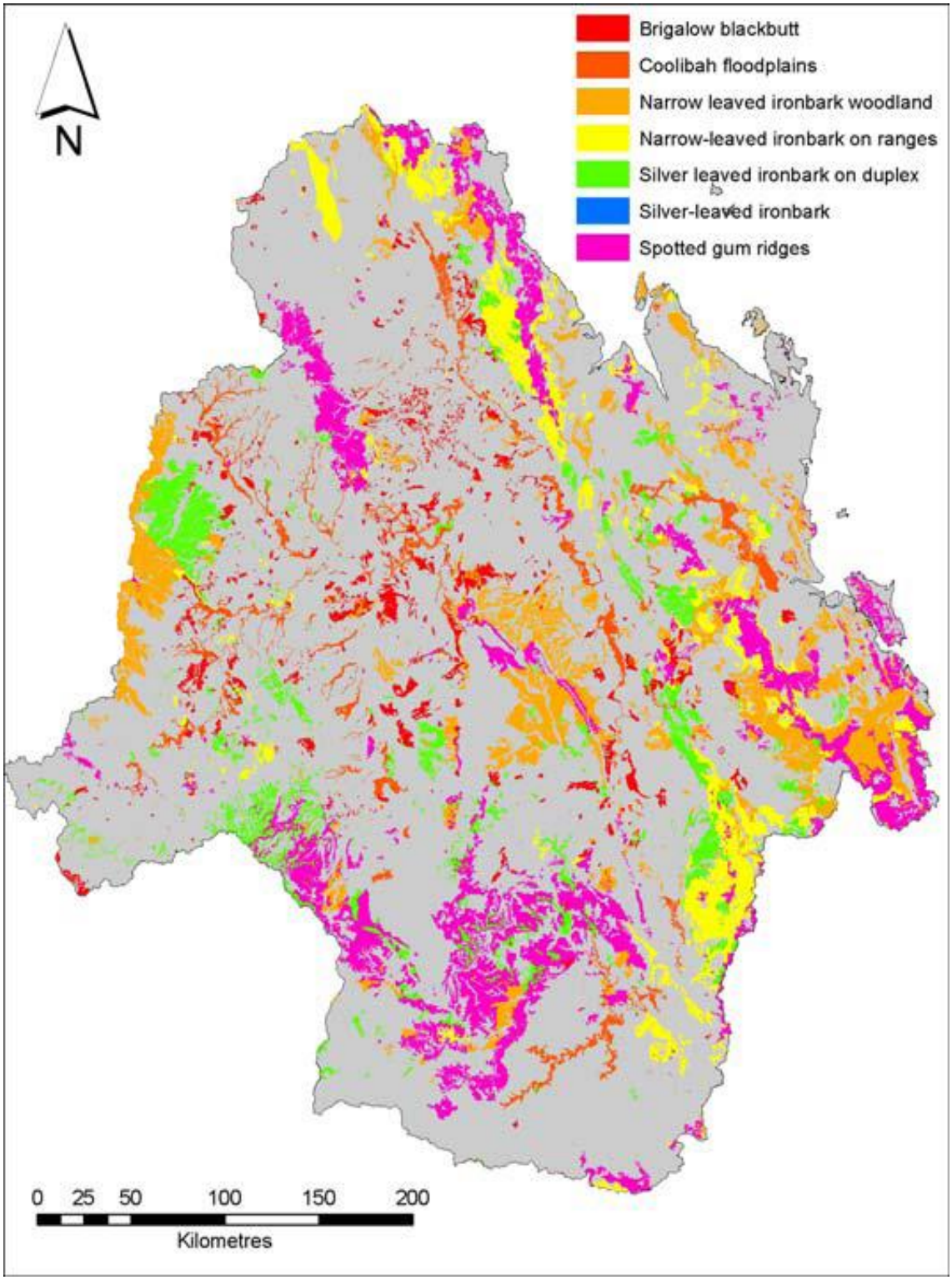


Figure 5.4 Land types selected for the bioeconomic model in the Fitzroy Basin

5.5.3 Burdekin Basin land types

Based on the same criteria as for the Fitzroy Basin land type selection, silver-leaved ironbark was selected. Previous grazing and soil loss trials have shown that silver-leaved ironbark has a high susceptibility to erosion and a decreasing trend in ground cover. Silver-leaved ironbark was selected to be modelled for two climate stations, which were on the border of the Fitzroy and Burdekin Catchments. Nebo and Springvale climate stations are actually just inside the Fitzroy boundary; however, they are trial sites where extensive soil modelling has occurred. The productivity between the two varies due to higher rainfall at Nebo and the Springvale site having duplex soils.

Finally, goldfields country was selected with the climate station of Virginia Park. Goldfields is a dominant land type that occurs extensively throughout the Burdekin Basin (Figure 5.5). The land type is open woodland with patchy understorey of false sandalwood and corkwood wattle. The preferred pasture composition is desert bluegrass, Queensland bluegrass, curly bluegrass, black speargrass and kangaroo grass. The soil is self-mulching black, sometimes red and brown and cracking clays. The land types selected are illustrated in Figure 5.5.

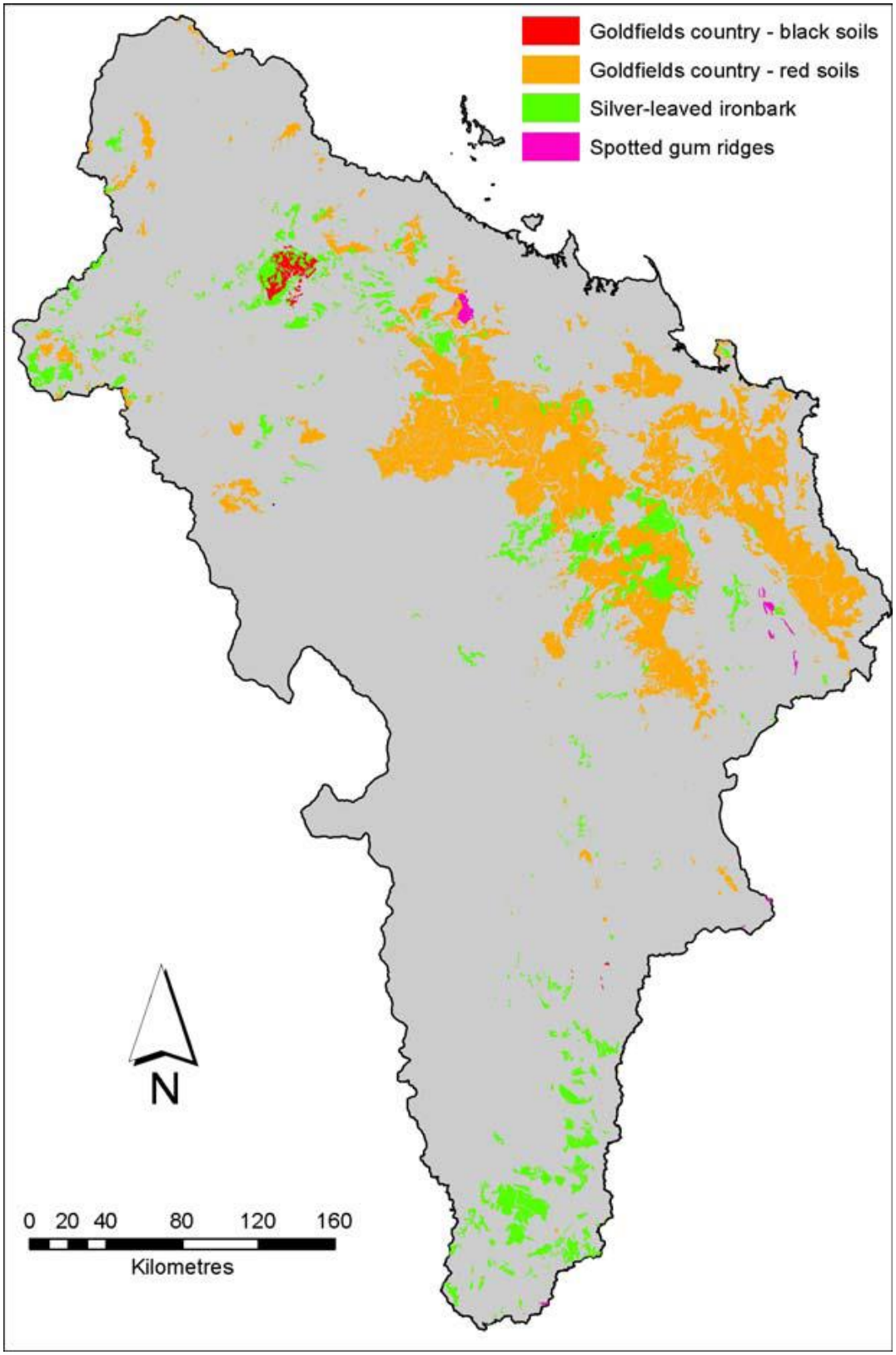


Figure 5.5 Land types selected for the bioeconomic model in the Burdekin Basin

5.5.4 Biophysical simulations

The biophysical model consisted of two sub-models. These were the pasture production model and the animal production model. The two have feedback loops into each other, depending on the rainfall and pasture growth. This section explores the temporal and spatial scale of the model, along with the recursive features of the model.

5.5.5 Temporal and spatial scale

To understand the interaction between different management practices, the bioeconomic model was developed at the landholder scale.

To account for the variability in rangeland rainfall and climate over time, twenty initial start years were selected. The years were chosen using a random number generator for the years between 1893 and 1983 and were selected in this method to ensure independence for statistical analyses. This method also accounts for the stochastic variability of rainfall over a large spectrum of years.

From each of these initial start years, 20 consecutive years of performance were modelled (e.g. 1896–1916). A period of twenty years was selected as this represents an indicative period in which management is held by one particular party.

The twenty starting years for each of the 20-year simulations were as follows:

1896	1917	1942	1962
1902	1924	1945	1967
1904	1929	1949	1971
1912	1936	1956	1981
1915	1941	1960	1983

For these start years, three additional variables were identified: tree basal area, initial land condition and grazing pressure. Tree basal area can be described as the square meters in one hectare that trees compete with pasture for nutrients and water.

Tree basal area was implemented to reflect the 'average' type of trees found in the remnant vegetation on the land type. To simulate a cleared landscape for all of the land types, a tree basal area of 0 m²/ha was also included. Table 5.3 demonstrates the tree basal area simulated for each of the land types.

Table 5.3 Tree basal area

Land type	Tree basal area (m^2/ha)	
	Cleared	Average
Brigalow blackbutt (Dawson's gum)	0	3
Narrow-leaved ironbark on mountains and ranges	0	15
Narrow-leaved ironbark woodlands	0	9
Coolibah floodplains	0	6
Goldfields country – red soils	0	3
Silver-leaved ironbark	0	7.5
Silver-leaved ironbark on duplex	0	5
Spotted gum ridges	0	11

5.5.6 Recursive modelling

Land condition has been defined by the Grazing Land Management framework (Chilcott *et al.* 2005) as the capacity of land to respond to rain and produce useful forage. The ABCD land condition framework has been widely used and adopted in the northern Australian grazing industry as a measure of how well the grazing ecosystem is functioning. The framework classifies land condition through pasture species composition, weeds, woodland thickening, bare ground and soil condition. 'A' condition land is described as good coverage of palatable, productive and perennial pasture species, less than 30 per cent bare ground in most years, few weeds, good soil condition with no surface erosion and no woodland thickening. 'D' condition land has a general lack of any perennial pasture species, forbs or grasses, severe erosion or scalding resulting in a hostile environment for plant growth, and thickets of woody plants which cover most of the area (Chilcott *et al*. 2005) (see Table 2.3 for further details). 'B' and 'C' condition land lie between these extremes.

Reflecting 'A', 'B' and 'C' condition, three initial start conditions of the pasture were simulated. These were modelled to provide insight into the impacts of grazing pressure on the land type and the impact that this has on sediment run-off and

economic performance. The start conditions selected were reflective of an 'A' condition, 'B' condition and 'C' condition pasture species composition and TSDM. The start conditions selected were as follows:

Table 5.4 Start condition for land types, % perennials

Land type	Land condition (% perennials)		
	A condition	**B condition**	**C condition**
Brigalow blackbutt (Dawson's gum)	88	70	20
Narrow-leaved ironbark on mountains and ranges	80	70	20
Narrow-leaved ironbark woodlands	80	70	20
Coolibah floodplains	88	70	20
Goldfields country - red soils	88	70	20
Silver-leaved ironbark	88	70	20
Silver-leaved ironbark on duplex	88	70	20
Spotted gum ridges	88	70	20

To examine the full range of impacts on sediment run-off and the relationship between sediment run-off and grazing pressure, 13 grazing pressure intervals were simulated with each of the climate stations. The grazing pressures were maintained for all of the land types selected and were based on the TSDM left at the end of the growing season (April). The per cent utilisation of this remaining standing dry matter was as follows:

10%, 15%, 20%, 25%, 30%, 35%, 40%, 45%, 50%, 55%, 60%, 65%, 70%

Utilisation thresholds that dictate whether the resource condition degrades or recovers were derived for each land type, following consideration of the long-term safe utilisation rates (Appendix B). The utilisation threshold values were based on utilisation of green material of total growth by the end of the growing season (30 April).

The decline in the per cent of desirable perennial grasses under heavy grazing was simulated by linking per cent perennial grasses in a pasture to a condition scale derived by Ash and Smith (1996) from observed data (McKeon *et al.* 2000). The condition of the resource (per cent perennials) was indicated by a scale that ranges from state 0 (lightly grazed, 90% perennials) to state 11 (heavily grazed, zero per

cent perennials). Depending on the condition of the resource, the state of a land type can change up and down in response to the impacts of heavy grazing. The potential rate of change between states for degradation or recovery is one state annually. Across the land types, the loss of desirable perennial grasses associated with heavy utilisation was simulated by changing model parameters as described in Table 5.5.

Table 5.5 Degradation and recovery threshold values based on utilisation of green material of total growth by the end of the growing season (30 April) for eight land types in the Burdekin and Fitzroy Basins

Land type	Recovery Threshold % Dry Matter Utilisation	Degradation Threshold % Dry Matter Utilisation
Brigalow blackbutt	22%	40%
Narrow leaved ironbark on mountains and ranges	12%	26%
Narrow leaved woodlands	14%	28%
Coolibah floodplains	18%	35%
Goldfields – red soils	15%	35%
Silver-leaved	15%	35%
Silver-leaved ironbark on duplex	15%	35%
Spotted gum ridges	10%	25%

The GRASP model uses an equation for daily soil loss, which was derived from soil loss measured on hillslopes in the Burdekin grazing lands (Scanlan *et al.* 1996). This equation has no adjustment for the known effects of hillslope length and slope on soil loss (Renard *et al.* 1997; Silburn 2011). Therefore, estimated soil losses were calculated for the typical slopes and slope lengths at the sites where the data was collected. The Revised Universal Soil Loss Equation (RUSLE) (Renard *et al.* 1997) used length (L) and slope (S) adjustments (or LS), which have been shown to relate well to measured soil losses on a range of slopes in grazing lands in Queensland (Silburn 2011b; Silburn *et al.* 2011a).

To correct soil losses to greater or lesser slopes and lengths found in the land types modelled, the GRASP soil losses were divided by the average Universal Soil Loss Equation (USLE) length-slope (LS) factor for the Scanlan *et al.* (1996) sites and multiplied by the average LS for the modelled land type, which were derived from

a digital elevation model. Thus, on land types typically steeper than modelled by Scanlan *et al.* (1996), soil losses will be increased; on flatter land types, soil losses will be decreased. Sediment exported to the GBR was calculated using a delivery percentage of 12.5 per cent , which is the estimated level of sediment movement in a hectare that actually leaves the paddock (Dougall *et al.* 2008).

Rainfall impacts not only pasture growth but also the sediment exported. GRASP modelling was completed for the following land types with these average long term rainfall levels.

Table 5.6 Average long-term annual rainfall

Land type	Location	Average long-term rainfall (mm)
Brigalow blackbutt	Blackwater	580 mm
Narrow-leaved ironbark woodlands	Blackwater	580 mm
Narrow-leaved ironbark on ranges	Balaclava Mountains	523 mm
Coolibah floodplains	Springsure / Mantuan Downs	964 mm
Silver-leaved ironbark	Nebo	733 mm
Silver-leaved ironbark – on duplex	Springvale	590 mm
Spotted gum ridges	Duaringa	715 mm
Goldfields	Virginia Park	599 mm

5.5.7 Animal production model

Annual live weight gain was calculated from per cent utilisation and percentage of days during the year where pasture growth index was above a threshold. The growth index is calculated using green growth, soil water, nitrogen and temperature indices. An additional 15kg/hd/yr to live weight was modelled to occur on years when pasture is burnt.

The biophysical simulated results were then integrated with the economic model to synthesise the data and produce the outputs. The syntheses of data from GRASP relied heavily on the economic model to further process and integrate the data. This ensured that there were feedback mechanisms between aspects of each model. It must also be noted that there a number of animal production parameters such as animal intake, and live weight gain refer to Section 5.41 for further information.

To account for mortality branding rates and drought feeding the following parameters were developed.

- Mortality was calculated in the stock flow sub-model for both the dry herd and for the breeding herd. This was dependent on the live weight gain and was derived from MacLeod *et al*. (2004). The dry herd was considered to be yearling heifers and steers up to two years old. A maximum breeding mortality rate of 20 per cent was applied. The equation was calculated as a function of live weight gain.

  $$\text{Mortality (breeders)\%} = 6 + 94e^{-0.027(LWG+50)} \qquad \text{(Equation 8)}$$

  $$\text{Mortality (dry stock)} = 2 + 88e^{-0.034(LWG+50)} \qquad \text{(Equation 9)}$$

- Branding rates for the breeding herd were based on MacLeod *et al*. (2004) and were determined as a function of live weight gain. This had a maximum rate of 75 per cent and a minimum of 30 per cent to reflect the variation in typical herd in the region. The equation is:

  $$\textit{Branding \%} = 30 \leq 15.6 + 0.488 \times LWG \leq 75 \qquad \text{(Equation 10)}$$

- It was assumed that in years where there was less than 50 kg live weight gain that drought feeding would occur, based on the work completed by MacLeod *et al*. (2004). In the model, when live weight gain was less than 50 kg per head then a urea-molasses lick supplement (Urea 8%-M8U) was fed. The feeding rule was two days of M8U feeding for each kilo of live weight gain less than 50 kg. For example, when the live weight gain was simulated by GRASP to be 10 kg, then the M8U ration was fed for 80 days. Where GRASP simulated that there would be a live weight loss, then it was assumed a ration of urea-molasses fortified with cottonseed meal (urea 3%, cottonseed meal 10% – M3UP38) was fed with one day of feeding for each kilo of weight loss. For example, if an animal was simulated to lose 20 kg, then there would be 20 feeding days of M3UP38 supplement and 100 days of M8U supplement.

5.6 Economic model

To combine the biophysical results into an economic model, the livestock and pasture outputs from the GRASP data were integrated into an economic model. The economic model was designed to explore the economic implications of different pasture utilisation rates for the different land types. This also enabled the dollar cost of reducing a tonne of sediment to be estimated for each land type for a range of pasture conditions. This component of the model also considered the scale and unit of decision-making, along with the recursive ability of the model.

The GRASP simulation provided much of the production data assumptions and the soil loss to determine the economic and environmental implications. The variables that provided the most information were the head of cattle per km^2, live weight gain per head and run-off.

5.6.1 Unit of analysis decision-making level

The unit of analysis was at the individual property and landholder level. This level was selected due to their ability to change management practices and to identify opportunity cost for changing the management systems at the enterprise level. It also allowed the model to integrate the biophysical outcomes into the economic model. The key parameters that were utilised in the economic sub-model were the Adult Equivalents (AEs) per hectare, which is a unit of measurement equivalent to a 400 kg steer per hectare, and live weight gain per animal (Figure 5.6).

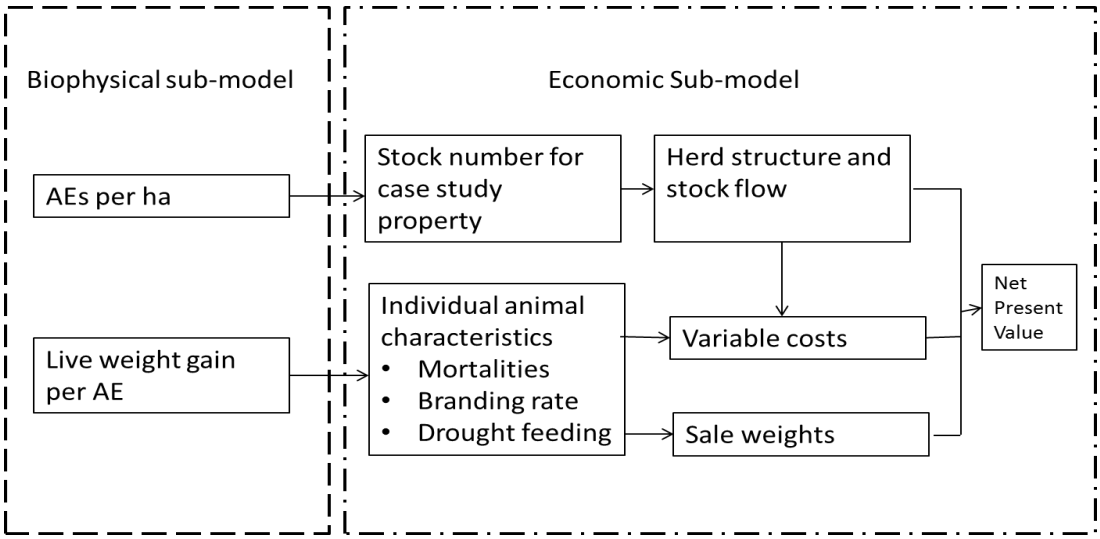


Figure 5.6 Integration of biophysical components into the economics sub-model

The economic model developed a 20-year flow of herd dynamics to match the climatic data and the simulated stocking rate. This was done to demonstrate the economic implications of adjusting stocking rate, and to allow the production data to be fully reflected. This also accounts for the fact that there may be a lag effect of adjusting the stocking rate, in that the base herd may then require a couple of years to re-build or de-stock depending on the year. For the stock flow to reflect a representative property, a number of assumptions were made, including:

- The AEs given to each animal class were based on the BreedCow Dynama program and are listed below:

Table 5.7 Animal equivalent per animal class

Animal class	Equivalent AEs (1AE = 400 kg steer)
Calves	0.35
Heifer weaners	0.28
Steer weaners	0.28
Heifers 1yr	0.73
Steers 1 yr	0.78
Heifers 2 yrs	0.98
Steers 2 yrs	1.14
Cows 3–10 yrs	1.1

- A base herd was initially developed for year one; however, depending on the available AEs determined by GRASP, the base herd was proportionally

adjusted across all animal classes or divided across all animal classes to adjust stock numbers up or down, depending on available pasture. From this base herd, the percentage sales and the percentage of males and females were determined.

- When there was an opportunity to purchase trade cattle, they were purchased in numbers that maintained the ratio of females to males of the base herd.

The land type, location, land condition, pasture growth and sediment exported were factors included in the biophysical sub-model. Variable costs and income were factors in the economic sub-model. The grazing pressure section was pivotal for the stock flow, which influenced the animal production depending on the live weight gain. The variable cost and income also had numerous assumptions, as follows:

- The representative property size was 5,000 hectares for the Fitzroy land types and 10,000 hectares for the Burdekin land types. The herd size varied by productivity of the land type. It was assumed to be a homogenous block of one land type.[3]
- The enterprises selected were a fattening operation producing Japanese oxen for more productive land types and a breeding operation producing trade store steers for the low productivity land types. This was done to reflect local production in the area and took effect in the stock flow sub-section of the model.
- The percentage of sales each year and the sale prices were kept constant; however, in order to account for the difference in turn-off age, Japanese oxen were held for two years and the trade steers were turned off at the yearling age group. Sales and cost prices were kept constant over the 20 year time period reflecting the competitive nature of the northern cattle market. The percentage sales and the price per kilo are listed below:

[3] Although it is acknowledged that does not reflect reality, this assumption was required to undertake the modelling and match the biophysical simulations.

Table 5.8 Percentage of herd enterprise sales and price

Percentage of base herd sold (%)			
Animal class	**Japanese oxen**	**Trade store steers**	**Price $/kg**
Calves	0	0	0
Heifer weaners	30	30	1.69
Steer weaners	0	0	0
Heifers 1 yr	42	42	1.57
Steers 1 yr	0	100	1.90
Heifers 2 yrs	42	42	1.35
Steers 2 yrs	100	0	1.90
Cows 3–9 yrs	42	42	1.35
Cows 10 yrs	100	100	1.35

- In order to ensure that the pasture utilisation is at the required level, particularly at the higher utilisation levels, there was a high amount of variation in stock numbers from year to year. In these cases there was drought selling and purchasing. In order to ensure that the required reduced number of AEs was met, additional drought sales occurred across the herd. Of the AE reduction required, 15 per cent was in weaners, 30 per cent in steers 1 year old, and 45 per cent in breeders.

The following assumptions took place in the variable costs section of the model:

- When drought selling occurs, a price penalty was incurred on the cattle sold. This was priced at $1.25 per kilo for breeders and $1.45 for steers.
- If the following year AEs increased, they were bought back at the same male to female ratio as the base herd in year one.
- The variable cost parameters used were the following:

Table 5.9 Variable herd costs

Drought feeding		Level kg/hd/day	$/kg	Total $/hd/day
Breeders				
M8U		5.5	0.3	1.65
M3UP38		4.5	0.3	1.35
Weaners				
M8U		5.5	0.3	1.65
M3UP38		4.5	0.3	1.35
Steers				
M8U		5.5	0.3	1.65
M3UP38		4.5	0.3	1.35
Variable costs	**Dip, drench vaccine, tag**	**Hay**	**Selling costs**	**$/hd total**
Weaner steer	11.22	10		21.22
Weaner heifer	11.22		14.3	25.52
Yearling steer	6.7		15	6.7
Yearling heifer	2.22		14	16.52
2 yr steer			18	18.07
2 yr heifer	1.11		16	17.11
3 yr + cow	1.11		16	17.17
Interest on additional livestock capital	8%			

- Interest on livestock capital was set at 8 per cent, which reflects the interest rate of a business loan at the time of analysis.
- Finally, the outcomes of the 10 per cent TSDM were used as the base, and the remaining pasture utilisations were compared in a discounted cash flow analysis. A real discount rate of 6 per cent was used to compare the marginal accumulated benefits over the 20-year period.

5.7 Summary

This chapter describes the objective of the bioeconomic model, along with the sub-models that exist within it, and a description of how they integrate. The features and considerations of bioeconomic modelling, described in Chapter Four, have been considered as the model has been developed, and the bioeconomic model allows some of the methodological impacts to be considered and tested.

CHAPTER SIX: UNDERSTANDING THE COSTS OF REDUCING SEDIMENT FROM RANGELANDS GRAZING SYSTEMS[4]

The bioeconomic results enable a greater understanding of the cost of sediment reductions and the variance that occurs due to land type, tree basal area and start condition parameters. Initially, an overview of the main parameters that have been explored in the model is presented. Following this, two land types will be presented in-depth to provide an understanding of the two general forms that the results achieved. The effects of land type, land condition and tree basal area will then be explored, with an example of how the results can be optimised. A discussion of the results will then be given and finally a summary of the results of the different land types is presented.

6.1 Bioeconomic model output

The bioeconomic modelling of the various land types resulted in a matrix of parameters to be explored. The land type was modelled for the three land condition states of A, B and C. For each of these the impact of trees was explored, with 0 tree basal area and with the 'average' tree basal area for that land type. Finally, the various grazing pressures of 15–70% TSDM were estimated (Figure 6.1).

[4] Components of this chapter have been published in Star, M., Rolfe, J., Donaghy, P., Beutel, T., Whish, G. and Abbott, B. (2013). Targeting resource investments to achieve sediment reduction and improved Great Barrier Reef health, Agriculture, Ecosystems & Environment 180, 148-156.

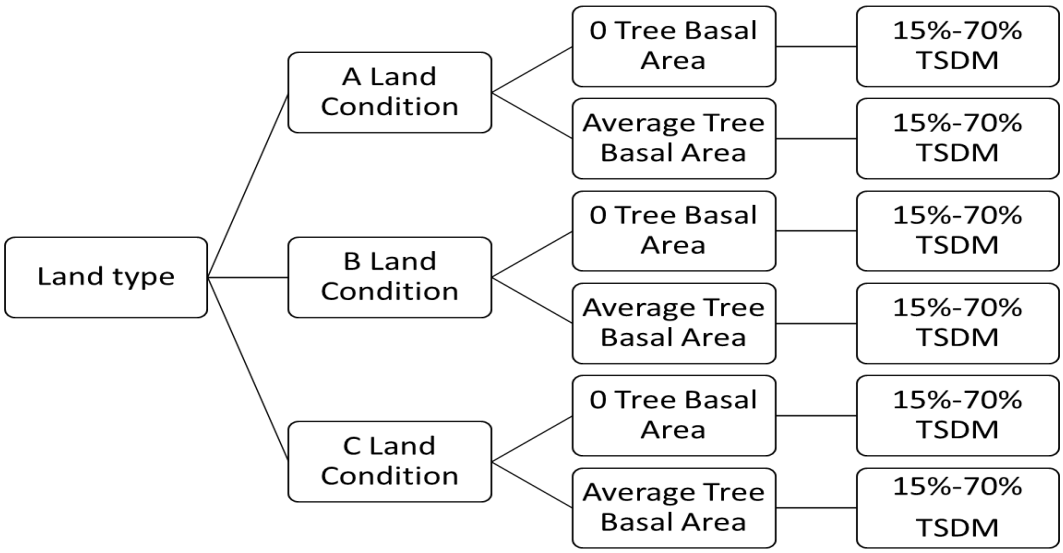


Figure 6.1 Output from the bioeconomic model for each land type

The various parameters have different implications for results, and these will be further explored in subsequent sections; however, as there are eight land types with this type of output, two examples will be stepped through and then the remaining land type results will be summarised. To allow comparisons and to understand the outputs, a lower productivity and a higher productivity land type were selected, with the former land type in the Fitzroy Basin and the latter in the Burdekin Basin. Brigalow blackbutt and Goldfields were the two land types selected to demonstrate the results.

As the trade-offs between sediment, pasture utilisation and profit were explored, the two results that will be explained are:

- the net present value, which was used as a proxy for income. The net present value represents the marginal change between 10% TSDM and the other 12 levels of grazing pressure. Calculated for the 20 random start years over a 20-year period, the value reported is the average net present value of the results. The tonnes are representative of the estimated total sediment per hectare exported over 20 years, with the 20-year blocks averaged
- the dollars per tonne of sediment , which represents the opportunity cost of operating at the specific grazing pressure divided by the tonnes exported; it is provided to understand the incentive or cost required for graziers to reduce grazing pressure.

6.2 Results for Brigalow blackbutt and Goldfields

6.2.1 Brigalow blackbutt, A condition, 0 trees

The results for Brigalow blackbutt will be stepped through, followed by the gold fields with land condition and tree basal area parameters explored. This section aims for a greater understanding of the impact of different landscape characteristics and parameters modelled for a productive land type (Brigalow blackbutt) turning off Japanese oxen and for a low productivity land type (Goldfields) turning off 18-month-old steers.

The Brigalow blackbutt had a start condition of 88% perennials, indicating that it was classified as A condition; the 0 tree basal area is defined as a cleared area; and the average tree basal area (m^2/ha) in this land type is 3 (see Table 5.3). The most profitable pasture utilisation rate is at 55% pasture utilisation (TSDM), where the net present value is $904,211. At this level of pasture utilisation, there are 1,895 AEs stocked on the 5,000 ha property. At 55% pasture utilisation, the average total sediment load leaving the paddock over 20 years is 3,730 tonnes. If the pasture utilisation rate is decreased to 50% in an attempt to reduce sediment movement the estimated cost per tonne of sediment reduced would be an average cost of $360.92; if the landholder were then to reduce pasture utilisation a further 5% TSDM to 45% TSDM, the cost per tonne of sediment reduced is $604.87. Conversely, at 60% pasture utilisation the grazier is operating past the most economic optimum point and has forgone $91,788 in income. At this level of pasture utilisation there is also an increase of 2,806 tonnes of sediment exported over the 20-year period.

The bell-shaped curve of the net present value provides insights for policy and programs to reduce sediment emission. Firstly, in relation to landholders operation to the right of the peak, in this case from 70% pasture utilisation to 60% pasture utilisation, there is an opportunity for extension and education to be the most effective mechanism for encouraging graziers to reduce pasture utilisation and increase profits. However, to allow graziers to reduce pasture utilisation back from the economic optimum point will require other policy mechanisms, such as incentives, as profit would be forgone to achieve this reduction in pasture utilisation.

It can be observed that the sediment exported increases by relatively small increments until 55% pasture utilisation. At this point there are significant increases in sediment exported until 70% pasture utilisation. This sigmoidal shape is an effect of the land type, soil structure and the resilience of the land type. Similarly, the pronounced peak of net present value is more the result of the model having perfect knowledge of the seasonal pattern. It is realistic to expect that the peak of the net present value curve would be relatively flatter, as the manager would be able implement other strategies not accounted for in the model, such as more regular adjustment of stocking rates. Large variations in NPV for each of the 20 year time series are hidden in the data and this helps to explain why landholders find it difficult to operate at the peak.

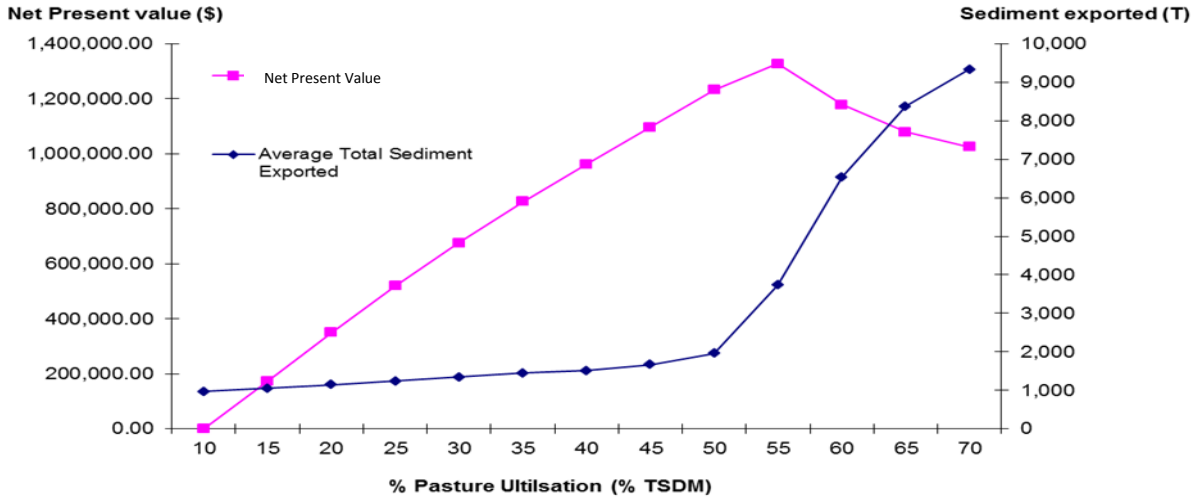


Figure 6.2 Brigalow blackbutt, Blackwater, A condition, 0 trees.

Table 6.1 Brigalow blackbutt, Blackwater, A condition, 0 trees. Net present value and sediment trade-offs.

Pasture utilisation (%TSDM)	15	20	25	30	35	40	45	50	55	60	65	70
Actual AEs for 5,000 ha	706	914	1,110	1,294	1,468	1,639	1,791	1,915	1,895	1,789	1,772	1,805
Net present value ($)	22,779	152,026	275,490	387,591	494,878	597,549	691,702	796,955	904,211	812,423	737,105	669,458
Ave total sediment exported (T)	1,055	1,146	1,240	1,342	1,445	1,511	1,666	1,958	3,730	6,536	8,377	9,336
$/T	21.58	132.66	1321.80	1096.34	1036.91	1572.00	604.87	360.92	60.52	-32.71	-40.92	-70.50

6.2.2 Brigalow blackbutt, A condition, with 3 m^2/ha tree basal area

As trees compete with grasses for nutrients and water, the impact of trees on the sediment run-off and the economic optimum point is significant. It is also important to consider the impact of leaf and organic matter from trees, which also contributes to the sediment exported. These Brigalow blackbutt results demonstrate the impact of trees: the optimal pasture utilisation has decreased from 55% pasture utilisation (Brigalow blackbutt with 0 tree basal area) to 45% pasture utilisation with a similarly large reduction in NPV. The effect of a tree basal area was similarly noted at the higher levels of pasture utilisation, with the net present value demonstrating a more bell-shaped curve.

For this combination of parameters, 45% pasture utilisation is the economic optimum point, and the net present value is $558,925. At this level the total sediment exported is 3,021 tonnes. There is a slight reduction ($1,878) in net present value from 45% to 50% utilisation and an increase of 830 tonnes of sediment exported. However from 55% utilisation to 60% utilisation there is a decrease in net present value of $208,936 and an increase of sediment of 3,361 tonnes. It is this particular type of scenario that can achieve reductions through extension and education. These results are depicted in Figure 6.3 and tabulated in Table 6.2.

The biophysical model GRASP has degradation and regeneration thresholds encompassed in the model (as described in Chapter Five). For Brigalow blackbutt, the degradation threshold is at 40% pasture utilisation. At 40% it can be observed that the peak of the net present value curve is approached and from this level of pasture utilisation the curve begins to plateau and then fall. This is a cumulative effect of the degradation threshold, large number of animals to be drought fed, and an unstable herd with high mortalities and low branding rates.

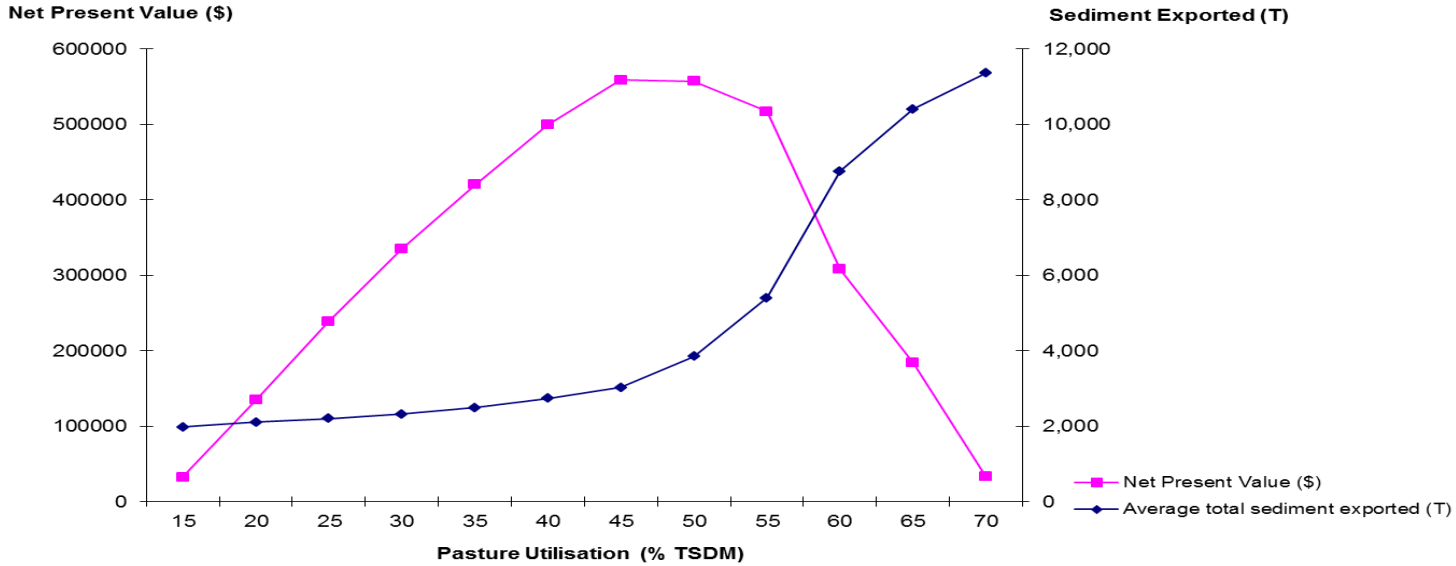


Figure 6.3 Brigalow blackbutt, Blackwater, A condition, 3 m^2/ha tree basal area

Table 6.2 Brigalow blackbutt, Blackwater, A condition, 3 m^2/ha tree basal area. Net present value and sediment trade-offs.

Pasture utilisation (%TSDM)	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**	**55**	**60**	**65**	**70**
Actual AEs for 5,000 ha	562	727	886	1,039	1,182	1,312	1,432	1,520	1,537	1,449	1,441	1,478
Net present value ($)	32,403	134,490	238,876	335,135	419,981	499,472	558,925	557,047	517,121	308,185	183,621	33,375
Ave total sediment exported (T)	1,972	2,106	2,206	2,323	2,485	2,732	3,021	3,851	5,394	8,755	10,403	11,356
$/T	16.43	761.67	1,043.21	824.49	523.54	321.01	206.08	-2.26	-25.87	-62.16	-75.60	-157.59

6.2.3 Brigalow blackbutt, B condition, 0 tree basal area

For Brigalow blackbutt in an initial B condition with 0 tree basal area, the highest net present value was achieved at 50% utilisation: $740,730 The tonnes of sediment exported were 4,015 on average over a 20-year period. This is a much higher net present value than land in initial A condition with a tree basal area of 3 m^2/ha, which was $557,047. This indicates that tree basal area has a greater impact on profit than land condition; however, there is not a substantial difference in sediment exported (which was 3,851 tonnes for land in A condition with average tree basal area), indicating that cheaper sediment reductions can be obtained by A condition land with trees. This result demonstrates the importance of understanding the influence of the different variables of the model.

It can also be observed that there is a significant increase from 4,015 tonnes to 7,383 tonnes in sediment when the pasture utilisation increases from 50% to 55%, but the profit only slightly decreases ($28,265 or 4%). This raises challenges in choosing the most effective policy mechanism for sediment reductions, as it presents increased risk both in profit and sediment exported. A grazier may choose to risk achieving an even (marginally) higher profit in good seasons. The downside risk is that the seasons may not be good, and land degradation will occur leading to a higher amount of sediment being exported.

It can also be noted that the most expensive reduction in pasture utilisation occurs at 20% TSDM, as the marginal increase in profit between 15% and 20% TSDM is significant, but the increase in sediment exported is relatively marginal. This results in a tonne of sediment reduced costing $1,337.78.

6.2.4 Brigalow blackbutt, B condition, 3 m^2/ha tree area

For both tree basal areas, the B start condition tree basal areas reflect the same findings as the A condition scenarios, except for each land condition the economic optimum peak has decreased by 10% pasture utilisation and the sediment exported has also slightly increased. In the scenario of B condition land with 3 m^2/ha tree area, the economic optimum pasture utilisation is 40% pasture utilisation; at this level of grazing pressure the profit obtained is $405,526 and 3,940 tonnes of sediment are exported.

In this scenario 40% is also the point from which sediment exported increases significantly, as well as being the threshold for degradation effects to take place in GRASP. It can also be noted that from 60% to 70% pasture utilisation, the sediment curve becomes much flatter; it can be interpreted that at 60% TSDM there is no topsoil profile left, and therefore sediment exported plateaus.

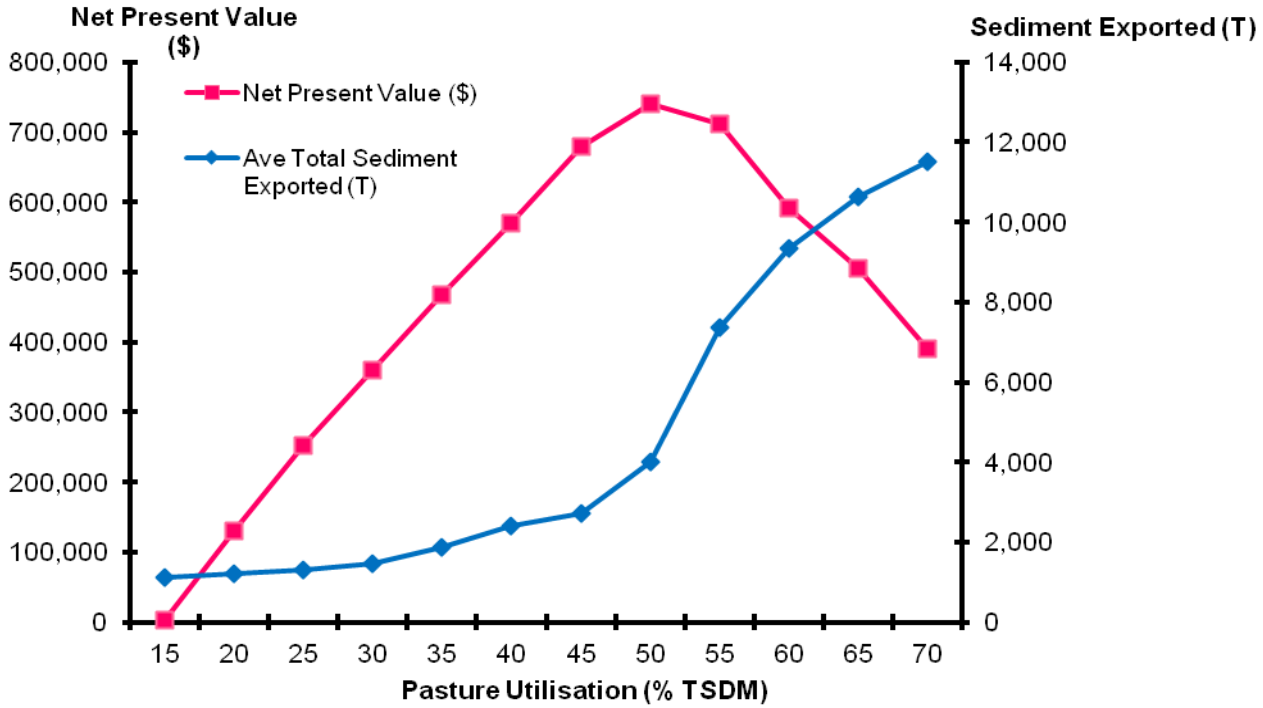


Figure 6.4 Brigalow blackbutt, Blackwater, B condition, 0 tree basal area

Table 6.3: Brigalow blackbutt, Blackwater, B condition, 0 tree basal area, Net present value and sediment trade-offs.

Pasture utilisation (%TSDM)	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**	**55**	**60**	**65**	**70**
Actual AEs for 5,000 ha	700	907	1,099	1,276	1,404	1,513	1,639	1,677	1,552	1,538	1,551	1,591
Net present value ($)	2,483	130,743	252,742	360,564	467,899	570,014	679,887	740,730	712,465	590,631	505,921	390,422
Ave total sediment exported (T)	1,109	1,205	1,316	1,454	1,867	2,414	2,735	4,015	7,383	9,357	10,649	11,511
$/T	2.24	1,337.78	1,103.44	781.67	259.87	186.64	342.55	47.51	-8.39	-61.73	-65.55	-134.02

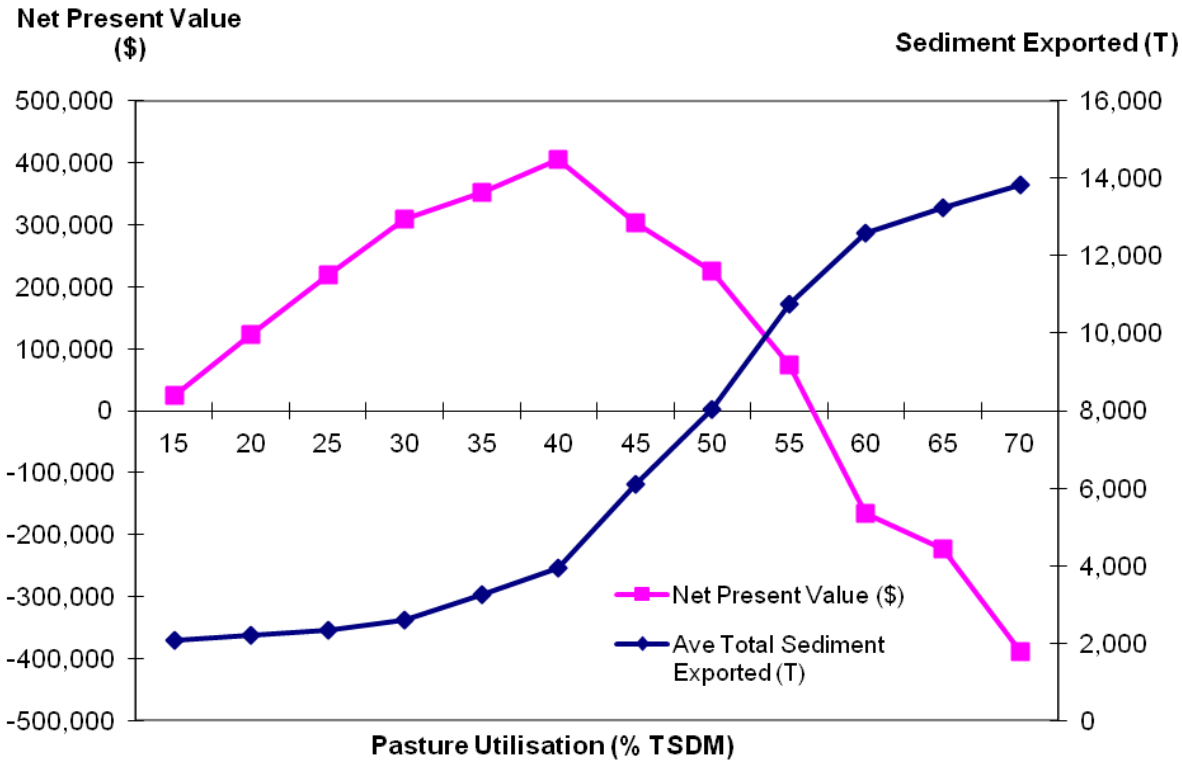


Figure 6.5 Brigalow blackbutt, Blackwater, B condition 3 m^2/ha tree basal area

Table 6.4 Brigalow blackbutt, Blackwater, B condition 3 m^2/ha tree basal area. Net present value and sediment trade-offs.

Pasture utilisation (%TSDM)	15	20	25	30	35	40	45	50	55	60	65	70
Actual AEs for 5,000 ha	558	722	878	1,022	1,135	1,237	1,255	1,251	1,208	1,213	1,257	1,307
Net present value ($)	23,855	123,907	218,316	308,791	352,076	405,526	303,916	225,548	74,446	-165,623	-222,173	-388,652
Ave total sediment exported (T)	2,066	2,215	2,345	2,605	3,261	3,940	6,110	8,025	10,767	12,602	13,246	13,823
$/T	11.55	671.07	727.28	348.23	66.01	78.71	-46.83	-40.91	-55.11	-130.86	-87.75	-288.29

6.2.5 Brigalow blackbutt, C condition, 0 trees

The economic optimal grazing utilisation for Brigalow blackbutt with an initial condition of C is 45% TSDM (Figure 6.6); this is 5% less than when the initial condition is B (Figure 6.4). At 45% pasture utilisation, the profit is $368,736, and 10,398 tonnes of sediment are exported (Figure 6.6). It can be observed that at 30% and 35% pasture utilisation, the cost per tonne of sediment reduced is the cheapest and therefore an incentives program is the most effective mechanism for sediment reductions at this pasture utilisation rate.

Figure 6.6 shows that the net present value curve does not follow a smooth distribution, unlike in the B and A condition scenarios; this can be attributed to the poor condition of the land type, resulting in an unstable system and more pronounced impacts of the land type resilience or degradation and regeneration.

6.2.6 Brigalow blackbutt, C condition, 3 m^2/ha tree basal area

The economic optimum for Brigalow blackbutt with a start condition of C and with 3 m^2/ha tree basal area was at 25% TSDM; this is the result of the combination of both a poor start condition and tree competition. The sediment exported at this economic optimum point is 4,539 tonnes, which is 56% more sediment than at 25% TSDM in the Brigalow blackbutt in C condition with no trees scenario. These results highlight the importance of the different interactions that occur in the biophysical environment and the cumulative effects of poor land condition and trees.

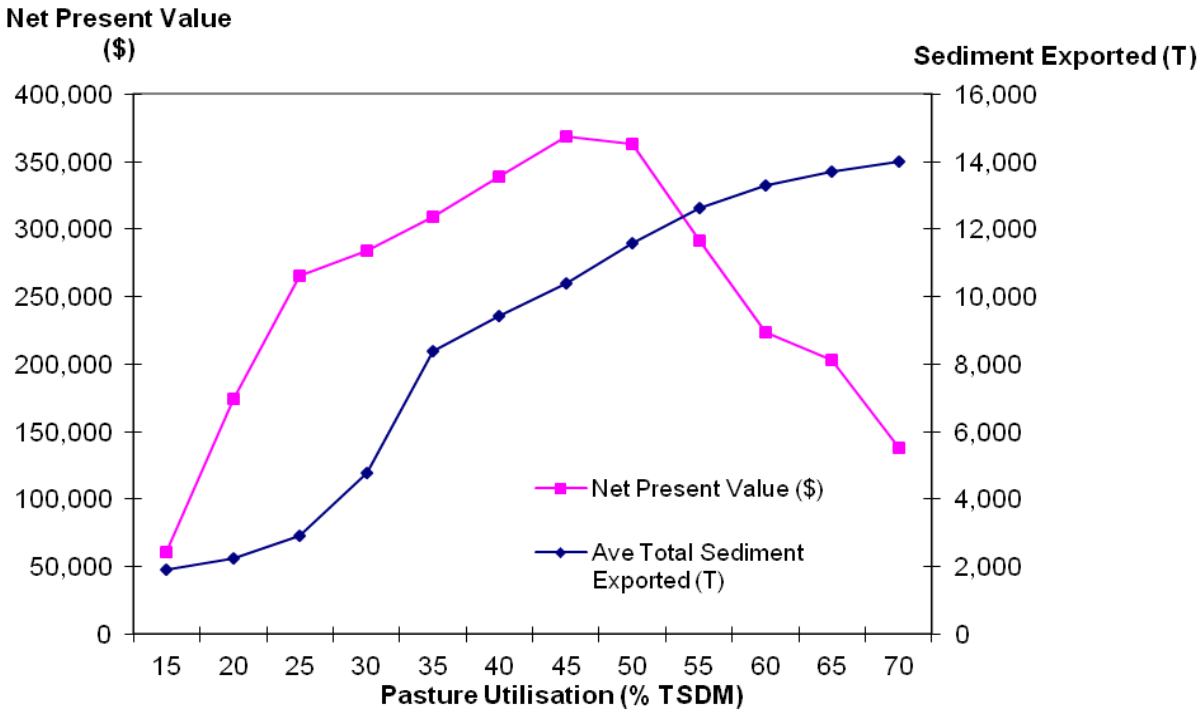


Figure 6.6 Brigalow blackbutt, Blackwater, C condition, 0 tree basal area

Table 6.5 Brigalow blackbutt, Blackwater, C condition, 0 tree basal area. Net present value and sediment trade-offs.

Pasture utilisation (%TSDM)	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**	**55**	**60**	**65**	**70**
Actual AEs for 5,000 ha	664	846	1,000	1,048	976	1,047	1,126	1,173	1,217	1,284	1,354	1,425
Net present value ($)	60,800	174,074	265,813	283,675	309,006	339,332	368,736	362,720	291,547	223,812	203,576	138,486
Ave total sediment exported (T)	1,905	2,263	2,918	4,771	8,369	9,440	10,398	11,592	12,644	13,309	13,721	14,020
$/T	31.92	316.44	140.08	9.64	7.04	28.31	30.70	-5.04	-67.68	-101.86	-49.13	-217.47

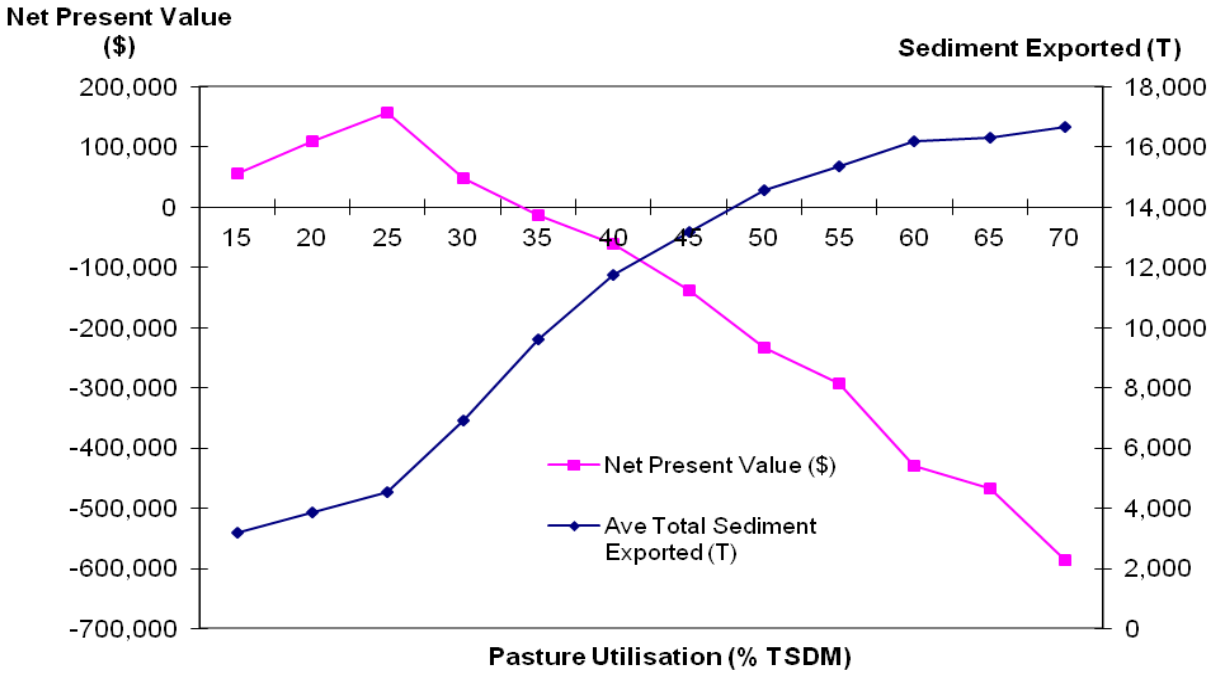


Figure 6.7 Brigalow blackbutt, Blackwater, C condition, 3 m^2/ha tree basal area

Table 6.6 Brigalow blackbutt, Blackwater, C condition, 3 m^2/ha tree basal area. Net present value and sediment trade-offs.

Pasture utilisation (%TSDM)	15	20	25	30	35	40	45	50	55	60	65	70
Actual AEs for 5,000 ha	527	670	795	825	834	868	908	941	980	1,022	1,083	1,138
Net present value ($)	54,989	109,396	156,224	47,849	-14,151	-62,022	-138,641	-233,558	-293,614	-429,070	-466,716	-586,195
Ave total sediment exported (T)	3,193	3,846	4,539	6,916	9,601	11,747	13,165	14,567	15,361	16,169	16,311	16,663
$/T	17.22	83.37	67.59	-45.58	-23.10	-22.30	-54.06	-67.66	-75.71	-167.60	-265.89	-338.95

6.2.7 Goldfields, A condition, 0 tree basal area

For A condition land with a 0 tree basal area (Figure 6.8), the economic optimum pasture utilisation rate is achieved at 25% TSDM where the net present value is $354,141, and 7,777 tonnes of sediment are exported. This is a significantly lower net present value and optimal pasture utilisation rate than on the Brigalow blackbutt land type in A condition with 0 trees. It can also be noted that substantial increase in sediment exported has occurred.

The lower productivity land types, as exemplified by the Goldfields land type, followed the same trend in the sediment exported curve, where initially large amounts of sediment were exported and then a point was reached where the curve flattens out. This can be explained by the soil characteristics of these land types: they initially erode very quickly, but then reach a point where there is limited topsoil left to erode further. When combined with B or C condition land, the curve is more pronounced.

6.2.8 Goldfields, A condition, 3 m^2/ha tree basal area

The A condition scenario with trees has a negative net present value for all pasture utilisation rates apart from 15% TSDM. This is due again to the low productivity of the land and the impact of trees. This was a common result for lower productivity land types, which indicates that it is not sustainable to operate grazing enterprises on these particular land types. However, it must be remembered that the assumption of one property consisting of only one land type in poor condition is not realistic. This information is useful to graziers who do have large areas of low productivity land types with treed areas and are grazing them; extension and education as to how to manage the enterprise with that land type may be an appropriate management strategy. For example, instead of grazing the land type all year, cattle may be grazed through for a couple of weeks before moving to a different land type.

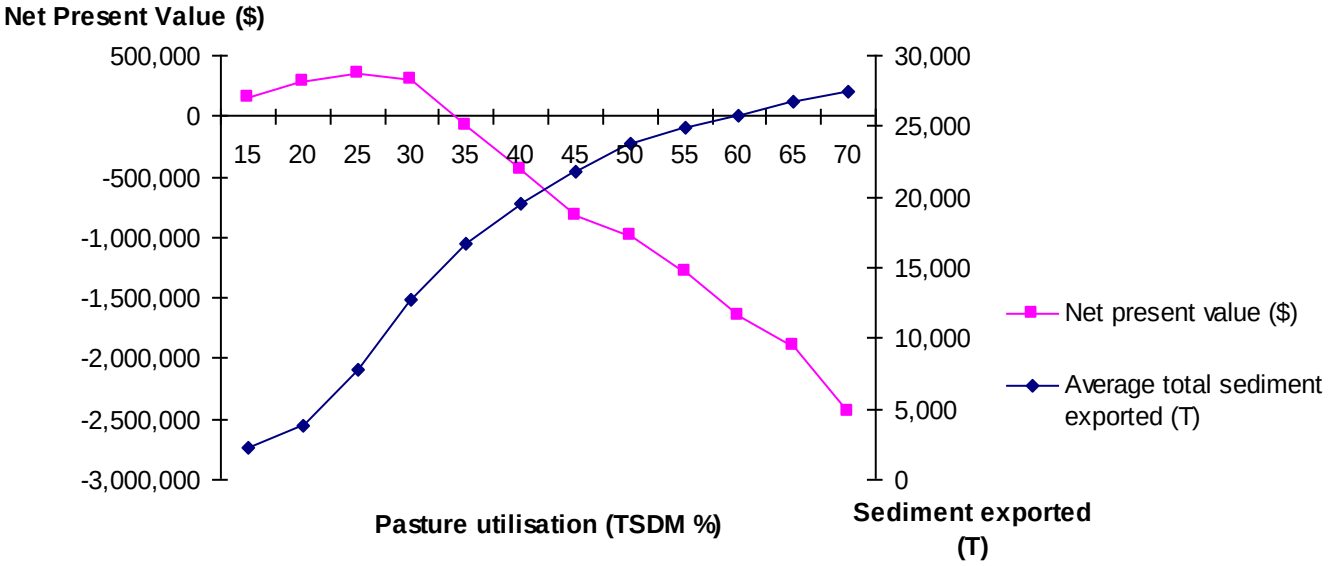


Figure 6.8 Goldfields,Charters Towers, A condition, 0 tree basal area

Table 6.7 Goldfields,Charters Towers, A condition, 0 tree basal area,Net present value and sediment trade-offs.

Pasture utilisation (% TSDM)	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**	**55**	**60**	**65**	**70**
Actual AEs for 5,000 ha	1,181	1,388	1,455	1,462	1,499	1,575	1,647	1,717	1,813	1,923	2,029	2,128
Net present value ($)	153,426	285,215	354,141	297,940	-80,267	-445,459	-828,560	-990,381	-1,285,572	-1,643,521	-1,894,407	-2,435,896
Ave total sediment (T)	2,247	3,862	7,777	12,802	16,674	19,536	21,842	23,710	24,954	25,808	26,712	27,397
$/T	68	82	18	-11	-98	-128	-166	-87	-237	-419	-278	-791

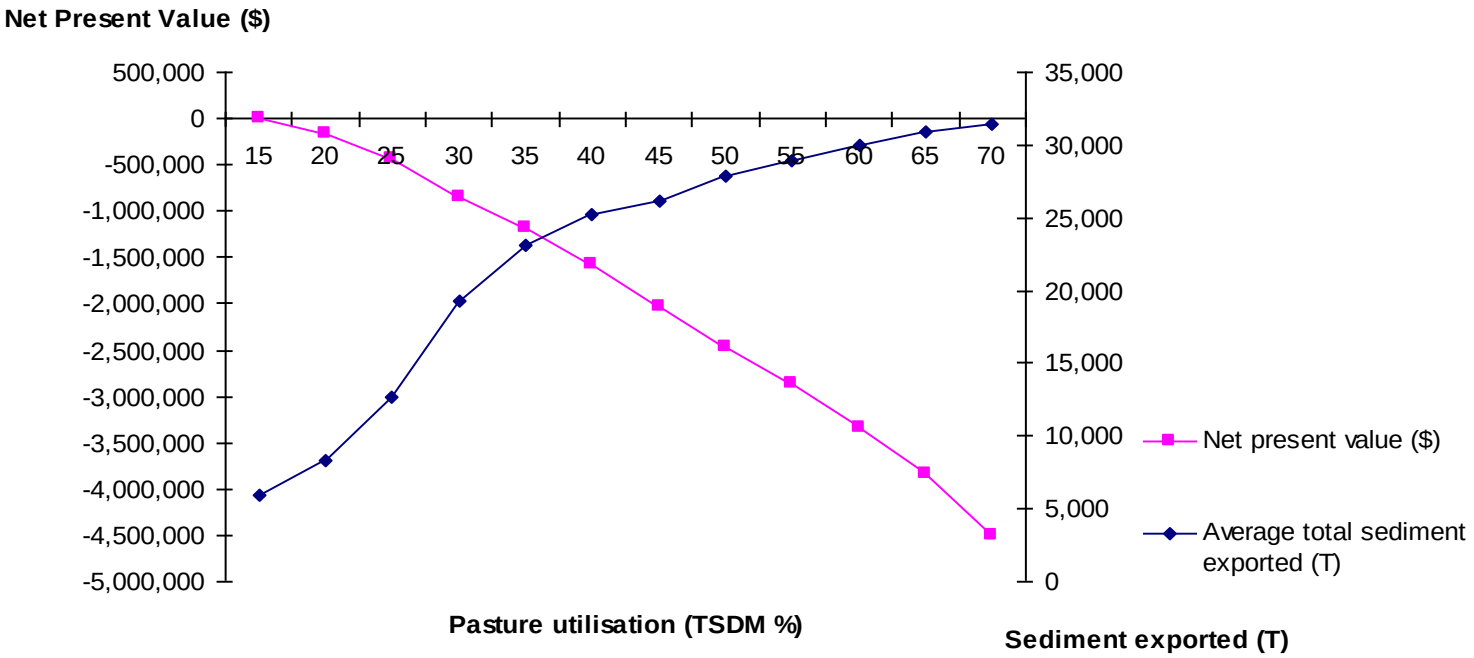


Figure 6.9 Goldfields, Charters Towers, A condition, 3 m^2/ha tree basal area.

Table 6.8 Goldfields, Charters Towers, A condition, 3 m^2/ha tree basal area. Net present value and sediment trade-offs.

Pasture utilisation (% TSDM)	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**	**55**	**60**	**65**	**70**
Actual AEs for 5,000 ha	732	874	940	907	926	976	1,055	1,107	1,161	1,225	1,282	1,346
Net present value ($)	3,315	-173,025	-439,079	-856,711	-1,180,129	-1,578,011	-2,041,281	-2,465,344	-2,859,285	-3,342,675	-3,843,146	-4,498,485
Ave total sediment (T)	6,001	8,288	12,637	19,222	23,171	25,212	26,167	27,888	28,979	30,025	30,887	31,441
$/T	1	-77	-61	-63	-82	-195	-485	-246	-361	-462	-581	-1,181

6.2.9 Goldfields in B condition, 0 trees and 3 m^2/ha tree basal area

For the B condition scenario with 0 trees, the economic optimum pasture utilisation rate is 25% (Figure 6.9). For B condition with a tree basal area of 3 m^2/ha, at 25% pasture utilisation the net present value is -$570,263, and 21,998 tonnes of sediment is exported. There is an additional 9,474 tonnes of sediment exported over the 20-year period compared to the Goldfields in B condition with no tree basal area scenario. The results indicate that it is not viable to have a whole property of Gold fields in B condition with trees – although it might be more viable when it is grazed in conjunction with other land types.

This scenario demonstrates that tree basal area is more important than starting land condition on the economic optimum point for pasture utilisation. As land condition improves from C to B with 0 tree basal area, the optimal pasture utilisation rate increases from 15% TSDM in C condition, where the net present value is $81,693 and 15,358 tonnes of sediment is exported, (Figure 6.12) to 25% TSDM in B condition with a net present value of $205,958 and 12,524 tonnes of sediment exported (Figure 6.10).

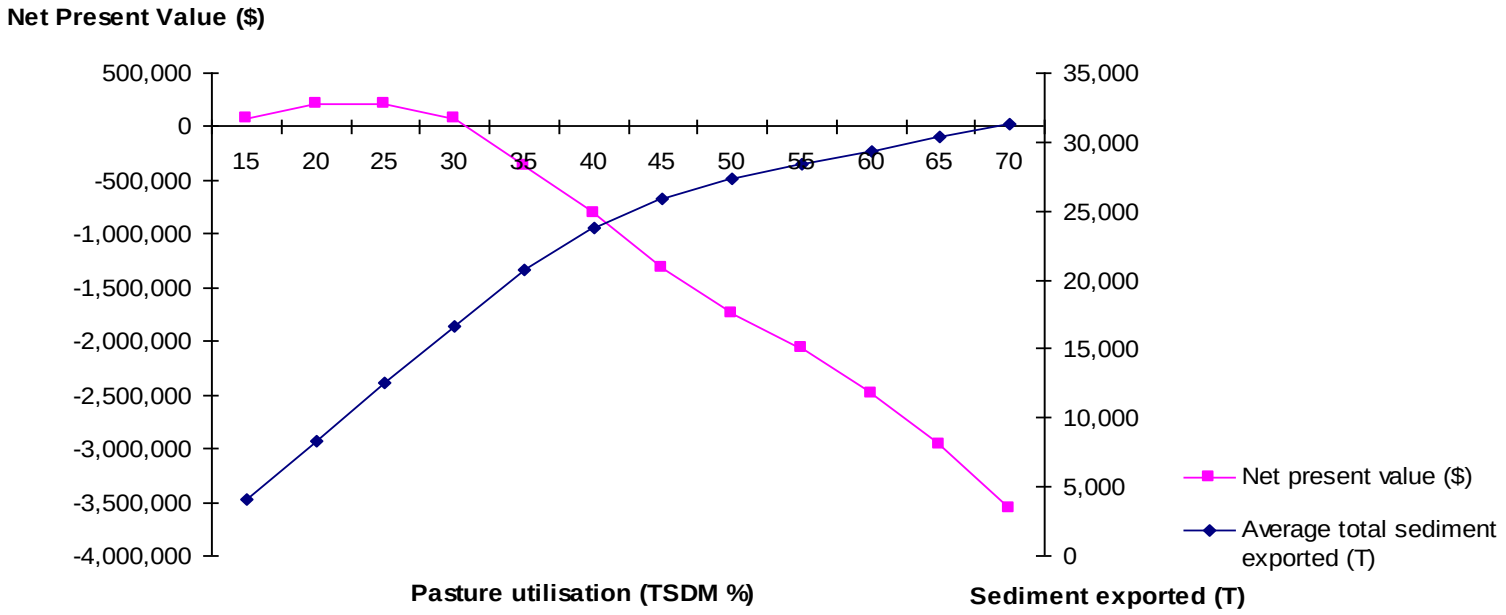


Figure 6.10 Goldfields, Charters Towers, B condition, 0 tree basal area

Table 6.9 Goldfields, Charters Towers, B condition, 0 tree basal area. Net present value and sediment trade-offs.

Pasture utilisation (% TSDM)	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**	**55**	**60**	**65**	**70**
Actual AEs for 5,000 ha	1,020	1,085	1,159	1,221	1,253	1,316	1,395	1,480	1,570	1,664	1,739	1,832
Net present value ($)	76,617	205,943	205,958	67,789	-366,795	-805,318	-1,322,347	-1,742,847	-2,058,271	-2,487,369	-2,967,895	-3,555,111
Ave total sediment (T)	4,102	8,385	12,524	16,679	20,770	23,799	25,938	27,297	28,363	29,280	30,438	31,326
$/T	19	30	0	-33	-106	-145	-242	-309	-296	-468	-415	-661

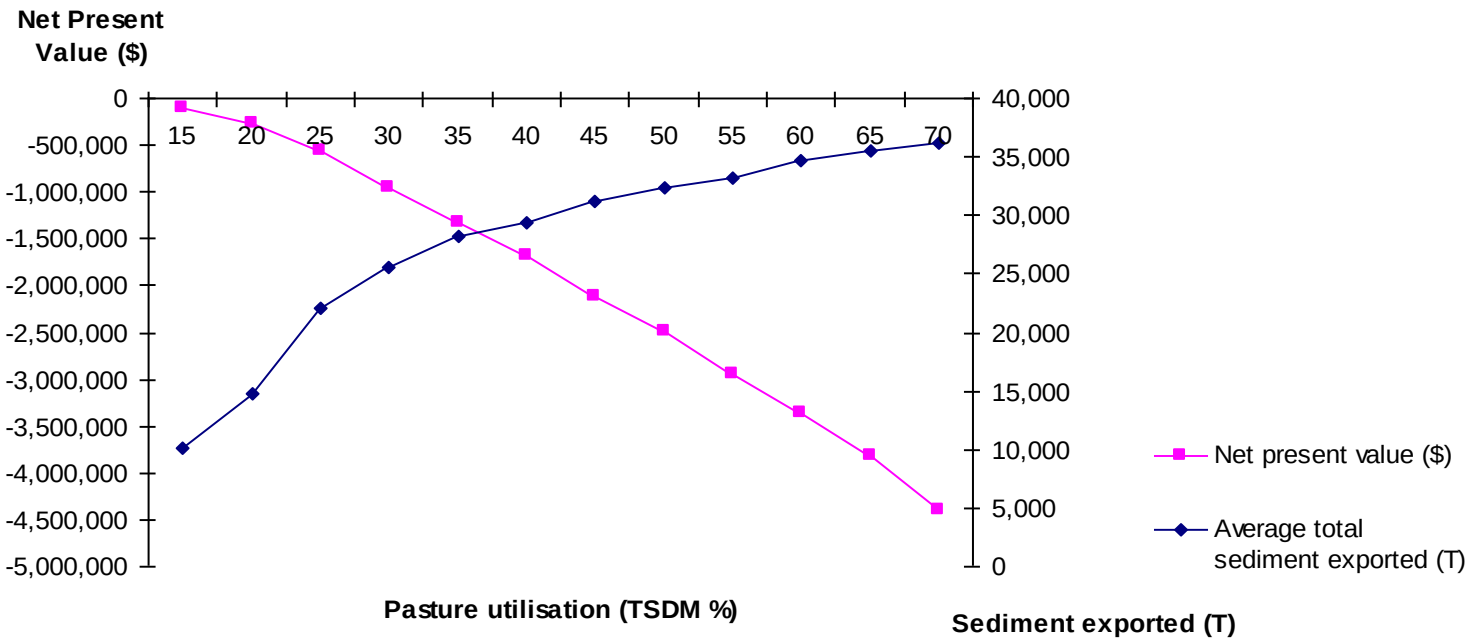


Figure 6.11 Goldfields, Charters Towers, B condition, 3 m^2/ha tree basal area

Table 6.10 Goldfields, Charters Towers, B condition, 3 m^2/ha tree basal area Net present value and sediment trade-offs.

Pasture utilisation (% TSDM)	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**	**55**	**60**	**65**	**70**
Actual AEs for 5,000 ha	610	680	669	709	757	822	863	921	978	1,008	1,066	1,116
Net present value ($)	-93,881	-266,397	-570,263	-961,902	-1,338,152	-1,675,291	-2,115,906	-2,494,846	-2,955,845	-3,353,871	-3,807,132	-4,399,970
Ave total sediment (T)	10,151	14,835	21,998	25,555	28,215	29,315	31,255	32,382	33,202	34,721	35,467	36,244
$/T	-9	-37	-42	-110	-141	-307	-227	-336	-562	-262	-608	-762

6.2.10 Goldfields in C condition, 0 trees and 3 m^2/ha tree basal area

In the scenario for C condition with 0 trees, the economical optimal is at 15% TSDM, where an income of $81,693 is achieved and 15,358 tonnes of sediment is exported. This demonstrates the low productivity of gold fields in poor condition, and soil characteristics for highly erosive soils. This highlights the importance of targeting investments where large reductions can occur at low opportunity cost to the landholder.

In the C condition scenario with trees, net present values are negative across all pasture utilisation rates. Also, the sediment curve – even at low pasture utilisation rates – is relatively flat; this is attributed to the decline of the land condition and soil to C condition. It can be noted that for 25% TSDM 30,440 tonnes of sediment is exported; for the same pasture utilisation (25% TSDM) in B and A conditions with trees, the amounts are 21,998 tonnes and 12,637 tonnes respectively. This highlights the importance of land regeneration on poor productivity land types.

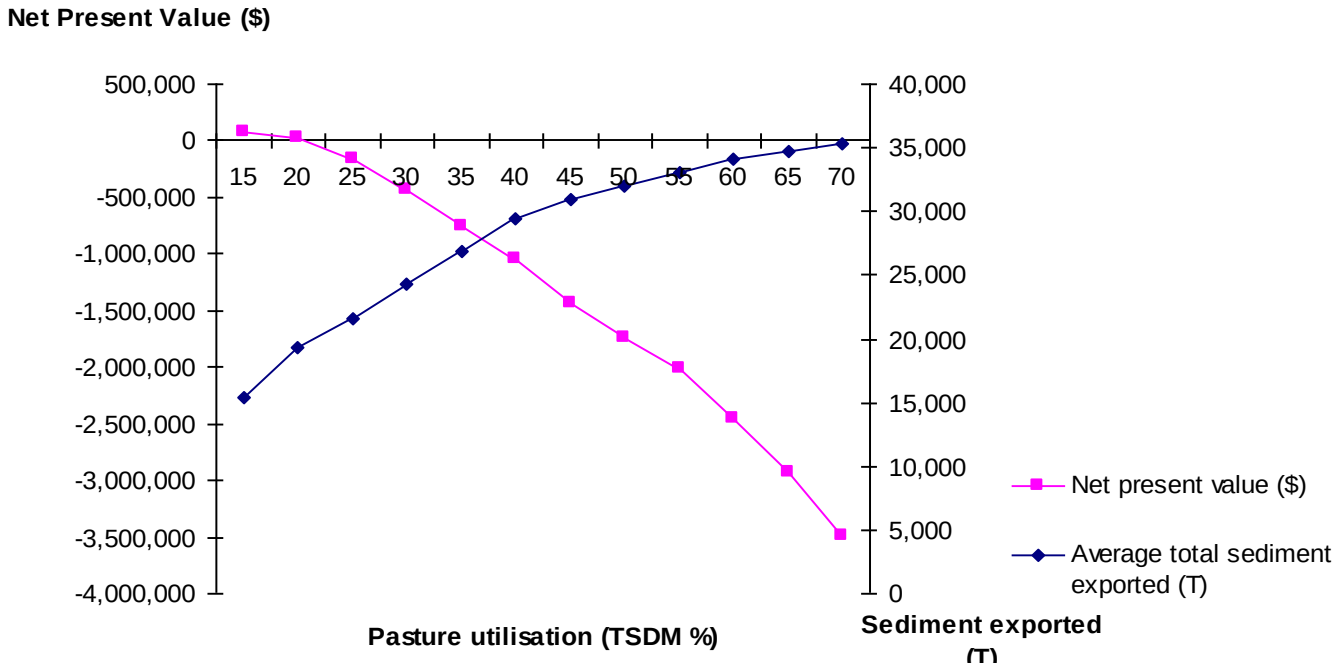


Figure 6.12 Goldfields, C condition, 0 tree basal area

Table 6.11 Goldfields, Charters Towers, 0 tree basal area. Net present value and sediment trade-offs.

Pasture utilisation (% TSDM)	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**	**55**	**60**	**65**	**70**
Actual AEs for 5,000 ha	600	692	813	895	971	1,033	1,117	1,199	1,280	1,357	1,435	1,512
Net present value ($)	81,693	20,172	-155,783	-432,525	-753,650	-1,051,633	-1,428,518	-1,741,390	-2,017,823	-2,458,369	-2,938,422	-3,486,411
Ave total sediment (T)	15,358	19,333	21,547	24,356	26,907	29,478	30,958	32,047	33,055	34,044	34,762	35,381
$/T	5	-15	-79	-99	-126	-116	-255	-287	-274	-445	-669	-885

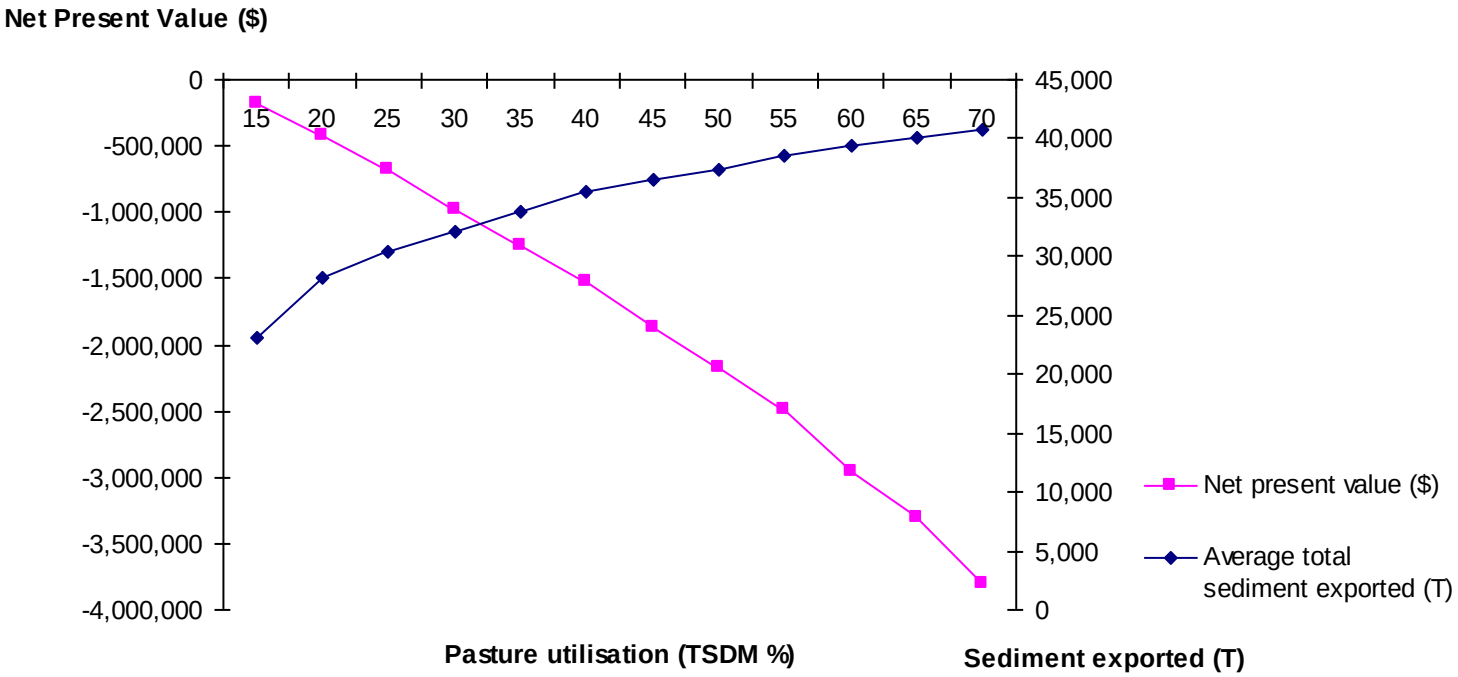


Figure 6.13 Goldfields, Charters Towers, C condition, 3 m^2/ha tree basal area

Table 6.12 Goldfields, Charters Towers, C condition, 3 m^2/ha tree basal area Net present value and sediment trade-offs.

Pasture utilisation (% TSDM)	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**	**55**	**60**	**65**	**70**
Actual AEs for 5,000 ha	367	403	464	523	571	612	664	712	756	799	846	888
Net present value ($)	-186,151	-418,186	-682,764	-978,820	-1,246,945	-1,521,958	-1,877,852	-2,175,705	-2,487,645	-2,964,791	-3,308,169	-3,798,971
Ave total sediment (T)	23,151	28,213	30,440	32,173	33,798	35,436	36,469	37,422	38,494	39,391	40,114	40,830
$/T	-8	-46	-119	-171	-165	-168	-344	-313	-291	-532	-475	-685

The impact of land condition and trees has been noted in both the Brigalow blackbutt and the Goldfields land types, with profit decreasing as land condition declined and with the presence of trees. This trend was observed for all land types, with the marginal differences depending on the productivity of the land type and climate.

The presence of trees impacted more heavily on profit than the declining land condition did, and declining land condition and trees also resulted in substantially more sediment exported. Modelling the impact of no trees on land types such as spotted gum and narrow-leaved ironbark is unrealistic, because these land types are seldom cleared for grazing due to their slope and inherit low fertility characteristics; therefore, there is little trial data to populate these land types and an unrealistic result occurs. When these land types are modelled for a cleared scenario, the net present value curve does not follow the bell-shaped curve; instead, it trends up and then flattens out (Figure 6.14). This will be further discussed as a limitation of the modelling.

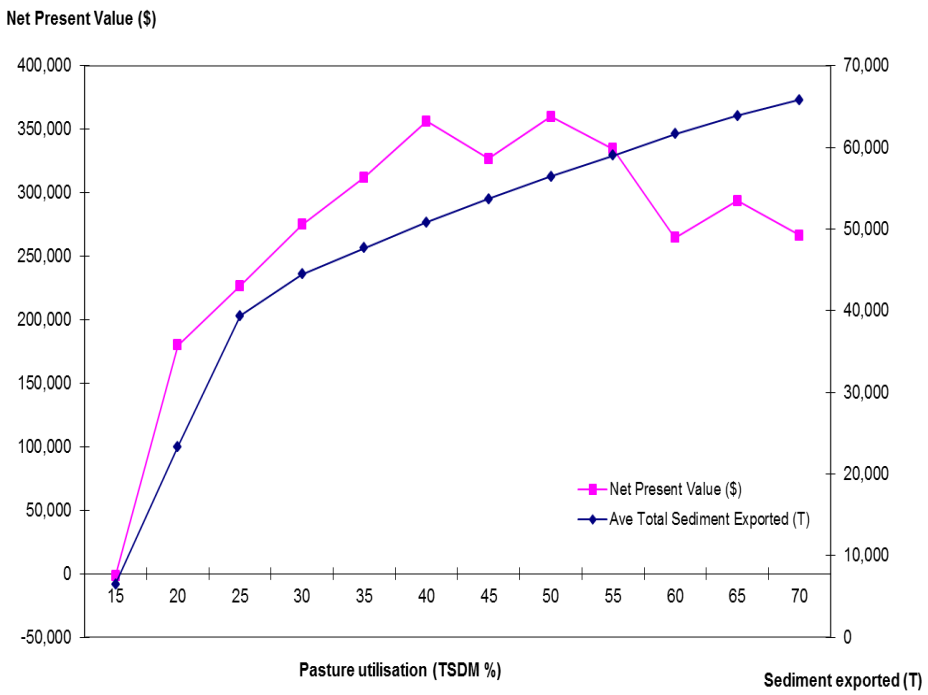


Figure 6.14 Narrow-leaved ironbark in A condition, 0 tree basal area.

The impact of land condition and the marginal difference between the profit obtained from A to B varied between land types, but all land types shared the

trend of profit declining with declining land condition. For example, when land condition was compared for 30% pasture utilisation in treed areas for Brigalow blackbutt, the results demonstrated the gradual decline from A to B in profit (from $335,135 down to $308,791) and a 12% increase in sediment. However, when land condition declined further to C condition, profit declined to just $47,849, and sediment increased by 2.65 times to 6,916 average tonnes exported.

Table 6.13 Brigalow blackbutt, 30% pasture utilisation, effect of land condition

Brigalow blackbutt			
Start condition 3 m^2/ha tree basal area	**A**	**B**	**C**
Pasture utilisation (%TSDM)	**30**	**30**	**30**
Actual AEs for 5,000 ha	1,039	1,022	825
Net present value ($)	335,135	308,791	47,849
Ave total sediment exported (T)	2,323	2,605	6,916
Net present value ($)/ Ave tonnes of sediment exported (T)	824.49	348.23	-45.58

6.3 Confidence intervals

To understand the confidence intervals surrounding the initial results, the results were bootstrapped. This is a process where a re-sample from the initial NPV results for each 20 year block was taken 10,000 times. It allows the variance among the results to be observed and the reliability of the results to be further understood. The net present value and the sediment loads were both tested for the confidence intervals. For the purposes of this example (Figure 6.15) Goldfields in B condition with 0 tree basal area was selected, but the same trends were observed across all the land types.

The results of the bootstrap illustrate the variance that occurs, particularly in the higher pasture utilisation rates, for both sediment and net present value. It can be observed that when TSDM is past 40% the confidence intervals increase their distance from the mean value. This is attributed to the large ‘boom and bust’ years (years that have high rainfall and low rainfall) which results in high or low economic optimal pasture utilisation rate. It also can be attributed to the fact that at such high levels of grazing pressure, limited trial data exist to populate GRASP, the biophysical model. Therefore, the confidence intervals demonstrate

that past 40% TSDM the results must be considered with caution. Although climate was implicitly accounted for price was not varied and therefore the confidence intervals will be understated. The confidence intervals of the net present value also show that it may be possible to achieve some level of profits – or the perception that it is possible.

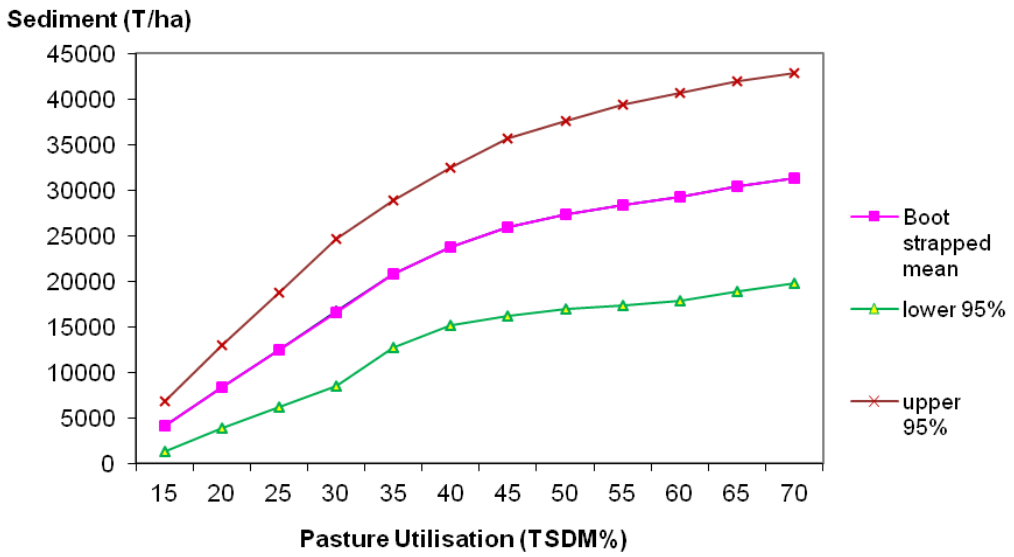


Figure 6.15 Goldfields, B condition, 0 tree basal area confidence intervals for sediment

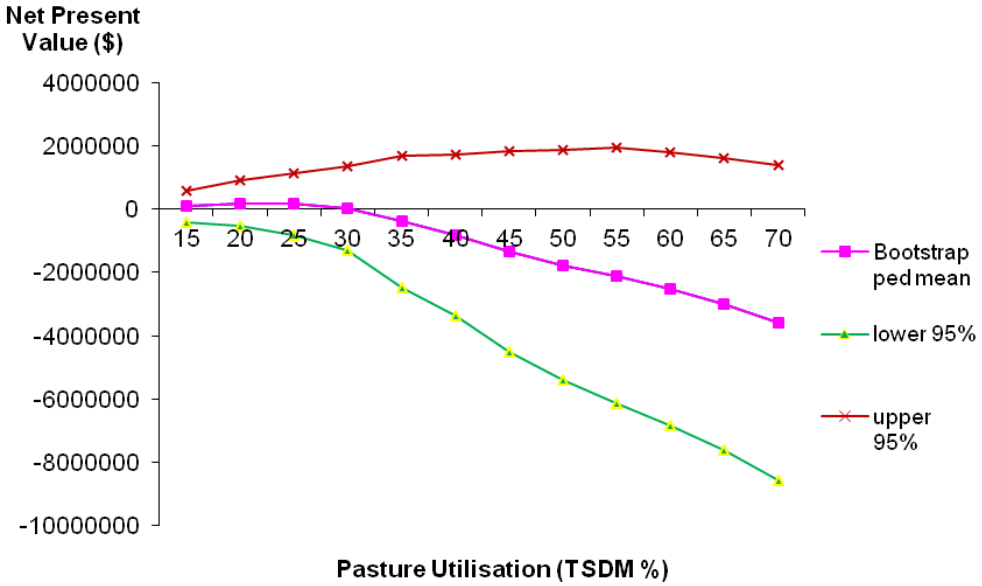


Figure 6.16 Goldfields, B condition, 0 tree basal area confidence intervals for net present values

6.4 Summary of results across remaining land types

The in-depth results reported in Section 6.2 provide the comparison between two of the eight land types that were considered. The results highlight the impact of land type, land condition and the impact of a tree basal area. This section provides a summary of the remaining six land types which were analysed. The land types followed a similar trend which reflected the inherent productivity, climate location, land condition and impact of trees. Table 6.14 provides a summary of the dollars per tonne of sediment for the cleared scenario, with the three different land conditions. A full summary of all land type results are presented in Appendix C.

Table 6.14 Summary of $ per tonne for the cleared scenario across A,B and C land condition.

Pasture utilisation (% TSDM)	15	20	25	30	35	40	45	50	55	60	65	70
A condition												
Coolibah floodplains	338	851	258	105	-134	-76	-279	-176	-128	-294	-522	-916
Narrow leaved ironbark woodlands	1	60	99	173	26	-331	-407	-101	-76	-46	-141	-206
Narrow leaved ironbark ranges	-2	-10	-29	-64	-81	-104	-239	-298	-426	-1,444	-289	-2,077
Silver leaved ironbark	158	82	51	48	-10	-23	-29	-73	-67	-68	-128	-69
Silver-leaved ironbark - duplex	41	58	12	13	-10	-21	-18	-52	-98	-139	-78	-143

Spotted Gum	18	12	17	8	5	19	-23	10	-27	29	47	75
B condition												
Coolibah floodplains	253	268	23	-106	-125	-147	-408	-286	-178	-578	-971	-801
Narrow leaved ironbark woodlands	-1	6	4	1	1	6	-23	-16	-22	-26	-33	-47
Narrow leaved ironbark ranges	8	44	318	56	90	-304	-221	-660	-108	-233	146	-163
Silver leaved ironbark	55	41	27	12	-19	-24	-69	-99	-142	-160	-133	-283
Silver-leaved ironbark - duplex	-2	-29	-41	-65	-48	-135	-177	-149	-241	-453	-207	-615
Spotted Gum	10	10	11	5	-8	-5	0	-7	-13	-16	-5	-37
C condition												
Coolibah floodplains	80	-46	-84	-126	-142	-192	-332	-469	-421	-803	-1,253	-1,361
Narrow leaved ironbark woodlands	2	3	2	2	1	0	-1	-2	-2	-4	-4	-6
Narrow leaved ironbark ranges	7	8	9	9	9	10	11	11	11	10	10	10
Silver leaved ironbark	55	41	27	12	-19	-24	-69	-99	-142	-160	-133	-283
Silver-leaved ironbark - duplex	3	0	-16	-24	-56	-55	-116	-104	-130	-203	-169	-307
Spotted Gum	10	10	11	5	-8	-5	0	-7	-13	-16	-5	-37

6.5 Comparison and optimisation of different land types

To understand the impact on productivity and to understand how much of an improvement could be achieved by targeting incentive payments to different land types, the results were combined with spatial data and put into an optimisation framework, based on the Fitzroy Basin Association (FBA) having $2 million[5] to allocate in incentives. For this exercise spotted gum, silver leaved ironbark and goldfields were not included. This was due to the factors of high tree cover (spotted gum, silver leaved ironbark) or low amount of the land type in the Fitzroy basin (Goldfields).

Consequently, the function for targeted funds is based on:

$$Max\left(\frac{\Delta J}{X}\right) = \frac{\sum\left(CC_{Lcb} * \alpha * E\right)}{\sum\left(Cost_{CCLcb} * \alpha\right)} \qquad \text{(Equation 11)}$$

where:

[5] $2 million dollars was a hypothetical amount selected for the analysis but was made in consideration to the Reef Rescue package

J is total sediment exported

X is a given quantity of funds

CC is condition change where

L is land with a particular land type (*b*) and condition (*c*)

α is the area (ha) that is proposed to be improved

E is the change in sediment exports associated with the relevant condition change

Cost is the opportunity cost of improving the condition of each land unit to achieve the potential reductions in sediment emissions.

Given the variety of possible land management actions and areas in the catchment where they could be applied, the optimisation problem is to select the most appropriate actions and land units that will achieve the highest reductions at least cost.

Subsequent research using satellite imagery (Abbott 2008; Karfs, Abbott et al. 2009), has seen the generation of modelled land condition imagery for the FBA region (Figure 6.17). This imagery is derived from long time series of ground cover images (Scarth *et al.* 2006) and allows sub-paddock mapping of land condition on grazing land where tree cover does not obscure ground cover estimates. The imagery is experimental – similar imagery has been successfully ground-truthed in the neighbouring Burdekin region but has not been tested rigorously in the FBA region. In addition, it does not model impacts of weeds on land condition. It does, however, provide a unique opportunity to make some objective estimate of coverage of different land condition classes at catchment scales.

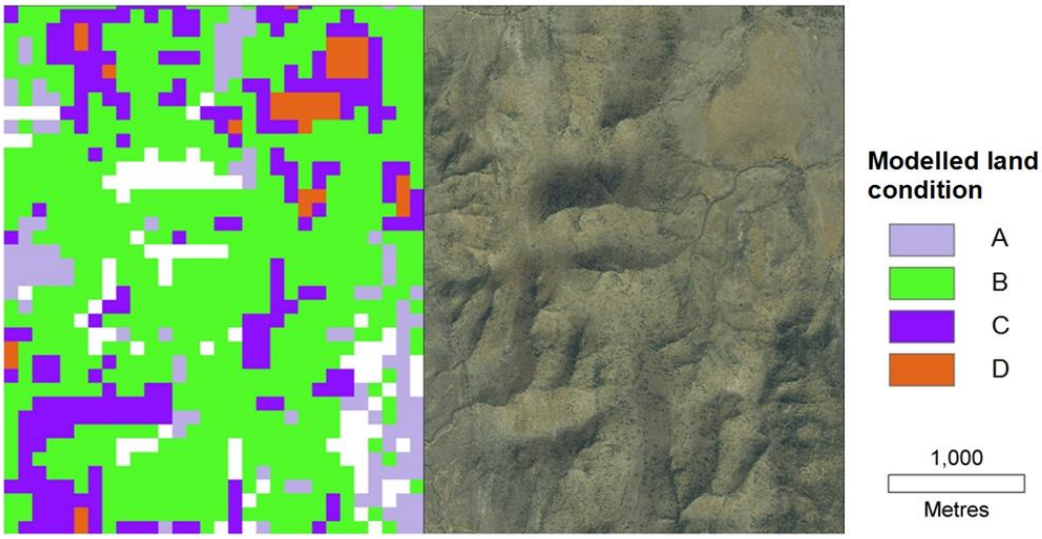


Figure 6.17 Satellite land condition map

The modelled current land condition image was overlaid with land type mapping to quantify the proportion of each of the five selected land types in each land condition class (Table 6.15). These data were then used to inform the optimisation analyses.

Table 6.15 Summary area (ha), percentage of catchment and modelled locality for selected land types in the Fitzroy Basin

Land type	Total area (ha)	% of FBA	Modelling locality
Brigalow blackbutt (*Acacia harpophylla* / *Eucalyptus cambageana*)	379,400	2.42	Blackwater (23.58°S 148.88°E)
Coolibah (*E. microtheca*) floodplains	3,844	2.45	Mantuan Downs (24.38°S 148.23°E)
Narrow-leaved ironbark (*E. crebra*) woodlands	11,399	7.27	Blackwater (23.58°S 148.88°E)
Silver-leaved ironbark (*E. melanophloia*) on duplex soils	6,369	4.06	Springvale (23.12°S 148.03°E)
Narrow-leaved ironbark (*E. crebra*) on ranges	6,943	4.43	Balaclava Mountains (21.60°S 148.97°E)

Areas of FBA land types in A, B and C condition were estimated according to Abbott (2008), who modelled land condition mapping for the FBA region (Table 6.16). Null areas are those not assessed due to either high levels of tree cover that limit satellite assessment of ground cover, or land uses not related to grazing (e.g.

mining or residential areas). These areas were not included in the optimisation analysis.

Table 6.16 Estimated percentage of land types in A, B and C condition in the Fitzroy Basin

	A	B	C	Null
Brigalow blackbutt	7.5	46.1	24.8	21.6
Coolibah floodplains	12.5	33.2	18.9	35.4
Narrow-leaved ironbark woodland	16.9	26.8	9.0	47.3
Narrow-leaved ironbark ranges	24.8	30.5	8.6	36.1
Silver-leaved ironbark on duplex	15.3	35.9	11.4	37.4

Utilising the information from the bioeconomic model and the bare-ground mapping, a simple linear programming (LP) model was developed in Solver (an Excel add-in). This allows the *a* vector to be determined. LP optimises a given variable subject to a number of given constraints (Cacho 1997; Lant *et al.* 2005). The LP is a simplification of an ideal optimisation model in that is did not account for measured benefits from the improvements. It was assumed that the maximum amount of money the FBA had to spend on incentives for landholders to reduce grazing pressure would be $2 million. The linear program design assumed that landholders were profit maximisers and therefore operating at the economic optimum point of pasture utilisation, having perfect knowledge regarding pasture utilisation. The targeted reduction in pasture utilisation was assumed to be 10 per cent. The linear program was designed using the constraints of inputs of dollars per tonne of sediment reduced at each pasture utilisation level, area of the catchment with that particular land type and land condition, and the maximum total dollars for sediment reductions ($2 million). The LP model solved the optimisation to find the cheapest way to reduce the maximum sediment exported, and to continue to choose each next cheapest option until the $2 million limit was reached.

The predictions of costs from the bioeconomic modelling is shown in Table 6.17, and the summary of cost information and bare-ground mapping in Table 6.18.

For each land type and condition, the costs of reducing pasture utilisation from the optimal production point are shown. For instance, Brigalow blackbutt country in A condition has optimal production at 55% pasture utilisation, and the costs per tonne of sediment of reducing utilisation by 10% is $421 ($61 for the first 5% reduction and $360.1 for the second 5% reduction). The results of the bioeconomic modelling demonstrate that optimal levels of pasture utilisation vary over land type and condition, and that costs of reducing sediment emissions by reducing grazing pressure vary between $4 and $421 per tonne.

Table 6.17 Predicted costs for 10% reduction in pasture utilisation for modelled land types, land condition and pasture utilisation

		Dollars per tonne of sediment reduced at each pasture utilisation (% TSDM)								
Land type	**Land condition**	**55**	**50**	**45**	**40**	**35**	**30**	**25**	**20**	**15**
Brigalow blackbutt	A	61	361							
Brigalow blackbutt	B		48	343						
Brigalow blackbutt	C			31	28					
Narrow-leaved ironbark woodlands	A					2	4			
Narrow-leaved ironbark woodlands	B							5	16	
Narrow-leaved ironbark woodlands	C								3	3
Coolibah floodplains	A						105	257		
Coolibah floodplains	B							23	268	
Coolibah floodplains	C									80
Narrow-leaved ironbark ranges	A		6	6						
Narrow-leaved ironbark ranges	B				6	1				
Narrow-leaved ironbark ranges	C						5	2		
Silver-leaved ironbark – duplex	A						13	12		
Silver-leaved ironbark – duplex	B								10	11
Silver-leaved ironbark – duplex	C								3	1

Table 6.18 Summary model parameter inputs into the cost effectiveness optimisation model

Land type	Land condition	Total cost ($/T)	Area of catchment (ha)
Brigalow blackbutt	A	421	28,440
Brigalow blackbutt	B	391	174,800
Brigalow blackbutt	C	59	94,042
Narrow-leaved ironbark woodlands	A	6	192,600
Narrow-leaved ironbark woodlands	B	21	305,413
Narrow-leaved ironbark woodlands	C	6	102,564
Coolibah floodplains	A	362	48,038
Coolibah floodplains	B	291	127,588
Coolibah floodplains	C	80	72,633
Narrow-leaved ironbark ranges	A	12	172,211
Narrow-leaved ironbark ranges	B	7	211,792
Narrow-leaved ironbark ranges	C	7	59,718
Silver-leaved ironbark – duplex	A	25	97,400
Silver-leaved ironbark – duplex	B	21	228,575
Silver-leaved ironbark – duplex	C	4	72,584

To determine the variance that occurs when key parameters change, sensitivity testing was used. The impact of discount rate and price sensitivity testing was modelled for all land types. Two per cent and 10 per cent were tested as the discount rate, and a 25 per cent price decrease and a 25 per cent price increase in beef cattle prices were modelled. All results were within the 95 per cent confidence interval of the base scenario, highlighting that the order of land types to target will not change if prices or discount rate vary.

The least expensive option for sediment reduction is to reduce pasture utilisation rates on silver-leaved ironbark on duplex soils in C condition in 15% TSDM. The catchment has an estimated 72,584 ha of silver-leaved ironbark on duplex soils in C condition (Table 6.19). In order to reduce a tonne of sediment run-off from this land type the bioeconomic model identified that an incentive of $4 per tonne would be needed before landholders would reduce stocking rates and increase ground cover. Given this inexpensive option, the LP model suggests the

entire 72,584 ha should have a 10% improvement in TSDM, costing approximately $295,416 in total. The reduction in pasture utilisation back to 10% reduces sediment exported to 186 kg per ha per year. Across the whole area this would result in a total reduction of 13,501 tonnes per year (Table 6.19).

The next most efficient target for incentive funds is narrow-leaved ironbark woodlands in A condition. The A land condition net present value curve is relatively flat at its peak, making initial reductions in pasture utilisation relatively inexpensive. This land type exhibits low levels of productivity and high sediment export rates. For A condition land an incentive of $6 per tonne is required to induce landholders to move from the economic optimum pasture utilisation rate of 35% back to 25% pasture utilisation, with corresponding lower rates of sediment loss. At this cost-effective rate, it is estimated that the complete 192,600 ha of narrow-leaved ironbark woodlands in A condition that exists in the basin should be targeted with incentives for sediment reduction. If all landholders in this land type and condition participated, it would result in a total reduction of 49,690 tonnes of sediment annually, with a total cost of $1,213,380 (Table 6.19).

The third least expensive option is targeting narrow-leaved ironbark woodlands in C condition to reduce the rates of pasture utilisation from the economic optimum point of 25% TSDM to 10% TSDM. Given the budgetary constraint only 81,867 ha can be achieve before $2m is reached. If this area were targeted, it would result in a reduction of 24,923 tonnes of sediment annually and cost approximately $491,202 (Table 6.19).

Sensitivity testing for the lower discount rate (2%) and higher beef prices (increase of 25%) was conducted. This resulted in an increased optimal pasture utilisation rate and subsequent cost to achieve sediment reductions, reducing the effectiveness of investments in water quality improvements. Conversely, an increased discount rate (10%) and lower beef prices (decrease of 25%) reduced optimal pasture utilisation rates and the opportunity costs of improving land management for water quality improvements.

The results for the two per cent discount rate (Table 6.20) resulted in the price per tonne of sediment reducing as the net present value increased with lower

opportunity costs for beef production. This resulted in the total tonnes of sediment reduced being 34,418 tonnes, with funding only available to cover 135,973 ha of the catchment. The 10 per cent discount rate resulted in lower net present values, and therefore larger areas were able to be targeted for reductions. Under this scenario the total area that was able to be targeted was 367,657 ha of the catchment achieving a reduction of 257,175 tonnes of sediment (Table 6.20).

The impact of a 25 per cent price decrease resulted in 416,556 ha of the catchment being able to be targeted, with a sediment reduction of 357,618 tonnes (Table 6.21). The price reduction also changed the mix of land types and the hectares that can be targeted. The impact of a 25 per cent beef price increase resulted in 182,425 ha of the catchment being targeted, and 1,588 tonnes of sediment being reduced (Table 6.21).

Table 6.19 Results of optimisation model for most cost-effective sediment reduction

Land type	**Silver - leaved ironbark – duplex**	**Narrow-leaved ironbark woodlands**	**Narrow-leaved ironbark woodlands**
Start condition	**C**	**A**	**C**
Cost at 15% pasture utilisation ($/ % TSDM)	3.09		3.00
Cost at 20% pasture utilisation ($/ % TSDM)	0.98		3.00
Cost at 25% pasture utilisation ($/ % TSDM)			
Cost at 30% pasture utilisation ($/ % TSDM)		4.00	
Cost at 35% pasture utilisation ($/ % TSDM)		2.30	
Cost per tonne ($/T)	4	6	6
Cost at catchment scale ($)	295,416	1,213,380	491,202
Area of catchment (ha)	72,584	192,600	102,564
Targeted area (ha)	72,584	192,600	81,867
Total sediment (kg/yr/ha)	186	258	243

Table 6.20 Results of optimisation for sediment reductions with sensitivity testing of a 2% and 10% discount rate

Land type	Silver-leaved ironbark – duplex	Narrow-leaved ironbark woodlands	Narrow-leaved ironbark woodlands	Narrow-leaved ironbark ranges
Start land condition	**C**	**A**	**C**	**C**
Sensitivity testing of 2% discount rate				
15% pasture utilisation (% TSDM)	3			
20% pasture utilisation (% TSDM)	0.8		23.0	
25% pasture utilisation (% TSDM)		26.0	6.0	
30% pasture utilisation (% TSDM)		1.2		
35% pasture utilisation (% TSDM)				3.0
40% pasture utilisation (% TSDM)				12.0
Cost per tonne ($/T)	0	27	29	15
Cost at catchment scale ($)	275,818	1,724,181	0	0
Area of catchment (ha)	72,584	192,600	102,564	59,718
Targeted area (ha)	72,584	63,389	0	0
Total sediment (kg/yr/ha)	186	330	0	0
Total sediment reduced from catchment (T)	13,501	20,918	0	0
Sensitivity testing of 10% discount rate				
Start land condition	C	A	C	C
15% pasture utilisation (% TSDM)	2			
20% pasture utilisation (% TSDM)	0.6		2.6	
25% pasture utilisation (% TSDM)			2.6	
30% pasture utilisation (% TSDM)				
35% pasture utilisation (% TSDM)		5.0		1.8
40% pasture utilisation (% TSDM)		2.6		2.7
Cost per tonne ($/T)	3	8	5	5
Cost at catchment scale ($)	188,718	1,009,227	533,322	268,731
Area of catchment (ha)	72,584	132,793	102,562	59,718
Targeted area (ha)	72,584	192,600	102,564	59,718
Total sediment (kg/yr/ha)	186	543	856	859
Total sediment reduced from catchment (T)	13,501	104,582	87,795	51,298

Table 6.21 Results of optimisation for sediment reductions with sensitivity testing of a 25% price decrease and a 25% price increase

Land type	Narrow-leaved	Narrow-leaved	Narrow-leaved ironbark ranges

	ironbark woodlands	ironbark ranges	
Results of optimisation for sediment reduction with sensitivity testing of a 25% price decrease			
Start land condition	B	A	B
15% pasture utilisation (% TSDM)	4		
20% pasture utilisation (% TSDM)	7		4
25% pasture utilisation (% TSDM)		2	0.5
30% pasture utilisation (% TSDM)		2	
Cost per tonne ($/T)	11	4	5
Cost at catchment scale ($)	358,083	688,844	953,064
Area of catchment (ha)	305,413	172,211	211,792
Targeted area (ha)	32,553	172,211	211,792
Total sediment (kg/yr/ha)	345	1,030	798
Total sediment reduced from catchment	11,231	177,377	169,010

Results of optimisation for sediment reduction with sensitivity testing of a 25% price increase			
Land type	**Narrow-leaved ironbark woodlands**	**Narrow-leaved ironbark ranges**	**Silver-leaved ironbark – duplex**
Start land condition	A	C	C
15% pasture utilisation (% TSDM)			2
20% pasture utilisation (% TSDM)			1
25% pasture utilisation (% TSDM)			
30% pasture utilisation (% TSDM)			
35% pasture utilisation (% TSDM)			
40% pasture utilisation (% TSDM)	10		
45% pasture utilisation (% TSDM)	14		
50% pasture utilisation (% TSDM)		7.8	
55% pasture utilisation (% TSDM)		1.9	
Cost per tonne ($/T)	24	10	3
Cost at catchment scale ($)	1,202,952	579,265	217,751
Area of catchment (ha)	192,600	59,718	72,584
Targeted area (ha)	50,123	59,718	72,584
Total sediment (kg/yr/ha)	543	859	186

The total reduction in sediment that could be achieved with the available $2 million allocated for sediment reduction would be 83,085 tonnes of sediment per year. This is based on the assumption that an initial payment would be required to ensure long term (20 years) management change[6]. It is estimated that the total sediment exported by anthropogenic sources is 14 million tonnes, of which 4.1

[6] The bioeconomic model is predicting the average cost over a 20 year period

million is estimated to be from the Fitzroy Basin (Queensland Department of the Premier and Cabinet 2011). Assuming a 20% reduction target from *Reef Plan* targets, these reductions in sediment modelled by the LP model would achieve 10% of the 820,000 tonnes required. While all other land types and land conditions would contribute to reduction in sediment exported, the land types and land condition selected by the LP model present the most cost-effective way to achieve a sediment reduction in the land types used in this case study.

6.6 Discussion

First, the analysis demonstrates the complexity of the economic trade-offs involved, and that large variations in optimal grazing pressures and opportunity costs of change exist across both land types and land condition factors. Second, the results show at a practical level that policymakers are able to target investments more strategically and cost effectively. It also reveals the costs of investing in management changes in other land types and conditions, where the high opportunity costs have the potential to absorb substantial public investments with little effective return. The third conclusion is that the results help to predict the levels of cost associated with different targets for sediment reduction and demonstrate that marginal costs of sediment reduction increase sharply as higher and higher levels of reduction are sought.

The results highlight the complexity of achieving sediment reductions from grazing lands for improved water quality outcomes and the importance of informed decision-making. This research identifies that the three key variables to consider when targeting sediment reduction investment are tree basal area, land condition and land type.

In economic terms, the impact of tree basal area significantly alters the viability of regenerating land, as shown in the bioeconomic modelling. In treed areas for all land types in this analysis, incentives are required to reduce pasture utilisation rates and sediment exported. This is due to the reduced productivity of the land when trees are competing with pasture species for nutrients and water. The large sediment export rates in the bioeconomic model also reflect that at low pasture utilisation rates, there will be increased leaf matter that initially will be exported.

Starting land condition also needs to be considered when determining which policy or program is most effective for sediment reduction. For cleared land initially in C condition, regeneration would be a viable investment for the landholder to undertake alone, particularly for the more productive land types such as silver-leaved ironbark at Nebo. Land condition also presents the cheapest option in all cases to reduce sediment loss.

For the land types with increased inherent productivity (e.g. silver-leaved ironbark at Nebo), regeneration works were economically viable for the landholder to undertake alone. In some cases, extension programs are recommended to reduce pasture utilisation back to the economic optimum levels. For land types with low productivity and high sediment rates, such as spotted gun and silver-leaved ironbark at Springvale, it does not appear that regeneration is economically viable when in D condition. However, these land types have cheap options for sediment reduction through decreased pasture utilisation, which could be achieved through providing incentives for landholders not to graze these land types.

For all scenarios where an economic optimum pasture utilisation rate was identified, it would be expected that landholders would want to operate at this level. However, it must be understood that the bioeconomic model has perfect knowledge at the start of the season regarding growth and pasture availability. In order to achieve sediment reductions and to get landholders to further decrease the level of pasture utilisation below the economic optimum level, a financial incentive may be required.

If the landholder is already operating to the right of the economic optimum point, then the cost of reducing a tonne of sediment further is increased, as the opportunity cost of not utilising the pasture increases. However, if the landholder is operating to the left of the economic optimum point, then sediment reductions are most effectively dealt with through extension and education activities.

There are some deficiencies in such an extensive modelling process and in the interactions between both the GRASP pasture modelling and the economic modelling. In all of the modelling, perfect knowledge has been assumed; in reality this is impossible for landholders to achieve, especially for extended

periods of time into the future. It must also be noted that the economic model maintains the AEs derived from GRASP, and this involves trading cattle and drought feeding. The economic parameters are based on an 'average' property, although it is acknowledged that some landholders may have deviations from this.

6.6.1 Efficient programs and policies

The optimisation analysis provides the basis for targeted investment on cost effectiveness, addressing criticism of previous NRM policies in Australia (Pannell and Roberts 2010). The bioeconomic modelling demonstrates that the cost of achieving sediment reductions varies by more than 100 times over the selected case studies, and that opportunity costs are very sensitive to the existing land condition. The results highlight the importance of targeted policy mechanisms based on biophysical and economic information to allow for efficient use of funding and selection of the most effective land types.

The ability of the FBA to allocate incentives focused on predicted outputs (graziers reaching the lower pasture utilisation) rather than on inputs (fencing, sowing of native pastures) would allow funds to be more efficiently targeted. This would allow the organisation to strategically focus on specific land types, with specific land conditions, in order to achieve the targets identified in its *Fitzroy Basin Water Quality Improvement Report* (2008).

The results also demonstrate the small percentage of sediment reductions that could be achieved given the incentives available. However it must be noted that although this case study used $2 million the actual amount of funds available to be spent across the Reef Rescue Program has been approximately $30 million. The sensitivity testing demonstrated that changes in beef prices were more significant on the results than changes in the discount rate, although the outcomes of all sensitivity tests were in the 95% confidence interval of the initial results.

6.6.2 Limitations

The confidence intervals that surround the use of both the biophysical and GRASP modelling must also be considered, along with the complexity and error margins around satellite imagery. This analysis involves an implicit assumption

that reducing sediment emissions at the paddock level will subsequently reduce emissions at the lagoon level, leading to improvements in water quality. However, the temporal lags between the load entering the river and the load entering into the lagoon have not been accounted for. The impact of gully and stream bank erosion has also not been accounted for (Thorburn and Wilkinson). This has cumulative impacts on the confidence intervals that are a limitation of the study.

Other limitations of the study should also be noted. The bioeconomic modelling has not captured all biophysical factors, such as site-specific effects, while the biological models may not have adequately reflected cumulative and threshold effects (Wu and Boggess 1999; Wu and Skelton-Groth 2002). Also, the potential for multiple environmental benefits has not been recognised in the budget allocation. The bioeconomic models are simplistic in assuming that each modelled property consists of one land type and that the pasture utilisation rates are applied uniformly across each property. The assumptions that landholders are always profit maximising and have perfect knowledge should also be noted.

6.7 Summary

The results from the bioeconomic modelling highlight the heterogeneity and complexity of rangelands grazing systems. The results provide insights into the land type characteristics and properties that affect the cost of sediment reductions and influence the policy mechanisms that are the most cost effective to implement.

CHAPTER SEVEN: MEASURING ENVIRONMENTAL BENEFITS

A good an individual gains contentment from and is willing to give up has a value. Some values, however, are not expressed through markets but are still an important component of the total economic value (TEV). The notion of TEV provides an all-encompassing measure of the economic value of any environmental asset. Figure 7.1 illustrates the methods to assess values for environmental benefits. These components will be explored throughout this section.

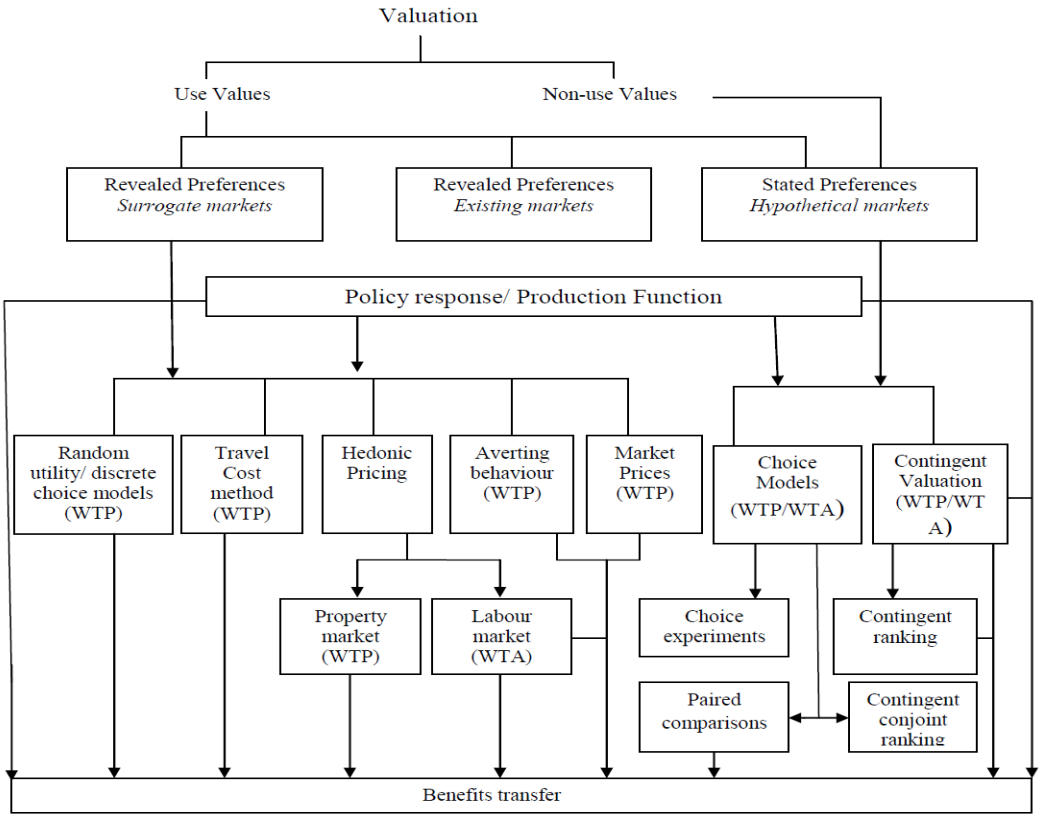


Figure 7.1 Valuation methods

Source: (Pearce *et al.* 2006)

Consider Figure 7.1 where valuation values can be divided into use and non-use values. Use values refer to the actual use of an asset such as timber from a forest, planned use or possible use. Possible use becomes 'option values', where there is WTP to maintain the good or preserve the good for future use (Plottu and Plottu 2007). Non-use values refer to the WTP to maintain some good even though there is no actual, planned or possible use. A common classification for these values is:

1. Existence: WTP is due to the individual feeling some 'stewardship' or responsibility for the asset
2. Altruistic value: an individual is concerned that the good in question should have provision for others in the current generation
3. Bequest values: held by individuals to see provision of the good for future generations.

(Plottu and Plottu 2007)

Market and non-market values can be estimated through revealed preference techniques (RP) and stated preference techniques (SP). In existing markets, revealed preferences are shown through the price mechanism. The focus of this research, however, is on values for non-market environmental goods. RP methods estimate people's WTP for non-market environmental benefits through observing their behaviours in surrogate markets that are most closely related to the value of interest (Adamowicz *et al.* 1994). However, they can only be used where there is a quantifiable relationship between the market and the non-market good. SP techniques involve valuing non-market goods by asking people about their preferences.

7.1 Revealed preference techniques

Revealed preference techniques involve looking at surrogate markets for private goods and services that are related to the environmental assets of concern. Individuals reveal their preferences for both the market good and the environmental good when purchasing the private good. Pearce and Moran (1994) explain that individuals leave a 'behaviour trail' as they make these actual

decisions that affect their lives. Surrogate market approaches included the revealed preference techniques of the travel cost method and hedonic pricing.

7.1.1 Travel cost method

The travel cost method is used to value recreational uses of the environment. The method uses information about the number of trips taken and the sacrifices made (trip cost) to value the good (Champ and Boyle 2003). Essentially, the longer the distance travelled the higher the cost and the lower the visitation. This gives rise to a 'trip visitation' function, which is used to estimate the surplus associated with visiting a recreational site (Bennett 2005a; Zerbe and Bellas 2006).

The travel cost method is relatively inexpensive to apply. However, issues in applying this method include choice of dependent variable, treatment of multi-purpose and multi-destination trips, calculation of distance costs, holiday-makers versus residents, the availability of substitute sites that may affect values, the value of time and sampling biases (Pearce and Turner 1990).

The data requirements of the approach are quite substantial, with a survey required to determine the number of visitors to the site, the visitors' places of origin, the duration of their journeys, the time spent at the site, direct travel expenses and values placed on time by the respondents. It can be challenging to value a whole range of environmental quality attributes for sites and substitute sites as well as socioeconomic and demographic characteristics (Pearce and Moran 1994).

7.1.2 Hedonic pricing

Hedonic pricing involves estimating the WTP of consumers for particular attributes through a statistical decomposition of their actual purchase decisions. The approach observes marketplace behaviour to identify relative values of the component parts of the good and is often used to assess environmental components of property values (Donnelly 1991). Hedonic pricing has been implemented for environmental valuation in relation to the individuals' WTP for a property with different sets of attributes. For instance comparing property prices with different levels of environmental attributes can be used to estimate WTP to secure the different attributes. However, this method cannot be used for

all environmental goods, as market-complementary goods are not always available (Mazur and Bennett 2008).

The method is applied mainly to housing price and labour market wage observations and involves assumptions that individuals choose a residential location that maximises their utility and that the marginal rates of substitution between local amenities and other goods will equal the price ratio (Bayer *et al.* 2009). Although the method allows for policies to be ranked through hedonic price indexes, it is often difficult to specify the functional form of the relationships (Donnelly 1991).

Hedonic pricing only accounts for direct use values. Services such as flood control, water quality improvement, and groundwater recharge may provide benefits to individuals beyond the consumption of the good. This results in some cases that cannot be fully valued with the technique (Birol *et al.* 2006).

7.2 Stated preference techniques

SP techniques can estimate both non-use and use values and are able to capture change in TEV. The methods involve asking people to state their preference for predefined alternatives of environmental outcomes (Boxall *et al.* 1996). SP approaches are survey-based and elicit participants' future behaviour in constructed markets. Non-use, or 'passive', values are held by people who may not use the asset, but who want to preserve or improve it. SP methods also provide more information than the WTP, such as the motivations for choice, such as altruism, stewardship, and concerns for future generations (Pearce 2002).

The survey used in an SP experiment describes a hypothetical market where the good concerned can be traded. This market defines the good itself, the context in which it would be provided and how it would be financed (Pearce *et al.* 2006). SP techniques are attractive to implement, because – in principle – all kinds of economic values related to a project policy can be elicited. There are two key methods: the contingent valuation method (CV) and the choice modelling method (CM). The main difference is that the CV method involves asking people to choose between the status quo scenario and a single alternative, whereas CM

presents respondents with choices between several alternatives (Bennett and Blamey 2001).

7.2.1 Contingent valuation (CV) method

The CV method has commonly been used to evaluate the effect of a single project on the environment (Bennett and Blamey 2001; Scarpa *et al.* 2000. People are asked either directly or indirectly how much they are willing to pay for an increase in the quality or quantity of an environmental asset. In the past, this method has used open-ended direct WTP questions. However, it has been argued that this approach makes it difficult for people to express their maximum or minimum WTP for a good. To overcome this, closed ended or 'dichotomous choice' questions in CV methods are now used more widely (Bennett and Blamey 2001).

There are three components to a CV survey. Firstly, it asks a set of attitudinal and behavioural questions regarding the good to be valued. This allows respondents to prepare for the valuation question and to reveal underlying attitudes to the good. Secondly, the contingent scenario is presented and respondents are asked for their monetary evaluation. The scenario includes a description of the good and the hypothetical terms under which it is offered. This hypothetical scenario includes a description of the policy of interest, a description of the constructed market and the description of the payment method. Finally, questions regarding the respondents' socio-economic and demographic characteristics are asked to determine the representativeness of the survey sample relative to the population of interest (Pearce *et al*. 2006). Econometric methods are then applied to the results to derive the desired welfare measures such as the mean or median WTP.

Concerns regarding CV- generated estimates were voiced in Australia after the valuation of environmental damage at the mine in Coronation Hill, adjacent to Kakadu National Park; similarly, controversy peaked in the United States with the Exxon Valdez grounding (Bennett *et al.* 1998; Carson *et al.* 2003). To address the debate about the validity of the technique, a panel of experts was drafted by the National Oceanic and Atmosphere Administration (NOAA) to assess the appropriateness of the CV to provide reliable estimates of non-use of

existence values. The panel concluded that CV can produce sufficiently reliable estimates to be the starting point for a judicial process. The panel also recommended a number of guidelines which largely related to debates over scoping and framing effects. The concerns around the methodology include:

1. Embedding effects
2. Part–whole bias
3. Hypothetical bias
4. Payment vehicle bias
5. Strategic bias
6. Starting point bias
7. Information bias
8. Non-response bias

Embedding effects happen when the mean estimated WTP for a particular good is lower when valued as part of a more inclusive good, rather than on its own (Morrison *et al.* 1996). Carson (1995) explained that there are two kinds of embedding effect: regular and perfect. Regular embedding is when substitute goods are 'embedded' under a larger grouping of the good (e.g. all remnant vegetation), resulting in respondents lowering their marginal values for successive units of substitutes. Perfect embedding is when only the 'warm glow' of giving is reflected in CV responses.

Related to this is part–whole bias, which is when respondents are insensitive to the scope of the good they are asked to value. That is, the value estimates of part of the good are found to be the same as the value estimates of the whole good, indicating that the scope of the good has no impact on the value estimates generated (Hanley *et al.* 1998a; Carson *et al.* 2001).

Bennett *et al*. (1998) explored these scope effects in the context of the proposal for protection of the Nadgee Nature Reserve on the NSW far south coast, and Carson (1995) examined the environmental costs likely to occur if mining at Coronation Hill was permitted. Scoping problems were identified by the NOAA panel as one of the most important internal arguments against CV and therefore critically important to consider. Carson (1995) findings highlighted the

importance of being aware of ‘perfect embedding’ and the need to communicate the context of the good. To do this effectively, focus groups and surveys are required to determine how familiar respondents are to the good and also the array of substitute and complementary goods. The two studies also highlighted the need to test the validity of results against a perfect embedding hypothesis to allow the analysis of scope sensitivity (Carson 1995; Bennett *et al.* 1998).

Hypothetical bias poses a problem for all SP techniques. It occurs when respondents do not believe that their answers to a question will have any policy significance. This can result in respondents having little incentive to consider the environmental good in question or the payment outcomes (Morrison *et al.* 1996).

Payment vehicle bias can occur in three forms. Firstly, the payment vehicle may not be relevant to all respondents. For example, in a survey that includes questions with a payment vehicle of increased water rates, the sample may contain people who do not pay these rates and therefore ‘free ride’ in the CV survey. Secondly, respondents may not believe that the payment vehicle will ever be implemented. Finally, respondents may object on ethical grounds to the implementation of the specified payment vehicle, in which case they may give protest responses (Blamey *et al.* 1999).

Strategic bias occurs when respondents with a pro-conservation disposition deliberately overstate their true bid, or a respondent who is pro-development underestimates their true bid. Strategic bias may occur because the respondents are not willing to go through the process as an interactive bidding process because they value their time highly or become bored (Boyle *et al.* 1985).

Starting point bias occurs when the amount at which the initial bid is set affects the bid distribution. This may occur if the starting bid implies to the respondent the approximate range of ‘appropriate bids’ or because the good is poorly defined. Information bias occurs when respondents feel that the information provided in hypothetical scenario descriptions is true and accurate. Background information regarding costs, existing payments or causes of the problem affect estimates, and these information cues can result in altruism affects.

Finally, non-response bias arises when there is a low response rate, which could mean that the sample is not representative of the population.

Choice modelling allows embedding effects to be managed with explicit inclusion of substitutes, and non-response bias to be managed with choices that are potentially more easily answered. The payment vehicle may also be de-emphasised with multiple payment mechanisms to avoid such bias. Choice modelling may be affected by similar strategic biases, but it has a stronger framework for dealing with such biases.

7.2.2 Choice modelling

Choice modelling (CM), also known as choice experiments and choice-based conjoin analysis, involves deriving responses to predefined alternatives (Boxall *et al.* 1996). It involves people choosing between different products (Adamonwicz *et al.* 1998). Using key product characteristics, variables or attributes in experimental design, the methodology formulates alternative product scenarios. Statistical methods are then used to value preferred attributes and simulate preferences, choices or value options (Bennett and Blamey 2001).

The information obtained from CM includes the attributes that determine the values that people place on non-market goods, the ranking of these attributes within the relevant population, the value of changing a bundle of attributes all together and the changes to the TEV of the good (Adamowicz *et al.* 1998).

There are four ways in which preferences can be measured through different types of stated preference experiments:

1. contingent rating: a series of alternatives are to be rated; however, careful design is required to determine consistent welfare estimates

2. contingent ranking: the respondent is required to rank scenarios on a scale of one to ten

3. paired comparisons: the respondent is presented with pairs of scenarios on a similar scale to choose between

4. choice modelling: where the respondent is required to select between two or more alternatives (where one is the status quo).

These techniques differ in their ability to provide quality welfare estimates and in their degree of complexity. The remainder of this section will focus on the CM method.

In CM, the respondent is presented with a series of choices described by attributes, levels and labels. One alternative in each choice set is the 'status quo' option or business-as-usual scenario. The respondent is then asked to select their preferred choice. The utility function consists of two components: the function of the attributes of the good and the unobservable influences on personal choice. The characteristics of the good described through use of attributes allow for the respondent's utility for the good to be derived. For example, if a respondent prefers option A to alternative options, this can be expressed as the probability that the utility associated with option A exceeds all other options. Socio-economic factors may also influence choice, and this may be quantified in the analysis (Pearce *et al.* 2006).

To create a choice experiment there are several steps:

- Firstly, the attributes of the good being valued must be determined through a literature review and focus groups. This involves understanding the impact of the prospective policy. Usually a monetary trade-off is included to allow the WTP to be derived.
- Secondly, the attribute levels are assigned. These should be assigned non-linearly, spaced between the maximum value and the minimum. The levels must also be realistic, and span the range of respondents' preference maps as determined through consultation with focus groups and the literature review. The business-as-usual or status quo scenario is also included.
- Thirdly, the experimental design is then selected. Statistical design theory is used to combine the levels of the attributes into a number of alternate scenarios presented to the respondent. Approaches include factorial design, fractional-factorial design and efficient design; they are explained further later in this chapter.

- The construction of the choice set is the next step in the process. The splits that are determined by the experimental design are then grouped together and presented to respondents.

Once completed, the respondents' preferences must be measured. Preferences can be measured through ranking, rating or choices. A statistical estimation procedure is then implemented to determine the part-worth or marginal change in WTP for the good in question.

The theoretical basis for CM is the random utility model (RUM) (Train 2009). The RUM is based on the researcher only being able to observe part of the respondent's utilities. The unobserved component is taken to be randomly distributed. Under the RUM, U_{an}, utility that the respondent n enjoys from choice alternative a can be described by:

$$U_{an} = V_{an} + \varepsilon_{an} \qquad \text{(Equation12)}$$

Where V_{an} is the deterministic observable component of the utility that respondent n has for option a. ε_{an} is the stochastic unobserved component of the utility associated with option a and consumer n.

The observed component (V_{an}) is a function of the attributes Z_{an} and of individual characteristics S_n and a set of unknown parameters (Rolfe et al. 2000).

$$U_{an} = u\,(Z_{an}, S_n) + \varepsilon_{an} \qquad \text{(Equation 13)}$$

Due to the random component, utilities can never exactly be determined. However, what can be concluded is that if respondent n chooses from choice set C_n, then it is probable that the deterministic and stochastic components of that option are greater than the deterministic and stochastic components of the other option j in the same choice set. This can be expressed as:

$$P(a/C_n) = P((V_{an} + \varepsilon_{an}) > (V_{jn} + \varepsilon_{jn})) \text{ for } j \text{ options in a choice set } C_n,\ a \neq j$$

(Equation 14)

The greater the difference in observed utility, the greater the probability of choosing alternative *a*. As the distribution of the random component is unknown, in order to estimate the probabilities, assumptions regarding the distribution of the random component must be made. The standard assumption is that the ε terms are independently and identically distributed Gambel random variables that lead to binary or multinomial logit models (MNL) (Train 2009).

Under this assumption, the probability that an individual *n* chooses alternative *a* over *j* can be represented as:

$P_a / C_n = exp\ (\lambda\ x_{an}) / \sum exp\ (\lambda\ x_{aj})$ for all *j* choice *C*; (Equation 15)

Where λ is a scale parameter which is usually normalised to one. The scale parameter is inversely proportional to the standard deviation of the error distribution (Rolfe et al. 2000). The MNL model generates a utility function of the form:

$$V_{an} = \beta_a + \sum_k \beta_k X_{kn} + \sum_p \theta_p Z_{pn} + \sum_{kp} \gamma_{kp} x_{kn} Z_{pn} + \sum_{pa} \Psi_{pa} \beta_a Z_{pn}$$

(Equation 16)

where (Mazur and Bennett 2008) explain:

> β_a is a vector of 'intercept' terms (alternative specific constants – ASCs) for *A-1* of the *a=1, …., A* choice options
>
> B_k is a matrix of *k=1,….,K* attributes that relate to choice options, X_{kn}
>
> γ_p is a matrix of *p=1,….,P* characteristics that relate to individual respondents, Z_{pm}
>
> γ_{kp} is a matrix of possible relationships of choice option attributes with the characteristic of the individuals, $X_{kn}Z_{pn}$
>
> Ψ_{pa} is a vector of possible interactions between individual characteristics and choice option intercepts.

The utility function estimated for each alternative contains the effects of attributes, an ASC and the individual characteristics that can be interacted with the attributes of the ASC (Blamey 2001). Any variation in choices that is not able to be explained by the attributes or the socio-economic characteristics is captured by ASCs (Rolfe *et al.* 2000).

Welfare estimates from the MNL model are expressed by:

$$CS = -1/\alpha \, (\ln \sum \exp V_{an} - \ln \sum \exp V_{jn}) \qquad \text{(Equation 17)}$$

Where:

> *CS* is the compensating surplus welfare measure and α is the marginal utility income as reflected by the β coefficient of the other monetary attribute, which is the consumer surplus for changes in a single attribute (Rolfe et al. 2000):

$$W = -1 \, (\beta \text{ attribute}/\beta \text{ money}) \qquad \text{(Equation 18)}$$

This formula allows an estimate of the trade-offs between the non-market attributes and the cost attribute. It estimates how much the respondent is willing to pay for gaining or losing units of the attribute (Mazur and Bennett 2008).

CM provides important advantages over other non-market valuation techniques, such as its flexibility and ability to assess non-use values and to decompose values by attributes. It also has appeal in providing information to policy through identification of marginal trade-offs between attributes.

Choice experiments provide a measurement of use and non-use (passive use) values. Passive use values measure the economic value from a change in environmental quality that is not reflected in an observable behaviour. Initially used in marketing and transportation, before being implemented in environmental valuation, the method provides an alternative or complementary valuation to contingent valuation (Adamowicz *et al.* 1998).

7.3 Choice modelling environmental applications

There has been significant use of non-market valuation for water quality improvements for environmental goods in Australia, such as the GBR, water supply options and preservation of remnant vegetation.

Adamowicz *et al*. (1998) highlight that the advantage of CM is the ability for the technique to provide a richer description of the attribute trade-offs that individuals are willing to make. Bateman *et al*. (2002) suggest that CM allows four messages to be conveyed in a policy context:

1. Attributes are significant determinants of the values people place on non-market goods.
2. The implied ranking of attributes in the target population allows values to be derived.
3. CM estimates the impacts on specific attributes. For example, to capture how one options alters another attribute's value.
4. CM allows the estimate the total economic value of a resource or good.

Blamey (1999) used CM to value multiple water supply options in the Australian Capital Territory. The study investigated different policy options, using the attributes of quality of water available for the household, quality and perceived quality of the water used, annual household costs of water, the aquatic and riparian environment, endangered species losing habitat, and appearance of urban environment.

Blamey *et al*. (2000) explored valuing remnant vegetation in central Queensland, using the attributes of reduction in population size of non-threatened species, endangered species lost in the region, and changes in regional income and employment. Respondents were presented with the status quo and two alternatives for increased vegetation protection. The study found that the community valued more stringent policies than were currently being applied.

Similarly, Rolfe *et al*. (2000) explored the community values for tree clearing in the Desert Uplands. The attributes used were reductions in the population size of non-threatened species and unique ecosystems, the number of endangered species lost to the region, and changes in regional income and employment. The

fact that the land being cleared was only marginally improving the production capacity of the land resulted in the community values for biodiversity to be higher. The community valued avoiding the loss of non-threatened species to be $1.69 for each 1% reduction, and WTP to avoid unique ecosystem loss was $3.68 per 1% reduction. Maintaining endangered species were valued at $11.39, job preservation to be $3.04 per job, and maintain each million dollars of regional income to be $5.60.

To evaluate policy measures for reef protection measures Rolfe and Windle (2010) also conducted a CM experiment. The choice experiment identified the pressures coming from land-based activities, urban and industrial activities, ocean-based activities, natural events and climate change. Using the attributes of the amount of the GBR in good condition, level of certainty that reef health would improve and cost of protection measures, the choice experiment was conducted with a split sample or labelled and unlabelled experiments.

The experiment explored the community values across the QLD population. Their results showed that the average Brisbane household was willing to pay approximately $26.37 for each additional one per cent improvement in GBR health. This gave an aggregated range between $132.8 million and $171.5 million per one per cent improvement, depending on the assumptions used regarding the discount rate. Rolfe and Windle (2010) highlight that with the current Reef Rescue program ($200 million) there would need to be a 1.2–1.5 per cent improvement in the health of the GBR to deliver net benefits. The impact of the labelled management options also indicated that 'Increased conservation zones' were valued slightly higher although not significantly higher than 'Improving water quality. However, there was increased variance in the values associated with 'Increasing conservation zones', indicating varying support of this management practice.

To understand the national value for the GBR and the impact of scope and scale in valuing non-market goods, Rolfe and Windle (2010) conducted a national choice experiment. Sampling from Townsville, Brisbane, Sydney, Melbourne, Adelaide and Perth populations, the experiment sought to understand if the WTP values for resource protection would decline with distance from the resource.

The results showed there was no impact of scope and scale on the communities values.

Rolfe and Windle (2011) used a CM experiment to calculate the community benefit of linking water quality science with agricultural pollutant emissions to the Great Barrier Reef. The choice experiment used the attributes of water quality improvements, the amount of the GBR in good health, the level of certainty that water quality improvements will happen, and an annual payment for five years. The levels for each of the attributes were determined by the current health of the GBR and the likely trends based on scientific research. For the first time policy makers were able to quantify the value of improved water quality and subsequent reef health. Previously the ability to do this has been limited due to three information gaps: scientific information to link management practices with improved reef health, understanding the values for improved reef health, and information about the costs of management practices. Rolfe and Windle (2011) demonstrated the ability to use this information to further inform policy.

This next part of this chapter will step through the critical aspects of completing a CM experiment. Firstly, survey design aspects will be covered (section 7.5), followed by experimental design (section 7.6), and data analysis (section 7.7).

7.4 Survey design

CM involves a complex design process which includes a number of steps to accurately capture the environmental values. Hanley (1998b) suggests that although CM is a structured approach of data generation, careful consideration must be given to the design to help reveal the factors influencing choice. The components that require further consideration will be highlighted below.

7.4.1 Framing the issue

CM offers a number of framing advantages compared with other non-market valuation techniques. This is because the technique can present a pool of substitutes or alternative goods, which ensures that respondents consider the

complementary and substitution effects of goods helping to internalise many scope issues. Secondly, when the attributes are framed correctly, respondents are able to trade-off their various opportunity costs. It also reduces bias issues, as the good can be hidden within the pool of available goods; finally, researchers can analyse and compare CM experiments (Rolfe *et al.* 2002).

Framing the CM survey provides an accurate context for the relevant issue. Framing effects occur when the respondent is unjustifiably sensitive to the context in which the trade-off is offered (Rolfe *et al*. 2002). The policy context and implications concerned with framing, along with the hypothetical market that has been created in a choice model, must be realistic. Unrealistic framing can impact on respondents who have very little understanding of the trade-offs and only respond to very high monetary incentives. In the real world, respondents have more time to consider and analyse their options (Windle and Rolfe 2005). The appropriate frame must make respondents aware of the competing demands for funding and highlight the ways in which people spend their private budget (Mazur and Bennett 2008).

Rolfe *et al*. (2002) conducted three tests for framing effects. The first is a test to check that violations in the model assumptions have not occurred. The presence of these violations would indicate that choices have not been consistent and therefore respondents have had difficulty in framing choices. Secondly, a test can be performed to determine if slight differences in the way choices are presented impacts on model parameters and therefore on the value estimation. If these differences in framing do not result in value estimate changes, then the parameters for the differently framed choice models should be identical. The third test can determine if substantial differences in framing between CM experiments cause changes in the value only to those attributes that are not shared by the choice framework, and potentially those attributes that are shared by the choice frameworks.

Substitutes must also be considered in framing environmental issues, with respondents appreciating other environmental issues that are on the government's agenda. If respondents do not consider these substitutes, the WTP may be overestimated (Blamey *et al.* 1999).

7.4.2 Attribute selection and description

Attribute definition affects all subsequent activities and therefore is one of the most important aspects of design in CM surveys. Ideally, attributes are selected based on what the target population prefers. Equally important is that attributes are able to hold policy-relevant information for decision-makers in the sense that they can realistically be influenced by changes to the current policy setting (Blamey *et al.* 2000; Bateman *et al.* 2002). Most importantly, attributes must be depicted in a comprehensible, realistic, measurable and believable manner, and have policy-relevant context if respondents are to take the survey seriously (Morrison *et al.* 1996; Mallawaarachchi and Quiggin 2001).

Methods to select attributes that adhere to the objectives above include literature review, expert consultation (e.g. with relevant scientists and environmental managers) and focus groups with the target populations (Bateman *et al.* 2002). Focus groups involve a small group of people (usually five to ten), typically consisting of members of the public or individuals who are representative of the population of interest. Focus groups can be used to aid the survey design process through various roles such as developing the overall framework and characteristics of interest for the good in question, selecting attributes, gauging people's understanding of the good and how they value it, describing attributes and determining appropriate willingness to pay (WTP amounts or other measures of change in attributes/level) (Mazur and Bennett 2008). Focus groups can also be used to road-test a survey that has already been drafted, in terms of comprehension and design features.

7.4.3 Scope effects

Scope effects are important aspects to be considered in defining attributes. Scope issues require identifying how respondents will react to scale variation in the choice set to be explored, and understanding whether diminishing marginal utility or threshold effects require the trade-offs to be treated in different ways. The scope of a good refers to the dimensions used to define it and the trade-offs involved; and to the quantities involved (Rolfe and Wang 2008). Windle and Rolfe (2010) explain that there are factors that confound tests between geographic scope effects. These factors include that people may consider

different substitutes as the set of resource possibilities increases. For example, a local population may feel more responsible for their local assets, and have more awareness or knowledge of them in comparison to a more distant population. Regional or local case studies may generate a 'queuing effect', where respondents empathise with the case study and use it to signal their values for the whole asset. As the geographical scope changes, respondents may vary between different attributes that describe a choice set, or between choice experiments that are structured in different ways. These factors may result in lower unit values for larger scoped goods (Rolfe and Windle 2010).

Similarly, it is important to consider embedding effects or the effect of valuing a good as part of an exclusive good rather than valued alone (Bennett *et al.* 1998). Carson (1995) reviewed 31 international studies for the effects of scope and found that all but two were sensitive to scope issues.

7.4.4 Hypothetical bias

Aside from considering framing effects, the researcher must also be mindful of the hypothetical nature of the survey and the effect that bias can have. Minimising hypothetical bias is critical in ensuring that WTP estimates are not overstated, as this then raises concerns about the credibility of the results and their ability to inform policy.

There are many theories about the reasons for hypothetical bias. It has been suggested that it occurs because a portion of respondents are 'fence sitters', and in hypothetical scenarios tend to jump to the positive side (Louviere *et al.* 2000). This is typically associated with a higher WTP; however, in a real payment situation individuals tend to do the opposite. Another suggestion is that it is possible that an individual actually has a range of WTP for a good rather than a specific point. This results in the respondent agreeing to pay a specific price in the hypothetical situation, but in an actual payment context this range is narrower and a negative result occurs (Conway 2005; Morrison and Brown 2009).

To combat these biases, researchers must ensure that the survey is incentive compatible. An incentive compatible survey is one that eliminates incentives for individuals to respond untruthfully or strategically. Another approach is of 'consequentialism' (Burton 2010): where respondents are reminded that their

choices will have consequences, for example, that the results of the study will be used by policymakers. A third approach is to identify biased responses so they can be isolated. For example, certainty scales can take the form of follow-up questions regarding how certain respondents are that they would actually pay the amount suggested in their response.

7.4.5 Cognitive burden

Cognitive burden is a design issue in CM and occurs when respondents are subject to choice tasks being highly complex or difficult to answer. In complex situations, respondents may use strategies or decision heuristics aimed at making the task easier (Bateman *et al.* 2002). The number of attributes, number of levels and number of choice sets may contribute to cognitive burden. Complexity and resulting heuristics may result in the status quo option being selected as it minimises the task required of the respondent (Caussade *et al.* 2005).

The error variance increases as the number of alternatives to select increases. It has been suggested that four alternatives is optimal (Caussade *et al.* 2005); however, DeShazo and Firmo (2012) used a quadratic specification and suggested 3.25 alternatives to be optimal. The number of attributes can also increase the error variance, as having too many attributes introduces cognitive burden. While the number of attributes must describe the public good, there should not be too many (Caussade *et al.* 2005). It has also been found that the number of choice sets was not as important to variance as the number of alternatives, and that nine to ten choice sets are optimal) (Train, 2009).

7.4.6 Payment vehicle

Payment vehicles – such as taxes or entrance fees – are the method by which the good or service will be paid for. It is important that they are acceptable and realistic to ensure validity of the survey. There are two broad categories of payment vehicles: voluntary and coercive. Voluntary payment vehicles include donations and entrance fees. Coercive payment vehicles include environmental taxes or increasing utility in a policy (Champ and Boyle 2003).

While voluntary payments may be appropriate under certain circumstances, they often do not elicit a reliable WTP. Respondents may not consider the payment to be binding, and payments such as entrance fees typically only capture the individuals who want to use the good and not those who may hold some level of non-use value. Voluntary payments can also encourage free-riding, as respondents overstate the hypothetical WTP to secure the good, with no intention of paying any money and relying on other people's contributions.

Coercive payments – such as taxes – introduce a new set of problems, in that respondents need to place trust in the government to spend the tax accordingly, and the payment is irrelevant to non–tax-payers. The payment vehicle that is realistic and representative of how that good would be funded if the hypothetical policy were implemented is the most appropriate vehicle (Champ and Boyle 2003). In an Australian context, many environmental programs are funded by tax dollars, making it an appropriate payment vehicle.

7.5 Sampling size and techniques

This section explains establishing a sample size, which ensure that sufficient information is collected to allow for a robust analysis.

Bateman *et al*. (2002) describe choosing a sample size as a balance between cost and precision, with the optimal size depending on:

1. the smallest sub-group required for sample estimates
2. the precision with which estimates are required
3. variation among the population with respect to the issue of interest.

However, the choice set dimensions and complexity of the choice are the key determinants of the minimum sample size required. Johnson et al. (2007) provide a rule of thumb that can be used to calculate an appropriate sample size, N:

$$N = 500\ (NLEV / NALT \times NREP) \qquad \text{(Equation 19)}$$

where:

NLEV = largest number of levels among attributes

NALT = the number of alternatives in each choice set

NREP = the number of choice sets to be viewed by each respondents.

CM survey techniques include:

1. face-to-face survey collection
2. mail surveys
3. drop-off/pick-up surveys, administered by door-knocking
4. phone surveys
5. internet or web-based surveys.

Face-to-face surveys are conducted by intercepting people at a particular location or by door knocking. This collection method is relatively expensive and interviewers must be trained to avoid interviewer bias.

Paper-based surveys or drop-off/pick-up (or mail back) paper surveys can be administered by door-knocking or through random address selection. These are a more cost-effective option than face-to-face surveys, but are not as efficient as internet or web-based surveys.

Phone surveys usually involve an initial mail or drop-off stage to provide the respondent with a paper copy of the questionnaire, which is often too difficult to describe over the phone. Video/DVD surveys combine both mail out and telephone, with the information framed in the DVD and a mail out survey for the respondent to complete after watching the video/DVD (Bennett 2011).

Internet or web-based surveys are cheaper for collecting data but have some sample selection problems, including the capture of an accurate random sample of the population (particular age and gender categories may be under- or over-represented), and some cities have greater internet connectivity than others. Some research has found that web-based surveys were more cost effective and had a shorter collection time than paper-based surveys (Windle and Rolfe 2009).

7.6 Experimental design

An appropriate experimental design provides enough data for statistical analysis but limits the number of choice sets involved so they are not overwhelming for

respondents in terms of the number of choice or alternatives that need to be answered. This section introduces the techniques for designing choice sets by considering full factorial designs, fractional-factorial designs and efficiency of design.

7.6.1 Full factorial designs

Choice sets comprise a set of alternatives from which respondents can choose their most preferred by making trade-offs between the varying levels of attributes provided in each alternative. Deciding how these levels and alternatives should be arranged in each choice set is a complex task. If respondents consider all possible combinations of all attributes and their levels, then if there are three attributes each with two levels, there are 2x2x2 $=2^3$ = 8 possible options of attributes. Generally speaking, there are L^A possible choice profiles, where L is the number of levels in each attribute and A represents the number of attributes. In simple attribute arrangement such as this, full-factorial design creates alternatives using every combination.

Table 7.1 illustrates that there are eight possible combinations in the choice set of three attributes where only two improvements are offered for each attribute. The advantage of using a full-factorial design is that every possible interaction between attributes can be observed, giving a full set of information (Champ and Boyle 2003). The effect of each attribute can be measured individually, as can the interaction effects; this is known as the design being completely orthogonal (Bateman *et al.* 2002). For example, in Table 7.1, A will appear with C and E, as well as with C and F, and so on, allowing all effects to be measured. It can also be noted that each attribute level appears the same number of times, meaning that they are balanced and the maximum amount of information can be gathered about each particular attribute (Bateman *et al.* 2002).

Table 7.1 All possible combinations of three attributes (X, Y and Z) each with two levels (A and B, C and D, E and F respectively)

	Attributes		
Alternative	**X**	**Y**	**Z**
1	A	C	E
2	A	D	F
3	B	C	F

4	B	D	E
5	A	C	F
6	A	D	E
7	B	C	E
8	B	D	F

7.6.2 Fractional-factorial designs

Because the number of choice sets with full-factorial design is impractical to offer to respondents, using fractional-factorial designs offers a compromise between completeness and a practical design to suit time, resources and respondent participation, although they lose some of the statistical accuracy of the estimated model (Johnson *et al.* 2007).

Fractional-factorial designs choose a fraction of the alternatives from the full-factorial design. For example, from Table 7.1, alternatives five to eight could be selected as a half fraction of the full-factorial (i.e. one in two alternatives are selected from the full design). This design is capable of measuring the main effects of the attributes while maintaining orthogonality and balance for such effects.

The principle of orthogonality does not hold true for all interaction effects in a fractional-factorial design; that is, because a fractional-factorial does not consider all possible combinations of attributes and attribute levels, some interaction effects will be correlated. Thus the interaction and main effects cannot always be separated and measured independently from one another (Johnson *et al.* 2007). Essentially, the fractional-factorial design relies on the assumption that particular attribute effects are not dependant on the levels that other attributes may take (Bateman *et al.* 2002).

The assumption that the main effects are the only relevant effects in a fractional-factorial design gives some cause for concern. Louviere *et al*. (2000) considered the 4^5 design mentioned previously, with a full-factorial comprising 1,024 alternatives, as a one in 64 fraction with 16 alternatives resulting. The main effects of this design will have 15 degrees of freedom (i.e. five attributes at four levels each = (4 – 1)*5 degrees of freedom). This means that only 15 effects are accounted for when considering only the main effects in a fractional-factorial

design. In relation to the full set of alternatives in the full-factorial, there are 1,009 (1,024 – 15) remaining effects that are not accounted for. These effects are interactions and it is likely that some of them could be significant; therefore removing them would bias the results.

Louviere *et al*. (2000) suggest that generally between 70 per cent and 90 per cent of variance is explained by the main effects, while only 5 per cent to 15 per cent of variance results from two-way interactions, and higher interactions are responsible for only a small percentage of the remaining variance. This implies that even in cases where interactions may be of significance, a main effects or fractional design will still provide valid estimates overall (Johnson *et al.* 2007).

Scarpa and Rose (2008) suggest that orthogonality may not be essential to uphold. Orthogonality is an efficient design characteristic only if respondent believes a priori, that all coefficients are equal to zero. However, in many cases a coefficient's value can be reasonably assumed a priori, based on relevant past studies and a general understanding of the target population. Similarly, information about an attribute can be gathered a priori to collecting data.

7.6.3 Efficient design

The resulting alternatives from a fractional-factorial design must be arranged into choice sets according to a pre-selected number of choice sets, and alternatives in each choice set. Obviously, the more alternatives in a choice set the greater the information collected, but the trade-off between this and the increased complexity and burden must be considered. Thus, choice sets must be designed carefully to ensure maximum information.

Efficient design is an alternative approach to orthogonal design, is informative design and ensures that variance is relatively low. Algorithms exist that can randomly search through selections of attribute levels and alternatives to create choice sets and identify optimal combinations of alternatives (Louviere *et al.* 2000) in an efficient design. Manual strategies to create choice sets are suited to small designs, but it is difficult to maintain balance or orthogonality when the design becomes more complex. Hence, it is often preferable to rely on one of the various software packages available.

There are several measures of efficiency, with D-efficiency being one of the most commonly applied. D-efficient or D-optimal designs are aimed at minimising the variance of parameter estimates (Street and Burgess 2007). Another useful efficiency measure is what Scarpa and Rose (2008) label S-efficient designs, which calculate the estimate (number) of replicates required to retrieve a significant attribute co-efficient by utilising the asymptotic standard errors of prior coefficient estimates. That is, S-efficient designs can be used to minimise the required sample size of a study.

7.7 Data analysis

The foundations for CM are presented in section 7.2, in reference to the random utility model (RUM), where the unobservable error component is introduced. The assumption of a particular distribution for the error term is the first step towards defining the probability function to estimate CM data. The independently and identically distributed (IID) restriction means there is no relationship between the unobserved portions of the utility between alternatives, and leads to estimation of probabilities through MNL, also known as the conditional logit, if the errors follow a Gumbel distribution (Train 2009). Empirically, assuming the Gumbel distribution, which implies a logistic distribution of the error differences, leads to similar results as assuming independent normal errors (Train 2009). IID is a restrictive assumption, but in practice seems to be valid in a number of applications. If it is suspected that the IID assumption is violated, alternative models should be used.

7.7.1 Multinomial logit

As specified by Train (2009) in the MNL, the probability of an alternative *i* being chosen by individual *n* is represented as follows:

$$Pni = \frac{e^{\lambda \beta' x_{ni}}}{\sum_j e^{\lambda \beta' x_{nj}}} \qquad \text{(Equation 20)}$$

Where lambda (λ) is the scale parameter. The scale parameter is inversely proportional to the standard deviation of the error term, essentially scaling the

attribute coefficients according to the variance of the unobserved utility (Bateman, Carson et al. 2002). It is not possible to separately identify the scale and beta parameters, thus the estimated parameters are interpreted as scaled marginal utilities. Usually the MNL is estimated by imposing a restriction on the scale parameter for identification purposes. However, the value selected does not affect inferences, and for measures of interest such as WTP, the scale parameter drops out of the ratio of any two coefficients (Train 2009).

7.7.2 Calculation of willingness to pay

A key output from a choice model is an estimate of how much people are willing to pay to receive one more unit of a particular attribute. In CM, WTP is estimated as marginal WTP, or marginal utility, or part-worth. It is the negative ratio of the coefficient for the non-monetary attribute (*a*) to that of the cost attribute (*b*) calculating the dollar value (Bennett and Blamey 2001):

$$Partworth = -\frac{\beta_a}{\beta_b} \qquad \text{(Equation 21)}$$

Note that equation 21 can also be applied to any two non-monetary attribute coefficients to determine the rate at which one can be traded off against the other.

7.7.3 The mixed multinomial logit

The ML model essentially assumes that there is a distribution of marginal utilities across the sample. The typical approach to an ML is centred on random coefficients, making the marginal utility of alternative *j* for individual *n*:

$$U_{nj} = \beta'_n X_{nj} + \varepsilon_{nj} \qquad \text{(Equation 22)}$$

where:

X_{nj} = *Observed variables for individual n and alternative j*

β'_n = *Vector of marginal utilities of the variables, x, for individual n*

ε_{nj} = *Unobserved utility for the individual n and alternative j*

The coefficients vary over individuals within a population of density *f(β),* which is a function of the mean and covariance of the population betas, represented by parameters θ. The primary difference between this and the MNL is that β is allowed to vary over individuals.

In the MNL the error variance is constant for all individuals. By using a heteroskedastic MNL function, scale can be varied across individuals. It is possible to identify the relative value of the error term and exogenous characteristics that vary within or across samples, but are constant across alternatives, including individual-specific characteristics and any imposed survey design differences (Swait 2006). The scale factor (λ) is specified for an individual (*n*) as follows:

$$\lambda_n = h\,(Z_n : \theta) \qquad \text{(Equation 23)}$$

where Z_n is a vector of exogenous characteristics and θ is a parameter vector. Swait *et al*. (2004) propose two appropriate functional forms for h() that maintain non-negativity scale factor:

$$\lambda_n = h\,(\theta\, Zn)^2 \qquad \text{(Equation 24)}$$

$$\lambda_n = exp\,(\theta\, Zn) \qquad \text{(Equation 25)}$$

Differences in scale factors are not the only sources of heterogeneity. Further heterogeneity can be contained within a sample by assuming that the marginal utilities of individuals are not constant but varied. Differences can be obtained by the collection of socio-demographic questions and de-briefing questions in the survey. Heterogeneity in attribute marginal utility can be modelled by respecifying β, from equation 22 for individual *n* and attribute *a* as:

$$\beta_{na} = \beta_a + \delta_a z_n \qquad \text{(Equation 26)}$$

where δ represents the impact of individual characteristics (z_n) on marginal utility.

7.8 Summary

CM provides a robust methodology to explore environmental values, as it allows marginal values and can be applied to policy options. However, CM is a complex methodology and there are various components that must be considered when applying it to a case study.

CHAPTER EIGHT: CHOICE EXPERIMENT: VALUING IMPROVED GREAT BARRIER REEF THROUGH MANAGEMENT PRACTICE CHANGES

The CM experiment methodology outlined in Chapter Seven provides the underpinning theory to this chapter. This chapter firstly introduces the GBR and gives a background to its iconic status and the current environmental issues (Section 8.1). Section 8.2 describes the choice set design, with the attributes and levels explored, then Section 8.3 gives an overview of the experimental design.

8.1 Research focus

The declining health of the GBR has resulted in investment of public funds through programs such as Reef Rescue to achieve targets set in *Reef Plan*. It has been established that sediments and nutrients are key pollutants to the health of the GBR, and a suite of management practices for both sugarcane and grazing have been developed to achieve reductions.

To increase adoption of improved management practices, programs such as Reef Rescue have been implemented to provide incentives for capital infrastructure and equipment. Although $200 million of public funds has been spent on these investments, there is little understanding of the public benefit or WTP to meet *Reef Plan* targets. This therefore frames the question for this CM experiment: what is the public benefit of improved management practices for pollutant reductions?

8.2 Choice set design

Preceding the experimental design of the choice sets, there are a number of factors to consider. Firstly, the number of alternatives and attributes needs to be determined, then the splits in the survey need to be considered to allow comparisons between samples and attributes.

8.2.1 Number of alternatives and attributes

With respect to cognitive burden and the complexity issues as discussed in Chapter Seven, the literature identifies three or four alternatives within each choice set as being optimal (De Shazo and Firmo 2002; Caussade *et al.* 2005). In this experiment two alternatives of altering conditions in the grazing and sugarcane industries were presented, along with the status quo in the choice sets. This reflects the two main industries and also fits with the recommendations in the literature.

To assess the views and values of the general public based both in communities in coastal regions close to the GBR and in Brisbane, a random sample across the two populations was taken. The random sample should be representative of the population of interest as well as capturing the diversity of demographic variation in the population. This gives the ability to understand if any of the particular demographics influence choice and, ultimately, WTP.

Focus groups were used to identify the key attributes and the frame of reference relating to water quality and sediment reductions. Results of the focus group sessions and a review of the literature indicated that the community were influenced by a limited number of attributes in their preference for water quality improvements. Using the focus groups these were condensed into the following key attributes to allow the policy options to be described adequately and reduce the potential for cognitive burden. (De Shazo and Firmo 2002; Caussade *et al.* 2005).

- Sediment reductions
- Nutrient reductions
- inshore reef health
- cost.

These attributes also ensure the links between pollutant reductions and improved GBR health were considered. The key pollutants from the grazing and sugarcane industries were identified as sediment and nutrient respectively. These pollutants, or inputs (sediment and nutrient), influence inshore reef health, which is essentially an output (reef health); however, there is a lack of directly related

certainty of achieving changes in reef health. Finally, cost is the input required by the public to achieve the change in inshore reef health.

Sediment

As described in Chapter Two, rangelands are complex ecosystems which vary significantly depending on land type, rainfall and land type resilience. A strong correlation between stocking rates and grazing strategy to ensure that land degradation does not occur has been established in Chapter five.

Setting the upper levels

The *Reef Water Quality protection Plan – First Report 2009 Baseline* (Queensland Department of the Premier and Cabinet 2011) indicates that 14 million tonnes of sediment exported each year into the GBR is attributed to human activity. With 90 per cent of the land use being for grazing, it is assumed that 80 per cent is attributed to the grazing industry, and the remaining 20 per cent is derived from the sugar industry. *Reef Plan* targets a 20% reduction in sediment by 2020 (Queensland Department of the Premier and Cabinet 2009). In the experiment the upper levels for the sediment reduced attribute were a reduction in sediment by 16% from grazing lands and a reduction of 4% from sugarcane-growing land.

Nutrient

Nutrient delivery to the GBR comes from a range of sources, such as urban storm water, atmospheric inputs from rainfall events, planktonic and microphytobenthic nitrogen fixation and deeper ocean supply. However, the largest single source comes from discharge from rivers and is predominately from application of fertiliser to crops (such as sugarcane). Additional losses of particulate-bound nutrients from soil tillage and decreased pasture cover lead to increased natural nitrogen and phosphorus from soil to waterways (McKergow *et al.* 2005b; Brodie *et al.* 2011).

Setting the upper levels

The *Reef Water Quality protection Plan – First Report 2009 Baseline* (Queensland Department of the Premier and Cabinet 2011) indicates that 66,000

tonnes of total nitrogen and 14,000 tonnes of total phosphorus exported each year are attributed to human activity. With sugarcane production being the main exporter of nitrogen, it was assumed for the experiments that 60 per cent was attributed to sugarcane, and that the remaining 40 per cent was derived from the grazing industry. *Reef Plan* targets a 50 per cent reduction in nutrients, both nitrogen and phosphorus (Queensland Department of the Premier and Cabinet 2003), by 2013, and therefore the upper levels for nutrient reduced attribute were a reduction in nitrogen of 40 per cent from grazing lands and a reduction of 60 per cent from the sugarcane industry. Although the *Reef Plan* targets specify a reduction by 2013, it is expected that changes in reef health will not occur immediately and to restore full health may be an extended time frame. Therefore 2050 was selected as the date by which to observe change in reef health.

Inshore reef health

There are various components to overall reef health, including coral health, diversity of coral species, seagrass health and fish species. These characteristics vary considerably along and across the GBR, depending on the location and proximity to the coast (Fabricius *et al.* 2011). However, for the purpose of this research, inshore reef health was the amenity to be valued, as the changed land management practices did not specifically target any one of these characteristics. Rolfe and Windle (2010c) explored the value for inshore reef health at three different locations: Cairns, Townsville and the Capricorn coast. They estimated from the research that currently for the Cairns region, 75 per cent or 282 km^2 of the inshore reef was in good health. Townsville had 45 percent or 117 km^2 and the Capricorn coast had 85 per cent or 23 km^2 of inshore reef in good health. From this research an average of the three proportions were calculated, and the current level of inshore reef in good health was set at 68 per cent or 2,870 km^2. The status quo for inshore reef good health was set at 50 per cent, based on estimates from Windle and Rolfe (2010a) who used the value for inshore reef health in 25 years in their choice experiment. Although the time period for the targets is by 2020, it was assumed that it would take until 2050 before a significant change in inshore reef health would be achieved.

Setting the upper levels

Rolfe and Windle (2010c) also estimate that the most improvement that could be achieved would be to have inshore reef good health at 70 per cent. This is derived from a maximum of 12 per cent improvement of inshore reef health from current levels made through the conservative estimates of De'ath and Fabricius (2010).

Cost

The payment vehicle was worded to capture the different ways that costs could increase as well to avoid protest responses. The cost attribute was described as potential increase in food prices, an annual increase in taxes, or an annual increase in council rates. The cost values were tested in the focus group to understand what the maximum was that respondents would be willing to pay and what costs they considered reasonable. $0 was given as the status quo for no change, and the upper limit was set at $250 per year, with five levels of cost options. Level differences followed a broadly logarithmic scale, rounded to convenient dollar amounts, so as to better capture sensitivity to amount changes.

Setting the upper levels

Completing a full-factorial design would have resulted in an unreasonably high number of alternatives, and so a fractional-factorial design was used. The design was created in the Ngene statistical software package. To allow for balance, the design of 12 choice sets was blocked into two groups (Table 8.1).

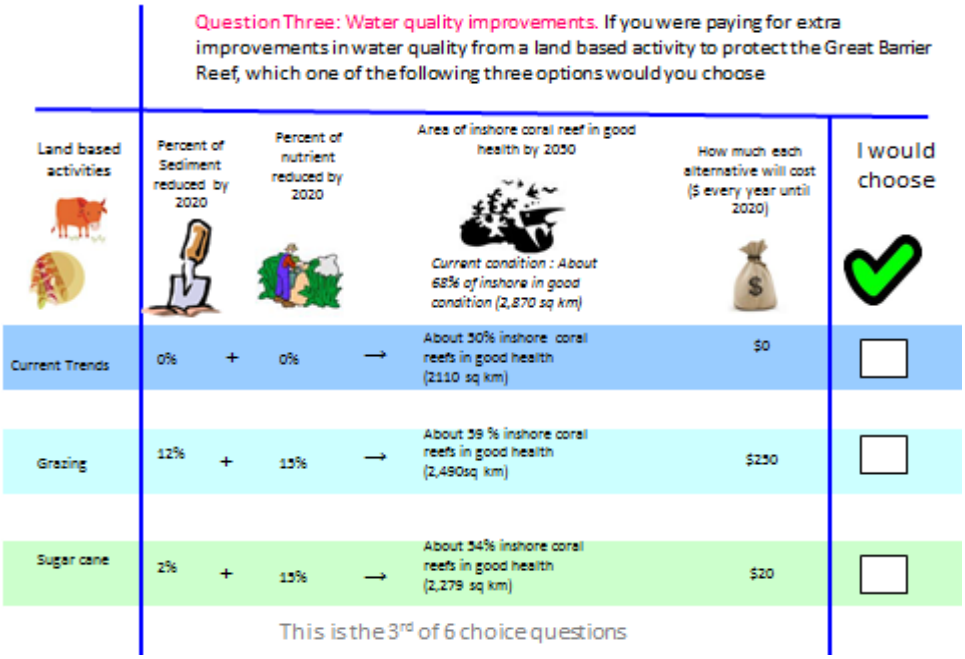
Table 8.2 Levels in the two experiment blocks

	% of inshore reef in good health by 2050	**Cost every year until 2020**
Status Quo	50	0
0	53, 54, 56, 59, 62	20, 25, 50, 100, 250
0	53, 54, 56, 59, 62	20, 25, 50, 100, 250

8.2.2 Splits

The survey was split into three different samples to achieve several comparisons between Brisbane and Coastal communities. The first split was between communities. A comparison was made between the preferences of Brisbane

residents and those of the coastal communities, which spanned Gladstone to Cairns, covering the three main cities of Townsville, Mackay and Rockhampton. The second split was between labelled and unlabelled experiments (see Figures 8.1 and 8.2). A comparison investigating the impact of the different industries such as sugarcane and grazing led to having labelled and unlabelled survey splits. The third split was to have a water quality path (between the attributes of sediment and nutrient) to understand if there was concern about how the improvement was achieved.

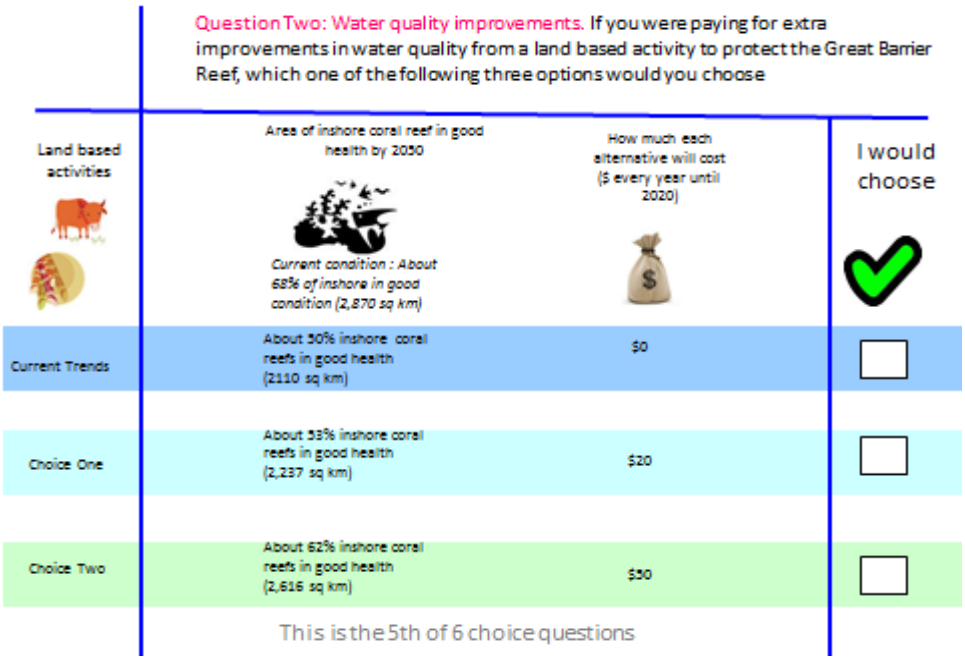


Question Three: Water quality improvements. If you were paying for extra improvements in water quality from a land based activity to protect the Great Barrier Reef, which one of the following three options would you choose

Land based activities	Percent of Sediment reduced by 2020		Percent of nutrient reduced by 2020		Area of inshore coral reef in good health by 2050 (Current condition : About 68% of inshore in good condition (2,870 sq km))	How much each alternative will cost ($ every year until 2020)	I would choose
Current Trends	0%	+	0%	→	About 50% inshore coral reefs in good health (2110 sq km)	$0	☐
Grazing	12%	+	15%	→	About 59 % inshore coral reefs in good health (2,490sq km)	$250	☐
Sugar cane	2%	+	15%	→	About 54% inshore coral reefs in good health (2,279 sq km)	$20	☐

This is the 3rd of 6 choice questions

Figure 8.1 Split with labels and water quality path (sediment and nutrient reductions)



Question Two: Water quality improvements. If you were paying for extra improvements in water quality from a land based activity to protect the Great Barrier Reef, which one of the following three options would you choose

Land based activities	Area of inshore coral reef in good health by 2050 (Current condition : About 68% of inshore in good condition (2,870 sq km))	How much each alternative will cost ($ every year until 2020)	I would choose
Current Trends	About 50% inshore coral reefs in good health (2110 sq km)	$0	☐
Choice One	About 53% inshore coral reefs in good health (2,237 sq km)	$20	☐
Choice Two	About 62% inshore coral reefs in good health (2,616 sq km)	$50	☐

This is the 5th of 6 choice questions

Figure 8.2 Split with no labels, water quality path and cost

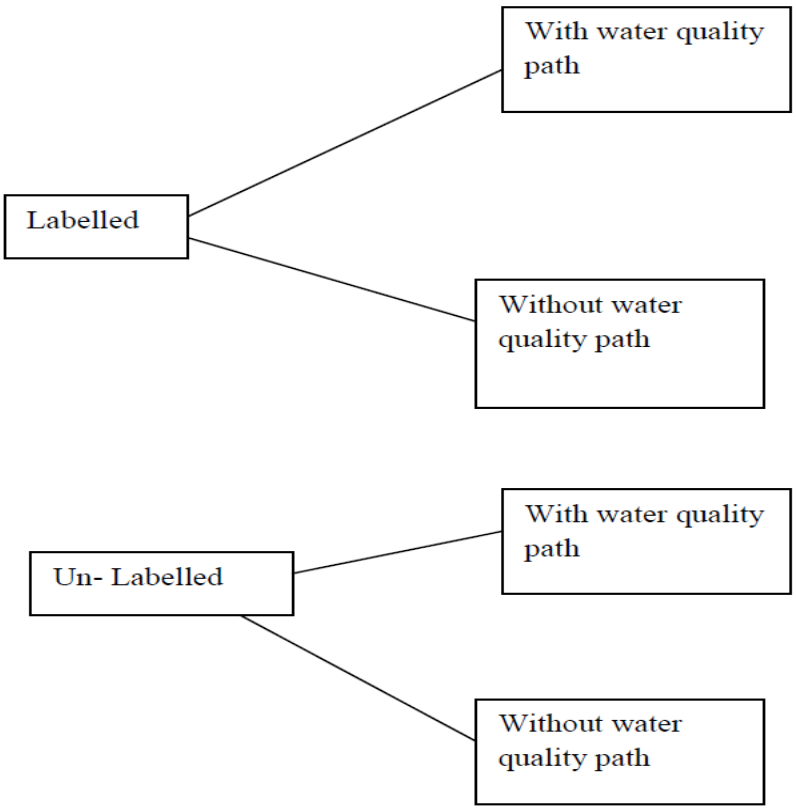


Figure 8.3 The splits provided to both coastal and Brisbane respondents

8.3 Experimental design

The experimental design of the choice sets was determined in several stages. The four sample splits were based on efficient designs. All designs were based on a fractional-factorial main effects design. This section describes the choice sets, blocks and levels.

8.3.1 Number of choice sets

Choosing the optimal number of choice sets for the survey depended on two particular aspects: cognitive burden for the respondents and finding a number suitable to create efficient experimental designs. This experiment involves a highly complex public good, and the questionnaire is quite lengthy given the large amounts of information initially required to frame the issue. Therefore, six

choice sets in the questionnaire was considered reasonable. This was discussed with participants in the initial two focus groups to confirm the optimal amount to consider without fatigue.

It was apparent that a blocking design would be required to achieve an efficient design. This also resulted in maintaining attribute-level balance, given that they are all multiples of the number of levels of the ecological attributes.

8.4 Questionnaire pre-testing

Pre-testing of the questionnaire occurred in two stages. Initially, a focus group was organised in Rockhampton where a different selection of attributes, formats and questions were tested. The focus groups involved eight people living in the Rockhampton region, with a mix of gender, age and income levels. The focus group provided comprehensive feedback on design issues, formatting and general questions. After this focus group, changes were made to the survey and a second focus group was held in Brisbane, with a similar mix of participants. Again, feedback was given and slight modifications made to the survey.

8.5 Questionnaire structure

The questionnaire consisted of three main sections: (1) a set of information and questions relating to the GBR and water quality improvements; (2) the choice sets; and (3) the generic socio-demographic questions. The complete questionnaire is included in Appendix D.

8.5.1 General Great Barrier Reef questions

The first questions asked in each section were aimed at generating an understanding of the respondents' knowledge and experience of the GBR. It was also to get them involved in the questionnaire at an early stage and to gain an understanding about whether the information that followed in the survey was the only influence on their decision-making. Specific questions included:

- how they would describe what has happened to the health of the GBR over the past ten years; response options were 'declined health', 'improved health', 'stayed the same' or 'don't know'.

- the factors they believe cause the greatest adverse pressure on the GBR; they were asked to rank the options of ‘climate change’, ‘over-fishing’, ‘nutrient run-off’ and ‘sediment run-off’.
- their motivation to improve the health of the GBR; they could selection one option from ‘to maintain recreational fishing’, ‘to ensure use for future generations’, ‘to maintain the tourism industry’ and to ‘visit it myself’.

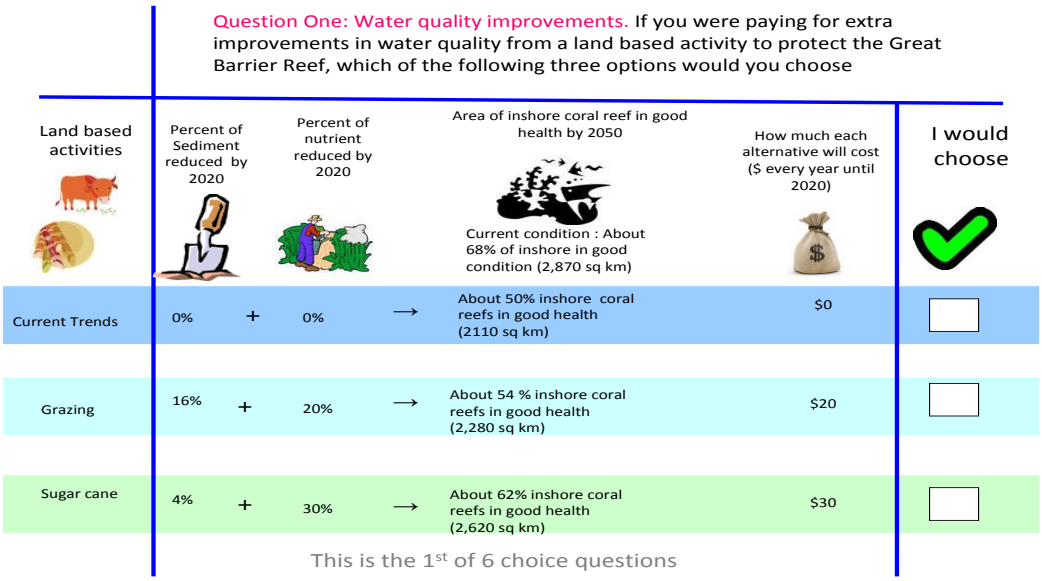
8.5.2 Information

Attribute information was presented before the choice sets and included information on the two different industries of grazing and sugarcane, how much sediment and nutrient they contribute to the GBR, what the targets of *Reef Plan* (2009) are, different management practices to reduce these land-based pollutants and the payment vehicle. It was also brought to the respondents’ attention that there are other World Heritage Areas in Australia and other environmental issues that may require further funding. Finally, instructions on completing the survey were given to the participants.

Information describing and defining the attributes was presented before the choice sets. The information discussed the land-based activity, addressing the problem, areas of improvement, economic importance of land based industries and tourism, and the spatial scale of the catchments and land based activities. The information then focused on the payment vehicle, budget constraints and an example choice set. Six choice sets then followed.

8.5.3 Choice sets

The choice sets were designed using graphics to allow a visual representation of the trade-offs to be considered. Respondents could also see where in the series of choice sets they were to maintain concentration and avoid fatigue. An example of a choice set is illustrated in Figure 8.4.



Question One: Water quality improvements. If you were paying for extra improvements in water quality from a land based activity to protect the Great Barrier Reef, which of the following three options would you choose

Land based activities	Percent of Sediment reduced by 2020		Percent of nutrient reduced by 2020		Area of inshore coral reef in good health by 2050 Current condition : About 68% of inshore in good condition (2,870 sq km)	How much each alternative will cost ($ every year until 2020)	I would choose
Current Trends	0%	+	0%	→	About 50% inshore coral reefs in good health (2110 sq km)	$0	☐
Grazing	16%	+	20%	→	About 54 % inshore coral reefs in good health (2,280 sq km)	$20	☐
Sugar cane	4%	+	30%	→	About 62% inshore coral reefs in good health (2,620 sq km)	$30	☐

This is the 1st of 6 choice questions

Figure 8.4 Format for the choice sets, and example of a labelled choice set with the attributes of sediment reduced, nutrient reduced, reef health and cost

8.5.4 Follow-up questions

Several de-briefing questions followed the choice sets to investigate respondent uncertainty and decision heuristics. To begin with, if the respondent always chose the status quo option, they were asked why. All respondents were then asked to do the following:

- To rank from 1 (strongly agree) to 5 (strongly disagree) the following statements:
    - I am confident that I made the correct choices
    - I understood the information in the questionnaire
    - I needed more information than was provided
    - I found the choice options to be credible
    - I found the choice options confusing
    - Cost was not important in the choices I made
- To rank from 1 (not important) to 5 (very important) the importance of the following management practices in the sugar and grazing industries for improved water quality outcomes:
    - Exclusion of stock from waterways
    - Improved application of nutrients and pesticides
    - Decreased bare ground in grazing lands

    - Reduced application of nutrient and pesticides
    - Excluding stock from an area of the property for the wet season
    - Improved timing of nutrient and pesticides to avoid application before a rainfall event
    - Improved management of gullies which are contributing sediment in grazing lands
- To rank from 1 (not important) to 5 (very important) the various components of the GBR which they thought were important:
    - Coral reefs
    - Fish
    - Seagrass
    - Marine turtles
    - Dugongs and dolphins
    - Sea birds

8.5.5 Socio-demographic questions

Once the questions were completed, respondents were asked about a series of general questions related to their confidence in government departments to impose conservation measures, and whether in previous conservation and development issues they have tended to favour one or the other. Respondents were also asked their prior knowledge of the issues raised in the survey.

Generic questions recording respondents' age, gender, income bracket, employment or status and industry, or whether they had any children were also asked in this section. Postcode and gender were asked once logged into the survey to ensure that there was the required sample size in coastal populations as well as Brisbane.

8.6 Sampling procedure and general statistics

A market research company was engaged to administer the survey using an internet database. The company emailed the survey to their collection panel but did not make mention of the survey topic area. This was done to minimise bias with respondents self-selecting. Respondents were offered a small cash incentive to complete the survey in compensation for their time. Sampling began at the start of December 2011 and was completed one month later. The survey was

collected from both coastal regional towns and cities in the GBR catchment and from Brisbane. This was done to explore the effects of scope and scale on the responses.

The sample and population characteristics (Table 8.2) generally reflected the current Queensland population, however there was a higher proportion of females, and a lower proportion of those with a postschool and tertiary education than generally exists in the population.

Table 8.2 Sample and population characteristics

	Coastal Sample	Coastal Population	Brisbane Sample	Brisbane Population
Gender				
Female	58%*	50%	58%*	50%
Age				
Median (category / years)	36-45	35	46-60	34
Education				
Postschool qualification	31%*	59%	31%*	59%
Tertiary degree	22%*	25%	30%*	25%
Income (gross)				
Less than $25,999 per year	22%	14%	21%	17%
$26,000 – $41,599 per year	23%	18%	22%	18%
$41,600 – $62,399 per year	21%	30%	21%	21%
$62,400 – $103,999 per year	21%	30%	24%	25%
$103,999 or more per year	13%*	14%	12%*	22%

Australian Bureau of Statistics 2011 Census. *Indicates a statistical difference between the sample and the population when applying: b = the normal approximation to the bionomial test.

8.9 Summary

This chapter has provided the methods for which the CM experiment was conducted. The complexity in defining attributes through focus groups and ensuring that the levels were balanced and justified were key aspects in designing the experiment. The results are explained in Chapter Nine.

CHAPTER NINE: CHOICE MODELLING RESULTS

In this chapter the results of the CM experiments are presented and the analysis applied to address each of the relevant hypotheses. First, the hypotheses and the relevant samples will be explained, followed by a review of the general statistics. Following this, each hypothesis will be addressed in detail, describing the utility function, the variables explaining the status quo, the model and model fit, WTP, (Poe *et al.* 2005) test and the results of the log likelihood ratio test. The results of the follow-up questions will then be explored to determine if and how complexity impacted on respondents' choices. Following this, the WTP across the state will be estimated and the results of the hypotheses will be discussed.

9.1 Hypothesis testing

To understand the community benefits of improved water quality, three hypotheses were developed;

- H_1: Coastal populations have a higher willingness to pay for improvements in GBR health than the Brisbane population.

  To test this hypothesis, the model for 'unlabelled with water quality path of sediment and nutrient' split sample experiment was compared between the coastal and Brisbane populations.

- H_2: Water quality path (sediment and nutrient reductions) impacts on people's willingness to pay for improvements in GBR health.

  To test this hypothesis, models for the 'labelled with water quality path' and the 'labelled no water quality path' split sample experiments were compared.

- H_3: Participants consider the labels (sugarcane and grazing) when selecting improvements in Reef health.

To test this hypothesis, models for the labelled and unlabelled split sample experiments with and without a water quality path of sediment and nutrient were compared. To understand the impact of labels and water quality path, the sample data for the two population data sets were pooled to respectively differentiate between the water quality path or the labels.

Table 9.1 Hypotheses testing with the following population and sample splits

	Brisbane population split	**Coastal population split**	**Pooled population**
Hypothesis 1	Unlabelled with water quality path	Unlabelled with water quality path	
Hypothesis 2			Labelled with water quality path Labelled with no water quality path
Hypothesis 3			Unlabelled with water quality path Labelled with water quality path

The results of these models will be presented in subsequent sections.

Some general observations are noted below with respect to the response patterns across the split-sample choice experiment. The choice frequencies across the sample appear not to favour the status quo, but between samples there are similar frequencies of choices for the various alternatives.

Table 9.2 Choice frequencies across alternatives

Choice frequencies	**status quo (%)**	**Alternative 1 (%)**	**Alternative 2 (%)**
Labelled with water quality path	21.2	41.9	36.8
Labelled no water quality path	23.1	40.3	36.6
Unlabelled with water quality path	19.8	45.7	34.4

The percentage of respondents who always selected the status quo was similar for the three samples: approximately 10 per cent of respondents in each sample, for each of the six choices.

Table 9.3 Percentage of respondents who always selected the status quo

Percentage (%) of respondents who always selected the status quo	
Labelled with water quality path	10.74
Labelled no water quality path	10.96
Unlabelled with water quality path	10.10

The labelled sample had a lower percentage (60% and 56% respectively) of respondents who never selected the status quo. The unlabelled sample had the highest percentage of respondents who never selected the status quo, potentially indicating that the labels influenced choices of respondents.

Table 9.4 Percentage of respondents who never selected the status quo

Percentage (%) of respondents who never selected the status quo	
Labelled with water quality path	60
Labelled no water quality path	56
Unlabelled with water quality path	63

9.2 Model form

Mixed logit (random parameter) models were developed for each of the split-sample experiments to take into account the panel nature of the data and the heterogeneity between respondents, as well as to avoid IIA/IID restrictions. Error component models were also developed for this purpose. A normal functional form was utilised for the randomised attributes after testing other forms, and 1,000 halton draws were used for the RPL models analysed. The five main socio-demographic variables were included in all models, even if they were not significant, and were modelled to explain the choice of the base or status quo alternative. In the samples with a water quality path (sediment and nutrient attributes), it was necessary to create a simple attribute to account for the perfect correlation between these two variables. A combined parameter termed Sednut

was created by multiplying the levels of the two attributes together (further information on this can be found in Chapter Seven). Details of the model variables are explained in Table 9.5.

Table 9.5 Variable explaining the status quo choice

Main variables	Description
ASC	Alternative specific constant
Reef health	Improvements in GBR health
Cost	Cost for a 1 per cent improvement in GBR health
Sednut	Sediment and nutrient pollutant reductions
Age	Age in years
Gender	Female = 1; male = 2
Children	Children = 1; no children = 2
Education	Coded from 1= primary to 5 = tertiary degree or higher
Income	Data were collected in a five-category format for gross weekly income. The data were converted to a single variable with the following mid-points applied to the income categories: $259, $650, $1,000, $2,000

Three tests for each hypothesis were performed, with each testing the difference in models between two samples. Firstly, part-worths for WTP and the confidence intervals for each sample were calculated. Part-worth tests can be used to identify where there is significant difference in WTP for particular attributes. These were then checked for overlapping confidence intervals, with no overlap indicating that there is not a significant difference between the two samples.

The WTP estimates for a 1 per cent improvement in GBR health were calculated as follows:

$$WTP = -1*\beta1(\text{Reef Health})/\beta2(\text{Cost}) \qquad \text{(Equation 27)}$$

Secondly, a Poe *et al.* (2005) test was used to test the difference between the two samples. This involves estimating the 95% confidence intervals using the Krinsky-Robb procedure, a parametric bootstrapping method, to draw a vector of 1,000 sets of parameters for each model, and differences calculated by taking one vector from another. Following Poe *et al.* (2005), this process is repeated 100

times by randomly re-ordering one vector of parameters. The 95 per cent confidence interval is approximated by identifying the part-worth differences that were less than zero.

Finally, a likelihood ratio test was performed to identify equivalence of parameter vectors. The ratio was calculated as follows:

Log Likelihood ratio= $-2 \times [LogL_{ab} - (LogL_a + LogL_b)]$ (Equation 28)

Where the $LogL_{ab}$ is the log likelihood value attached to the MNL model of the pooled dataset and the log likelihoods $LogL_a$ and $LogL_b$ relate to individual datasets. The resulting likelihood ratio statistic follows an asymptotic chi-square distribution with (P+1) degrees of freedom, where P is the number of parameters across the models involved (Rolfe *et al.* 2000).

9.3 H_1: Coastal populations have a higher willingness to pay for improvements in GBR health than the Brisbane population

The utility function was calculated as below for both the Brisbane and the coastal samples, with *Reef health* the only variable randomised and the five main socio-demographic variables included in the models. Only the unlabelled split-sample experiments were compared to understand the impacts of the water quality path (*sednut*) and the population impacts.

Unlabelled sample utility function:

$U_{(\text{Status Quo})}$ = ASC + β1(Reef health) + β2(Sednut) + β3(Cost) + β4(Education) + β5(Income) + β6(Age) + β7Children) + β8(Gender)

$U_{(\text{Alt 1})}$ = β1(Reef Health)+ β2(Sednut) + β3(Cost)

$U_{(\text{Alt 2})}$ = β1(Reef Health) + β2(Sednut) + β3(Cost)

The coastal model had a lower chi-square, as the sample size at 696 was less than half the Brisbane sample size (1620). Both higher levels of *Reef health* and lower levels of *Cost* were consistently preferred across both models. Lower levels of pollutants were also preferred across both models. Both models are relatively

well fitting with relatively high McFadden pseudo R^{2-}values and low Akaike Information Criterion (AIC) values.

Table 9.6 Mixed logit model for the Brisbane and coastal unlabelled with water quality path samples

	Brisbane coefficients	**SE**	**Coastal coefficients**	**SE**
Random parameters in utility functions				
Reef health	0.581***	0.149	1.011***	0.239
Non-random parameters in utility functions				
ASC	-0.384	0.660	-1.706	1.163
Cost	-0.008***	0.000	-0.007***	0.000
Sednut	-0.166***	0.050	-0.359***	0.080
Education	-0.183*	0.097	-0.466***	0.169
Income	0.26100d-04	0.000	0.000	0.000
Age	-0.017	0.083	0.424**	0.188
Gender	-0.177	0.251	0.095	0.437
Children	-0.271	0.224	-0.477	0.429
Model statistics				
Observations	1620		696	
Log L	-1333		-568	
AIC	1.658		1.661	
McFadden R^2	0.251		0.257	
Chi-squared (D.o.F = 9)	8925		392	

*Note. ***=p<0.01, **p<0.05, *p<0.10*

The socio-demographic variables had some influence on the selection of the status quo option by respondents. *Education* was the only significant variable in both models, with *Age* also significant in the coastal sample. *Income* was not significant in either of the models, suggesting that some respondents did not fully consider their budgetary limitations and indicating that there may be some unexplained or unobserved reasons underlying the respondents' choice selection. Neither model had a significant ASC, indicating that there were no unobserved factors impacting the choices made.

The mean WTP part-worth estimates for a 1 per cent improvement in Reef health was estimated for the Brisbane sample at $73.79 and for the coastal population at

$147.40. The coastal population had a larger range ($62.36– $295.38) than the Brisbane population ($32.09–$129.73). Given there is overlap between the two samples, the hypothesis is rejected for this particular test.

Table 9.7 Part-worths for a 1 per cent improvement in Reef health

	Minimum	Mean	Maximum
Brisbane	$32.09	$73.79	$129.73
Coastal	$62.36	$147.40	$295.38

The Poe *et al.* (2005) test was also conducted on each attribute. The results of 0.93116 for *Reef health*, 0.8162 for *Cost* and 0.02363 for *Sednut* indicate that for *Cost* there is no significant difference (at the 5 per cent level) between the values held by the Brisbane and coastal sample, but there is significant difference for the water quality improvement path (*Sednut*).

Table 9.8 Results of the Poe *et al.* (2005) test

Reef health	ASC	Cost	Sednut
0.93116	0.16375	0.8162	0.02363

The key test for the population hypothesis is a likelihood ratio test. Here, the log-likelihood values for the individual models are compared to the log-likelihood values for the pooled models. The log likelihood of the pooled Brisbane coastal model is -1915.42; the test statistic is therefore:

=-2*(-1915.42-(-1333+-568))

=28.84

The appropriate chi-square with ten degrees of freedom is 16.92; therefore, the hypothesis that the models are equivalent must be rejected.

Follow-up questions were used to clarify the extent to which respondents were confident in their selection of choices, understanding and credibility of the

options. Although both samples indicated that they were confident in the choices, there was a small percentage (9 per cent for the Brisbane population and 10 per cent for the coastal population) who found the choice options confusing. The percentage of respondents who strongly agreed that they needed more information was 11 per cent for the Brisbane population and 16 per cent for the coastal population.

Table 9.9 Percentage of respondents rating confidence, credibility and understanding of their choices

Respondents: sample set and statements ranked	**1 (strongly agree) to 5 (strongly disagree) (%)**				
Brisbane sample	**1**	**2**	**3**	**4**	**5**
I am confident that I made the correct choices	23	37	18	16	6
I understood the information in the questionnaire	31	29	16	16	8
I needed more information than was provided	10	28	18	34	10
I found the choices to be credible	15	32	26	22	5
I found the choice options confusing	9	20	16	37	18
Cost was not important in the choices I made	15	24	10	33	18
Coastal sample					
I am confident that I made the correct choices	28	33	18	12	9
I understood the information in the questionnaire	33	31	14	17	5
I needed more information than was provided	16	26	18	23	17
I found the choices to be credible	16	34	19	24	7
I found the choice options confusing	10	25	13	30	22
Cost was not important in the choices I made	13	16	12	41	18

Another follow-up question asked respondents to rank the importance of a number of management practices. Respondents rated the majority of practices as important to very important (rankings 3 to 5), but, interestingly, the management practices of excluding stock from water ways and excluding stock from an area of the property for the wet season had a higher distribution of not important to important (rankings 1 to 3)

Table 9.10 Percentage of respondents ranking importance of management practices

Respondents: sample set and statements rated	**1 (not important) to 5 (very important) (%)**				
Brisbane sample	**1**	**2**	**3**	**4**	**5**
Exclusion of stock from waterways	3	13	15	50	19
Improved application of nutrients and pesticides	1	4	4	45	46
Decreased bare ground in grazing lands	1	4	8	42	45
Reduced application of nutrients and pesticides	1	3	5	42	49
Excluding stock from an area of the property for the wet season	4	16	20	37	23
Improved timing of nutrient and pesticides to avoid application before a rainfall event	1	4	8	41	46
Improved management of gullies which are contributing sediment in grazing lands	1	5	9	44	41
Coastal sample					
Exclusion of stock from waterways	5	14	18	47	16
Improved application of nutrients and pesticides	2	3	0	43	52
Decreased bare ground in grazing lands	3	3	5	49	40
Reduced application of nutrients and pesticides	2	4	5	39	50
Excluding stock from an area of the property for the wet season	6	15	16	45	18
Improved timing of nutrient and pesticides to avoid application before a rainfall event	1	4	5	44	46
Improved management of gullies which are contributing sediment in grazing lands	1	3	6	53	37

A third set of follow-up questions asked respondents to rank the components of the GBR they thought important. Responses were dominated by very high and high rankings (Figure 9.1), with coral reefs having the highest percentage of very important ranking and seabirds having the lowest percentage of respondents selecting it as very important.

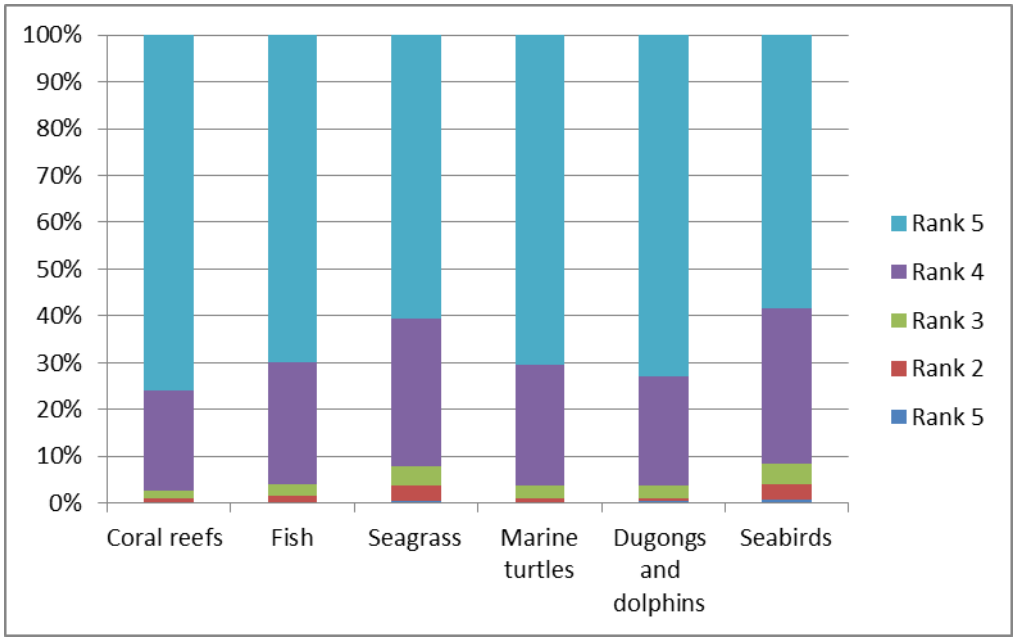


Figure 9.1 Brisbane ranking of the importance of marine life

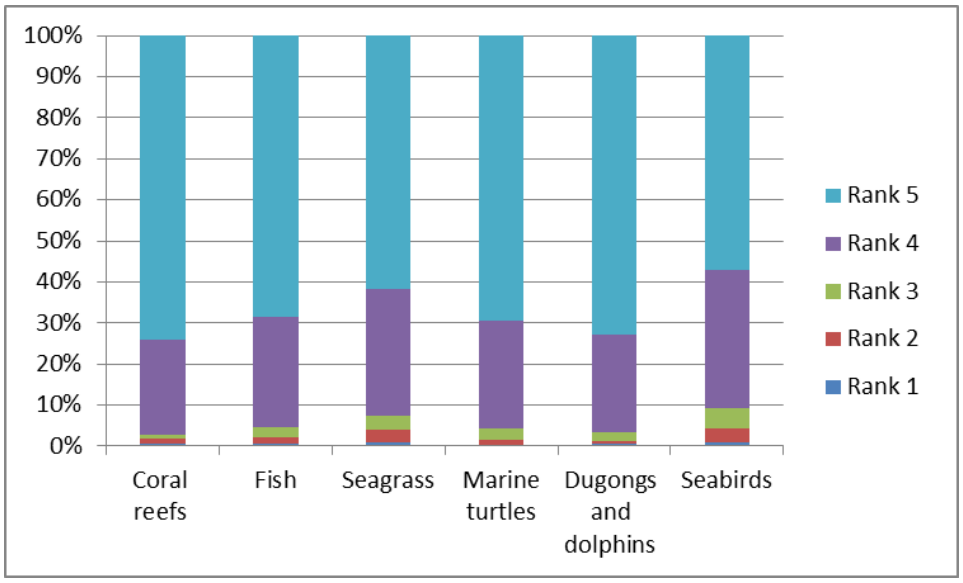


Figure 9.2 Coastal ranking of the importance of marine life

The Mann–Whitney U test was conducted on the results of these follow-up questions to determine if there was a significant difference in the distribution of rankings for the two samples at a 5 per cent level of significance. For the Brisbane and the coastal populations at 5 per cent level of significance, there was no statistical difference between the two populations (Table 9.11).

Table 9.11 Mann–Whitney U test results for coastal and Brisbane populations, follow-up questions on confidence, confidence and understanding

	Median		Percentiles							
			25	50	75	25	50	75	Mann–Whitney	
	Coastal	Brisbane	Coastal			Brisbane			U test	P-value
I am confident that I made the correct choices	2	2	1	2	3	2	2	3	16,065	0.676
I understood the information in the questionnaire	2	2	1	2	3	1	2	3	16,388	0.454
I needed more information than was provided	3	3	2	3	4	2	3	4	16,224	0.563
I found the choice options to be credible	2	3	2	2	4	2	3	4	15,764	0.914
I found the choice options confusing	4	4	2	4	4	2	4	4	16,097	0.696
Cost was not important in the choices I made	4	4	2	4	4	2	4	4	14,482	0.226

The Mann–Whitney U test between samples for management practices did not result in any significant difference in rankings. The high medians (predominately 4, 5) for the two samples indicate the relatively high importance which both populations place on management practices for water quality improvements.

There was also no difference between the two populations for the importance of marine life at a 5 per cent significance level. The median ranking was 5 for both populations, with very limited variation of other rankings as demonstrated by the percentiles for each group being 4 and above, or important and very important on the scale.

Table 9.12 Results of Mann–Whitney U test for the Brisbane and Coastal populations

	Median		Percentiles							
			25	50	75	25	50	75	Mann–Whitney	
	Coastal	Brisbane	Coastal			Brisbane			U test	P-value
Exclusion of stock from waterways	4	4	3	4	4	3	4	4	16,813	0.218
Improved application of nutrients and pesticides	5	4	4	5	5	4	4	5	14,542	0.215
Decreased bare ground in grazing lands	4	4	4	4	5	4	4	5	16,065	0.658
Reduced application of nutrient and pesticides	5	4	4	5	5	4	4	5	15,678	0.984
Excluding stock from an area of the property for the wet season	4	4	3	4	4	3	4	4	15,936	0.774
Improved timing of nutrient and pesticides to avoid application before a rainfall event	4	4	4	4	5	4	4	5	15,403	0.779
Improved management of gullies which are contributing sediment in grazing lands	4	4	4	4	5	4	4	5	15,754	0.918

Table 9.13 Results of Mann–Whitney U test for Brisbane and Coastal populations for the importance of marine life

	Median		Percentiles							
			25	50	75	25	50	75	Mann–Whitney U test	P-value
	Coastal	Brisbane	Coastal			Brisbane				
Coral reefs	5	5	4	5	5	5	5	5	16,633	0.204
Fish	5	5	4	5	5	4	5	5	16,567	0.267
Seagrass	5	5	4	5	5	4	5	5	15,152	0.556
Marine turtles	5	5	4	5	5	4	5	5	16,246	0.469
Dugongs and dolphins	5	5	4	5	5	4	5	5	15,701	0.958
Sea birds	5	5	4	5	5	4	5	5	16,563	0.308

9.4 H_2: Water quality path (sediment and nutrient reductions) impacts on people's willingness to pay for improvements in GBR health

To test this hypothesis the labelled with water quality path and the labelled with no water quality path splits were analysed with the population splits pooled.

Reef health was the only variable that was randomised to identify the level of support for improved *Reef health*, and the implications the pollutants had on the respondents' willingness to pay. The utility function for the status quo and the alternatives are shown as follows.

Labelled sample with water quality path utility function:

$U_{(\text{Status Quo})} = \beta1(\text{Reef health}) + \beta2(\text{Sednut}) + \beta3(\text{Cost}) + \beta4(\text{Education}) + \beta5(\text{Income}) + \beta6(\text{Age}) + \beta7\text{Children}) + \beta8(\text{Gender})$

$U_{(\text{Alt 1})} = \text{ASC}_{\text{grazing}} + \beta1(\text{Reef health}) + \beta2(\text{Sednut}) + \beta3(\text{Cost})$

$U_{(\text{Alt 2})} = \text{ASC}_{\text{sugarcane}} + \beta1(\text{Reef health}) + \beta2(\text{Sednut}) + \beta3(\text{Cost})$

Labelled sample with no water quality path utility function:

$U_{(\text{Status Quo})} = \beta1(\text{Reef health}) + \beta2(\text{Cost}) + \beta3(\text{Education}) + \beta4(\text{Income}) + \beta5(\text{Age}) + \beta6(\text{Children}) + \beta7(\text{Gender})$

$U_{(\text{Alt 1})} = \text{ASC}_{\text{grazing}} + \beta1(\text{Reef health}) + \beta3(\text{Cost})$

$U_{(\text{Alt 2})} = \text{ASC}_{\text{sugarcane}} + \beta1(\text{Reef health}) + \beta3(\text{Cost})$

The model results are presented in Table 9.14. The models for both the samples are significant (high chi-squared values) and the *Reef health* and *Cost* attributes were significant and signed as expected. The labels were only significant for the sample with the water quality path, with higher levels of reductions from grazing ($\text{ASC}_{\text{grazing}}$) preferred to reductions from sugarcane ($\text{ASC}_{\text{sugarcane}}$). Both models are relatively well fitting with relatively high McFadden pseudo R^2 square values

(0.2491 and 0.1916 with water quality path and with no water quality path respectively) and low Akaike Information Criterion (AIC) values (1.651 and 1.781 with water quality path and with no water quality path respectively). The socio-demographic variables had some influence on choice. Income was significant in both samples, indicating that respondents considered their budgetary limitations.

Table 9.14 Mixed logit models for labelled with water quality path and labelled with no water quality path

	Labelled with water quality path		**Labelled with no water quality path**	
	Coefficient	**S.E**	**Coefficient**	**S.E**
Random parameters in utility functions				
Reef health	0.099***	0.022	0.060***	0.179
Non-random parameters in utility functions				
$ASC_{grazing}$	1.347**	0.525	0.370	0.490
$ASC_{sugarcane}$	1.197*	0.515	0.412	0.488
Cost	-0.008***	0.000	-0.006***	0.000
Sednut	-0.018***	0.006		
Education	-0.189**	0.074	0.056	0.061
Income	0.141**	0.070	-0.000***	0.000
Age	-0.207***	0.067	0.103*	0.061
Gender	0.400**	0.176	-0.117	0.186
Children	0.174	0.183	-0.450***	0.157
Model statistics				
Observations	2346		2244	
Log L	-1935.25		-1988.60	
AIC	1.659		1.781	
McFadden R^2	0.2491		0.1916	
Chi-squared (D. of F. = 100)	1284.18		953	

*Note. ***=p<0.01, **p<0.05, *p<0.10*

The range of WTP estimated for the no water quality path sample was $3.56 - $16.96 and for the water quality path sample was $6.54 - $20.25. Therefore overlap was identified between the two samples and the hypothesis of a difference in WTP was rejected (Table 9.15).

Table 9.15 Part-worths for a 1 per cent improvement in Reef health

	Minimum	Mean	Maximum
Water quality path	$6.54	$17.53	$20.25
No water quality path	$3.56	$9.54	$16.96

The results of the Poe *et al*. (2005) test indicate that there is a significant difference between *Reef health* in the two models indicating that water quality path does impact the WTP. There was no significant difference for the *Cost* attribute. The results indicate that the models are equivalent in the areas where the case studies were similar, but vary when the attributes have very different levels in the different samples.

Table 9.16 Poe et al. (2005) test Reef health and cost

Reef health	Cost
0.03839	0.54195

A log-likelihood ratio test was also undertaken to determine if the samples were independent:

=-2(-3953.43-(-1935.25+-1988.6))

=59.17

The appropriate chi-square with ten degrees of freedom is 18.31, therefore the hypothesis that the models are equivalent must be rejected.

Follow-up questions were used to clarify the extent to which respondents focused on the inputs, output or both. In addition, it is important to understand the complexity of how the valuation scenario affected the respondents' behaviour and to assess the potential impact on the WTP estimates.

Confidence in making choices and understanding the choices was the focus of one of the follow-up questions. Respondents were asked to rank statements from 1 (strongly agree) to 5 (strongly disagree). There was no significant difference

between the two samples for any of the follow-up questions (Appendix E). Respondents were also presented with a range of management options and asked to rank them relative to their importance for improved water quality, with 1 indicating that they did not believe it to be important and 5 indicating it was very important.

There was a definite preference for a ranking of four across all the management practices for both samples, with the water quality path sample having a higher amount of management practices ranked as very important than the no water quality path sample. This indicated that the management practices were ranked important to very important for the majority of respondents. There was an even stronger preference to rank components of the reef highly, with 93 per cent of respondents across both the samples ranking them as high or very high importance (4 and 5). This is opposed to 80 per cent of respondents for the no water quality path (Appendix E) ranking management practices high to very high and 79 per cent of respondents with water quality path.

All components of the reef were ranked high to very high, with coral reefs having the highest percentage of respondents ranking it very high in both samples. The water quality path sample next ranked dugongs and dolphins, then marine turtles; the no water quality path sample ranked dugongs and dolphins and then fish (Appendix E).

The distribution of rankings was significant between the two samples for the "I needed more information than was provided" statement. The Mann-Whitney U test was 17,137 and the resulting p-value was 0.042. The Mann-Whitney U test demonstrated that for all other remaining follow-up questions there was no significant difference across the two samples in their ranking of importance.

Table 9.17 Results of Mann–Whitney U test for the labelled with water quality path and without water quality path.

	Median		Percentiles			Percentiles				
			25	50	75	25	50	75		
	Water quality path	No water quality path	Water quality path			No water quality path			Mann-Whitney U test	P-value
Questions about confidence, credibility and understanding										
I am confident that I made the correct choices	2	2	2	2	3	2	2	3	76,213	0.294
I understood the information in the questionnaire	2	2	2	2	3	1	2	4	74,861	0.554
I needed more information than was provided	3	3	2	3	4	2	3	4	79,137	0.042
I found the choice options to be credible	3	3	2	3	4	2	3	4	72,748	0.873
I found the choice options confusing	4	3	2	4	4	2	3	4	71,551	0.597
Cost was not important in the choices I made	4	4	2	4	4	2	4	4	75,408	0.441
Questions about management practices										
Exclusion of stock from waterways	4	4	3	4	4	3	4	4	73,202	0.976
Improved application of nutrients and pesticides	5	4	4	5	5	4	4	5	70,097	0.272

	Median		Percentiles			Percentiles				
			25	50	75	25	50	75		
	Water quality path	**No water quality path**	**Water quality path**			**No water quality path**			**Mann-Whitney U test**	**P-value**
Decreased bare ground in grazing lands	4	4	4	4	5	4	4	5	71,723	0.622
Reduced application of nutrient and pesticides	4	4	4	4	5	4	4	5	74,207	0.696
Excluding stock from an area of the property for the wet season	4	4	3	4	4	3	4	4	77,639	0.121
Improved timing of nutrient and pesticides to avoid application before a rainfall event	4	4	4	4	5	4	4	5	71,825	0.643
Improved management of gullies which are contributing sediment in grazing lands	4	4	4	4	5	4	4	5	71,601	0.589
Questions about marine life										
Coral reefs	5	5	5	5	5	4	5	5	72,011	0.629
Fish	5	5	4	5	5	4	5	5	72,652	0.854
Seagrass	5	5	4	5	5	4	5	5	70,291	0.296
Marine turtles	5	5	4	5	5	4	5	5	70,798	0.353
Dugongs and dolphins	5	5	4	5	5	4	5	5	72,483	0.795
Sea birds	5	5	4	5	5	4	5	5	71,116	0.465

9.5 H_3: Participants consider the labels of Grazing and Sugarcane when selecting improvements in Reef health

To test this hypothesis, two samples – one with labels of grazing and sugarcane and the other with option one and option two being the descriptors used – were modelled and the difference between them tested. The utility functions for the two samples were as follows:

Labelled sample utility function:

$U_{(\text{Status Quo})} = \beta1(\text{Reef health}) + \beta2(\text{Sednut}) + \beta3(\text{Cost}) + \beta4(\text{Education}) + \beta5(\text{Income}) + \beta6(\text{Age}) + \beta7\text{Children}) + \beta8(\text{Gender})$

$U_{(\text{Alt 1})} = \text{ASC}_{\text{grazing}} + \beta1(\text{Reef health}) + \beta2(\text{Sednut}) + \beta3(\text{Cost})$

$U_{(\text{Alt 2})} = \text{ASC}_{\text{sugarcane}} + \beta1(\text{Reef health}) + \beta2(\text{Sednut}) + \beta3(\text{Cost})$

Unlabelled sample utility function:

$U_{(\text{Status Quo})} = \text{ASC} + \beta1(\text{Reef health}) + \beta2(\text{Sednut}) + \beta3(\text{Cost}) + \beta4(\text{Education}) + \beta5(\text{Income}) + \beta6(\text{Age}) + \beta7\text{Children}) + \beta8(\text{Gender})$

$U_{(\text{Alt 1})} = \text{ASC1} + \beta1(\text{Reef health}) + \beta2(\text{Sednut}) + \beta3(\text{Cost})$

$U_{(\text{Alt 2})} = \text{ASC2} + \beta1(\text{Reef health}) + \beta2(\text{Sednut}) + \beta3(\text{Cost})$

The models are presented in Table 9.18. Both models had high explanatory power and were significant. The *Cost* and *Reef health* attributes are significant and signed as expected. Both higher levels of *Reef health* and lower levels of *Cost* are preferred consistently across both models. The McFadden R^2 values of 0.2474 and 0.2472 and low Akaike Information Criterion (AIC) of 1.659 and 1.663 for labelled and unlabelled respectively indicate that both models are relatively well fitting.

The socio-demographic variables of *Education* and *Children* were significant for the unlabelled sample and *Income, Age, Gender* and *Education* were significant for the labelled sample. This suggests that some respondents did not fully consider their

budgetary limitations in the unlabelled sample, perhaps indicating heuristics in the choice process. The labelled sample resulted in both labels being significant, indicating that respondents did consider the labels in selecting their choices. The label for grazing was also slightly higher indicating that there is a greater preference for reductions to come from the grazing industry than the sugarcane industry.

Table 9.18 Mixed logit models for the labelled and unlabelled sample

	Labelled		**Unlabelled**	
	Coefficient	**S.E**	**Coefficient**	**S.E**
Random parameters in utility functions				
Reef health	0.099***	0.022	0.241***	0.062
Non-random parameters in utility functions				
$ASC_{grazing/1}$	1.349**	0.525	0.639	0.589
$ASC_{sugarcane/2}$	1.197**	0.515	0.184	0.587
Cost	-0.008***	0.000	-0.008***	0.005
Sednut	-0.018**	0.006	-0.006***	0.002
Education	-0.189**	0.074	-0.198**	0.081
Income	0.141**	0.070	0.78134D-04	0.000
Age	-0.207***	0.067	0.066	0.071
Gender	0.397**	0.174	-0.011	0.190
Children	0.174	0.183	-0.319	0.214
Model statistics				
Observations	2346		2316	
Log L	-1935		-1893	
AIC	1.659		1.645	
McFadden R^2	0.2474		0.2556	
Chi-squared (D. of F.= 11)	1284		1300	

*Note. ***=p<0.01, **p<0.05, *p<0.10*

The part-worths resulted in the unlabelled sample having the higher mean of $29.97 and larger range than the labelled sample. Given the overlap between the two samples, the hypothesis is rejected for the test.

Table 9.19 Part-worths for a 1 per cent improvement in Reef health

	Minimum	Mean	Maximum
Unlabelled	13.31	29.97	51.31
Labelled only water quality	6.54	17.53	20.25

The Poe *et al.* (2005) test indicated that there is a significant difference between the labelled and unlabelled samples for *Reef health* but not *Cost*. This indicates the hypothesis that the two samples participants consider the labels is accepted.

Table 9.20 Poe *et al.* (2005) test for Reef health and cost

Reef health	Cost
0.9999	0.2914

The log-likelihood test was also completed, with the pooled model having a log likelihood of -3,871.30. The test resulted in the follow equation:

=-2*(-3,871.30-(-1952.08+-1893))

=86.44

This indicates that the models are different given that the appropriate chi-squared statistic at 5 per cent significance is 19.68. This indicates that the labels do have a significant impact on the choice selected. Therefore the hypothesis that labels affect respondents' willingness to pay is accepted.

The follow-up questions again followed a similar trend to the other samples regarding confidence in making choices and understanding the questionnaire. The unlabelled sample did have a higher percentage than the labelled sample of respondents who strongly agreed that more information was needed (12 per cent as opposed to 5 per cent).

The results of the Mann–Whitney U test identified that there were only differences between the two samples for information (p-value 0.008) and for the ranking of seagrass (p-value 0.03). All other follow-up questions did not demonstrate a significant difference between the two sample populations.

Table 9.21 Results of the Mann–Whitney U test for the unlabelled and labelled results

	Median		Percentiles			Percentiles			Mann–Whitney U test	P-value
			25	50	75	25	50	75		
	Unlabelled	Labelled	Coastal			Brisbane				
Questions about confidence, credibility and understanding										
I am confident that I made the correct choices	2	2	2	2	3	2	2	3	71,943	0.245
I understood the information in the questionnaire	2	2	1	2	4	1	2	3	71,074	0.147
I needed more information than was provided	3	3	2	3	4	2	3	4	63,376	0.008
I found the choice options to be credible	3	3	2	3	4	2	3	4	74,932	0.860
I found the choice options confusing	3	4	2	3	4	2	4	4	80,744	0.082
Cost was not important in the choices I made	4	4	2	4	4	2	4	4	72,790	0.379
Questions about management practices										
Exclusion of stock from waterways	4	4	3	4	4	3	4	4	74,280	0.685
Improved application of nutrients and pesticides	4	4	4	4	5	4	4	5	77,523	0.463
Decreased bare ground in grazing lands	4	4	4	4	5	4	4	5	79,774	0.134
Reduced application of nutrient and pesticides	4	4	4	4	5	4	4	5	78,365	0.307
Excluding stock from an area of the property for the wet season	4	4	3	4	4	3	4	4	76,496	0.729
Improved timing of nutrient and pesticides to avoid application before a rainfall event	4	4	4	4	5	4	4	5	78,099	0.357
Improved management of gullies which are contributing sediment in grazing lands	4	4	4	4	5	4	4	5	76,538	0.707
Questions about marine life										
Coral reefs	5	5	4	5	5	4	5	5	75,050	0.862
Fish	5	5	4	5	5	4	5	5	76,717	0.625

Seagrass	5	5	4	5	5	4	5	5	81,200	0.036
Marine turtles	5	5	4	5	5	4	5	5	77,793	0.361
Dugongs and dolphins	5	5	4	5	5	4	5	5	77,847	0.334
Sea birds	5	5	4	5	5	4	5	5	79,266	0.173

9.6 Values for improvements in the health of the GBR

A major challenge for policymakers managing the GBR is to identify when measures to improve water quality through agricultural pollutant reductions generate net benefits to society. The results contribute to policymakers' understanding of the community benefits of improving GBR water quality to achieve targets both through agricultural reductions and overall improved Reef health. The results contribute to understanding the implications for WTP to achieve the *Reef Plan* targets.

This experiment explored the impacts of scale, attribute scope and the effects of labels. The experiment contributes in several different ways. Firstly, it used policy links or the *Reef Plan* targets for sediment and nutrient reductions to frame the experiment to value the benefits of subsequent improved Reef health. Secondly, it demonstrated how improvements to Reef health from water quality improvements are viewed consistently across populations. Thirdly, the information provided regarding the relevant agricultural industries helps policymakers improve management decisions.

The results indicate that there is no significant impact on respondents' WTP based on their location or distance from the GBR. This highlights the iconic nature of the asset and the importance of considering the value to the wider Queensland population. It supports previous research completed by Rolfe and Windle (2010) which indicates for such a large natural asset the values for populations further away do not decrease significantly.

In response to the second hypothesis respondents were more sensitive to the water quality path of pollutant reductions than to the labels of grazing and sugarcane. This may indicate that respondents are more sensitive to what the pollutants are rather than to the where they came from. The mean WTP results for pollutant reductions from the labelled results were higher than for the water quality path alone, indicating that respondents prefer knowing where the

pollutant reduction is coming from, rather than having no direct understanding of how the reductions will be achieved.

Table 9.22 Results of hypothesis testing

Hypothesis	Part-worth test	Poe *et al*. (2005) test	Log-likelihood test
H_1: Coastal populations have a higher willingness to pay for improvements in Reef health than the Brisbane population.	Reject	Reject	Reject
H_2: Water quality path (sediment and nutrient reductions) impact on people's willingness to pay for improvements in GBR health.	Reject	Accept	Reject
H_3: Participants consider the Land use labels (sugarcane and grazing) impact on people's willingness to pay for when selecting improvements in Reef GBR health.	Reject	Accept	Accept

There was no particular demographic aspect that was significant apart from income, which was not significant in the first hypothesis test, indicating that perhaps respondents did not fully consider their budgetary limitations or were using heuristics in the choice process; income was of little significance across the other samples.

Respondents' part-worth estimates calculated in previous sections have been focused on a 1 per cent improvement in Reef health, given that the status quo was identified as 50 per cent of the Reef in good health and a maximum of 70 per cent in Reef health set as the upper limit. A 1 per cent improvement is equivalent to 112,000 tonnes of sediment reduced. A 5 per cent improvement was also considered to allow a comparison between the level of sediment reduction that would be achieved. At 5 per cent improvement in water quality 560,000 tonnes of sediment would be reduced.

Present values of benefits for Queensland households were calculated to understand the value of achieving the *Reef Plan* sediment targets by 2020. The average WTP across all the labelled and unlabelled split samples was used to extrapolate to all households in the state. The net present values were estimated with annual WTP values calculated at 5, 8 and 12 per cent to allow for sensitivity to the discount rate. Given that respondents completed the survey in 2012, a time frame of eight years was calculated to account for the payment vehicle occuring every year until 2020; the average of the mean WTP was used to extrapolate to the broader population.

Two potential participation rates of 70 per cent and 90 per cent were used to extrapolate values from the sample to the relevant population based on a response rate of greater than 80 per cent in a similar paper-based version of the survey where accurate response rates were recorded (Rolfe and Windle 2011).

Table 9.23 Present values of willingness to pay per person with an eight-year time frame

		Discount rate (%)		
	Water quality pollutant reductions (tonnes)	**5%**	**8%**	**12%**
1% improvement in water quality ($)	Sediment 112,000	$232	$183	$602
5% improvement in water quality ($)	Sediment 560,000	$1,162	$915	$696

Table 9.24 Present values of willingness to pay for Queensland households with an eight-year time frame

	Discount rate (%)		
70% Qld households	5%	8%	12%
1% improvement in water quality ($M)	139	123	106
5% improvement in water quality ($M)	695	618	534
90% Qld households			
1% improvement in water quality ($M)	178	158	137
5% improvement in water quality ($M)	893	794	687

The results of the assessment of benefits indicate that the public benefit of improved Reef health by 1% (112,000 tonnes sediment) ranges between $106 million and $178 million. At a 5% level of improved Reef health (560,000 tonnes of sediment) the benefit ranges between $534 million and $893 million depending on the discount rate used.

9.7 Summary

This chapter has provided community benefit estimates of improved Reef health through pollutant reductions of sediment and nutrient from the grazing and sugarcane industries. The research has provided insights into the community benefits through further understanding the scale effects of valuations, labels and scope of attributes. The following chapter integrates the benefits and the costs to enable a policy framework for water quality pollutant reductions.

CHAPTER TEN: TRADE-OFFS AND IMPLICATIONS: COST-BENEFIT ANALYSIS

A key challenge for policymakers managing the GBR in Australia is to identify measures to improve water quality that generate net benefits to society. An important policy issue is determining if the public benefits of reducing emissions in agricultural run-off to improve the health of the GBR are sufficiently large to outweigh the costs involved. This aim of this thesis is to contribute to the knowledge gap through understanding the net social welfare of water quality improvements from sediment reductions in grazing lands in the Fitzroy and Burdekin basins. Integrating the estimated costs associated with sediment reductions from the bioeconomic modelling and community benefits estimated through choice modelling allows an assessment of the net benefits to society of measures to improve reef health.

Chapter Six highlighted the heterogeneity of costs for sediment reductions among different land types and conditions. Similarly, different grazing pressures and subsequent pasture utilisation has a significant impact on the cost of sediment reductions. This is a result of the pasture productivity and soil characteristics influencing profitability and the opportunity cost of changing grazing pressure. The level of profitability and sediment exported subsequently impacts on the cost per tonne of sediment reduced. Variation in the land types' inherent productivity and soil characteristics across different land types are also contributing factors to the level of resilience and rate of recovery from overgrazing.

The results of the bioeconomic modelling highlight the potential for different extension and incentive mechanisms that may be required to ensure cost-effective reductions can be achieved. The ability to ensure long-term change in

grazing practices occurs is also another aspect that must be considered. The heterogeneity that exists highlights the importance of considering a range of mechanisms, which reflect the public and private benefits and cost heterogeneity in the landscape. The results of Chapter Six highlight the importance of considering a number of factors before making investment decisions about policy mechanisms, and that for appropriate policymaking, the task of selecting the correct mix of policy mechanisms is very complex.

As discussed in Chapter Nine, the choice modelling experiment revealed that each 1% improvement in Reef health is valued between $52m and $201m to Queensland households depending on the discount rate and percentage of households holding values. The choice modelling results also revealed that the community valued the pollutant reductions differently when labels to identify industry water quality path to improved Reef health (i.e. the sediment and nutrient reductions would result in improved Reef health) were used.

This chapter will initially complete a cost benefit analysis to provide insights into the net social welfare of reducing agricultural pollutant loads and the subsequent public benefits of achieving the *Reef Plan* targets. It will then discuss the key findings and contribution to knowledge along with the policy implications and opportunities for further research.

10.1 Cost–benefit analysis

To understand the net social welfare of achieving sediment reductions, a cost benefit analysis was undertaken. To integrate the cost and benefits for achieving improved Reef health through sediment reductions, and complete a cost benefit analysis there are a number of complexities need to be accounted for to ensure equivalent comparison can be achieved. These include:

1. Ensuring that the benefits and costs align to the *Reef Plan* targets
2. Extending the benefits over a 20-year time frame with the same discount rate as the costs

3. Estimating the benefit of a tonnage reduction of sediment. Although other pollutants impact on reef health only sediment benefits will be focused on.
4. Comparing a one tonne cost in sediment reduction to a one tonne benefit of sediment reduced.

These will be stepped through to complete the cost benefit analysis.

1. *Alignment to Reef Plan*

To allow for an equivalent comparison between the cost of sediment reduced and the benefit of sediment reduced a cost benefit analysis was linked into the *Reef Plan* targets. It is these targets that current policy is based on and therefore provides relevance to the net social benefit. This however does require that both the costs and benefits be both evaluated in the same units and in this case tonnes of sediment was the selected unit.

Currently there is an estimated 14 million tonnes of anthropogenic sediment entering the GBR per annum (Queensland Department of the Premier and Cabinet 2011); an estimated 80 per cent of this sediment entering the GBR is attributed to grazing, approximately 11,200,000 tonnes. The 20 per cent *Reef Plan* target reduction is therefore equivalent to 2,240,000 tonnes per annum, and a one per cent reduction is therefore equivalent to a reduction of 112,000 tonnes per annum. Both a 20 per cent reduction and a one per cent reduction will be considered in the cost benefit analysis.

2. *Equivalent time frames and discount rate*

To allow the costs and benefits to be compared the same time frames and discounts rates were required. Sediment reduction costs were over a 20 year time frame to reflect the length of time of management, therefore the benefits also had to be modified to match this time frame. Similarly, the costs were calculated with a six per cent discount rate therefore the benefits were then also estimated using a six per cent discount rate.

The choice modelling values from Chapter Nine were derived providing an understanding of the benefits of achieving *Reef Plan* targets. The *Reef Plan* targets are 20% reduction in sediments and a 50% reduction in nutrients. The estimated average annual WTP from each of the choice modelling split samples were extrapolated out to 20 years and then discounted back by a six per cent discount rate to generate a net present value. This value was then multiplied by the portion of households likely to hold values to estimate the benefit per tonne of sediment reduction and the benefit to achieve a one per cent (112,000 tonnes) target and the *Reef Plan* 20 per cent target.

3. Estimating the benefit of sediment reductions

To estimate the community benefits per tonne of sediment reduced presented a challenge, given that the choice modelling experiment presented a combination of sediment and nutrient pollutants to achieve an improvement in reef health. It was also explained to respondents that agricultural run-off is one of a number of factors impacting the health of the reef. To ensure that the benefit of sediment reduction estimates were not overvalued, research by Waterhouse *et al*. (2012) provided insights to the relative risk of sediment, inorganic nitrogen and herbicide pollutants from particular catchments. Waterhouse *et al.* (2012) explored the relative risk of the pollutants on the reef ecosystem based on an anthropogenic load score, reef condition score and reef exposure score. Catchments were given an overall relative risk score out of a maximum score of 70 and a minimum score of 14. The Fitzroy basin received a score of 36 and the Burdekin a score of 33 out of the possible 70. Converting these scores to percentages of relative risk of 64 per cent for the Fitzroy and 59 per cent for the Burdekin. Although this considers all pollutants and not just sediment, it is acknowledged that sediment pollutants were noted as the priority contaminant for grazing lands, and a score of six was given out of nine (or 66 per cent) for reef exposure and impact (Waterhouse *et al.* 2012). The average percentage of relative risk values for the two catchments (62 per cent) was then divided by the

66 per cent (reef exposure impact) to estimate that a 41 per cent of improvements in reef health can be attributed to sediment from grazing lands.

The results estimate that the benefit of a one per cent improvement in reef health for 70% per cent of Qld households (1,153,970 households) (Rolfe and Windle 2011) is a mean net present value over 20 years of $246m total. When this is converted into sediment reductions (which account for only 41 per cent of the improvement in reef health) the community benefit has a mean net present value of $9 per tonne per household. For the 20 per cent reduction to be achieved the community benefit for the sediment reduction is $181 per tonne (Table 10.1).

Table 10.1 Net Present Benefits for reef health and sediment reductions using a six percent discount rate

	Minimum ($)	Mean ($)	Maximum ($)
70% Qld households ($M/1% improvement in Reef health)	99	246	359
Benefit per tonne to achieve 1% of *Reef Plan* sediment targets at 70% of Qld household ($/t)	4	9	13
Benefit per tonne to achieve 20% of *Reef Plan* sediment targets at 70% of Qld household ($/t)	73	181	264

4. *Comparing a one tonne cost in sediment reduction to a one tonne benefit of sediment reduced.*

To estimate the costs of generating sediment reductions, all opportunity costs across the land types and conditions that were positive in the bioeconomic modelling were averaged (Table 10.2). The average costs for the A condition, with no trees, encompassed the largest range of costs across all of the land types. Conversely, C condition land with trees had the smallest range of positive costs.

Table 10.2 Bioeconomic modelling land condition costs

	Minimum	**Mean**	**Maximum**
Average costs for sediment reductions ($/t)	1	158	1,572
A no trees ($/t)	1	367	1,572

A trees cost ($/t)	1	290	1,043
B no trees ($/t)	1	241	1,338
B with trees ($/t)	1	220	727
C no trees ($/t)	1	38	316
C trees ($/t)	1	37	83

The bioeconomic modelling showed that to achieve the 1 per cent reduction (112,000 tonnes) in sediment would cost on average $17.6 million per annum. However, there are a number of options of sediment reductions below $158 per tonne, which provide more cost-effective ways to reduce sediment, as highlighted in Chapter Six. To achieve the 20 per cent target in *Reef Plan*, a reduction in sediment using the average cost of $158 per tonne and a target of 2.24 million tonnes would be $353.9 million.

The estimated benefit of $9 per tonne is out outweighed by the average cost of sediment reduced from grazing lands of $158 per tonne. The community values for sediment reductions targets are roughly equivalent indicating that the current level of government investment adequately addresses the public benefit of obtaining GBR health into the future.

10.2 Contribution to knowledge

Understanding when improvements in water quality from changed grazing management practices generate net benefits to society is a key challenge to policymakers. This research has evaluated these aspects by comparing the marginal benefits of improvements in Reef health against the costs of the required sediment reductions required to reach *Reef Plan* targets. This type of analysis has been limited in the past because of information gaps such as scientific information to link management practices with improved reef health, understanding the values for improved reef health, and information about the costs of management practices.

This research contributes in several ways. First, it uses improved understanding about the links between management changes and *Reef Plan* targets to bring

together an analysis of the costs and benefits of marginal changes, helping address the policy questions relating to GBR protection. Second, it uses bioeconomic modelling to help better understand the asymmetric costs and trade-offs between management changes and sediment reductions from grazing systems. Third, it uses *Reef Plan* targets to frame a choice experiment to value the benefits of improved Reef health, explicitly linked to changes in targeted reductions in agricultural pollutants. Fourth, the information on marginal costs and benefits that is reported provides policymakers with information to help improve management decisions.

The results of the assessment of the benefits and costs in the analysis indicate that the public benefit of achieving the sediment reduction targets is equivalent to the opportunity costs of landholders achieving these reductions. The net social welfare across all pollutant is therefore for likely to be higher than the current level of government investment, indicating that further policy mechanisms to ensure the protection of the GBR is likely to be warranted. Although on a per tonne basis the benefits were less than the costs there are a number of options for lower cost opportunities in the Fitzroy and Burdekin Basin through targeting land in poor condition and in the lower productivity land types (Chapter Six). The range of costs provides a ceiling for future investments per tonne of sediment and allows further prioritisation relative to where the most cost-effective options are across the catchments.

The use of a bioeconomic model in rangelands grazing and exploring the profit and sediment trade-offs is the first application of its kind. This approach contributes to the knowledge of the dynamic nature of grazing systems, the impact of production over time and the environmental trade-offs which in this case focused on sediment.

The application of choice modelling has allowed the valuation of improved Reef health through reduction of pollutant reductions to be considered. This is the first time reported in the literature where the links between agricultural pollutants and subsequent reef health has been valued. This allowed the current context of a

policy action and ability to assess the actual *Reef Plan* targets and actions. The choice modelling experiment allowed for trade-offs to be made between pollutant, industry and improved Reef health allowing a valuation of improved Reef health to be derived.

The implications of using the same discount rate and time period to compare the costs and benefits presents challenges in application, as *Reef Plan* finishes in 2018. However, the costs required to have long-term change are long term management actions and therefore a 20 year time period was selected. Similarly, while the discount rate of five per cent may be an appropriate rate for discounting Reef health, the cost of changing management practice indicates that a higher discount rate is required. The bioeconomic modelling and the choice modelling also have large confidence intervals and therefore consideration of the sensitivity of the caveats is critical.

Caveats to note are that tests for preference stability over time have not been conducted and that it is difficult to extrapolate response rates accurately from an Internet panel survey to the population (Windle and Rolfe 2009). Similarly, the impact of climate and other pressures impacting the GBR have not been fully considered in this research. There are a number of other actions and targets that are a part of *Reef Plan* which has not been integrated into this research. The threshold, impacts and interactions between sediment, nutrient and pesticide pollutants, have limited science to express these relationships and therefore were not considered in this research.

10.3 Reflecting the literature

The results of this research reflect the complexity of NRM issues and difficulty that is faced by policymakers in achieving outcomes (Allan and Wilson 2009). The heterogeneity of the results highlights the opportunity for asymmetric information to exist and the difficulty for NRM groups in understanding what the actual investment will purchase as a quantified outcome (Williams 1996; Godden 1997; Fraser 2013).

The mix of policy mechanisms required and ability to be flexible within a catchment also presents challenges to overall program design (Bennett 2005; Mullen, 2000; Pannell 2009). The heterogeneity of costs in a variable climate also makes understanding the public and private trade-offs difficult. Therefore, the correct policy mechanism to reflect these trade-offs is more complex and difficult to implement correctly than current NRM frameworks suggest (Pannell 2009).

The heterogeneity in the cost of sediment reductions based on land types and climate locations creates complexity in implementing the most effective policy mix. The ability for NRM groups to quantify the results of their purchases and report outcomes such as tonnes reduced rather than hectares of land the investment covers (Rolfe *et al.* 2007; Pannell 2009) would require extensive resources, further skills and resources. It would also require a shift in paradigm based from previous reporting approaches.

This presents challenges in applying such frameworks when the public and private trade-offs vary significantly over the landscape and climate locations. However, the implementation of a bioeconomic model highlights this heterogeneity and the variance in costs, which has previously not been able to be captured or well understood by policymakers. It also indicates that although there are large amounts of heterogeneity the importance of targeting and prioritising investments is warranted (Pannell 2009).

The bioeconomic modelling also provides insights for rangelands grazing and sustainable grazing management practices. Although there has been a number of studies looking at the interactions between grazing management practices and the pasture and animal production changes there has been limited understanding on the economic implications of these practices (McIvor and Monypenny 1995; McKeon *et al.* 2000; O'Reagain *et al*. 2009; Orr and O’Reagain 2011; Orr and Phelps 2013).

The use of bioeconomic models to help predict the costs of sediment reductions provides a useful estimate of the costs and highlights the variance among land types and climate locations (Bennett 2003; Bennett 2005; Whitten and Bennett 2005). However, the complexity of capturing all the required variables and interactions in the model highlights the need for further pasture trials to generate data for a more robust model. Likewise, only one management approach was accounted for, which does not account for the ability for flexible and opportunistic management that occurs in variable climates and rangelands grazing (O'Reagain *et al.* 2011; Orr and O'Reagain 2011; Doole and Pannell 2013; Orr and Phelps 2013).

The ability for the Fitzroy Basin Association and NQ Dry Tropics to implement mechanisms such as tenders and auctions to reduce asymmetric information which arises from the landscape heterogeneity may present a more cost-effective option for future investments, and this work highlights there are significant opportunities to achieve this. Rolfe and Windle (2011b) found that using an auction mechanism in the Burdekin catchment sediment reductions could be achieved for $89 per tonne.

Previous work using choice modelling to value improvements in GBR health estimated a range of between $132.8m and $171.5m for a one per cent improvement in reef health (Rolfe and Windle 2011). These results are comparable to the range of $42m to $182.6m estimated in this research. Rolfe and Windle (2011) did not find any significant difference between the value of the Townsville, Brisbane, Sydney, Melbourne, Adelaide and Perth populations highlighting the iconic nature of the asset.

The use of choice modelling allows consideration of the community benefits of achieving water quality improvements for improvements in GBR health. Although previous non-market valuation studies have been completed valuing the recreational aspects and health of the GBR (Windle and Rolfe 2005; Kragt *et al.* 2009; Prayaga *et al.* 2010; Rolfe and Windle 2011), linking agricultural pollutant reductions to improved GBR health has not been previously completed.

The implementation of the cost–benefit analysis highlights the large disparity between current levels of government investment and community values, indicating the justification for further funds to be allocated and for more efficient use of funds to be of higher priority for NRM programs.

10.4 Policy implications

This research has contributed to NRM and specifically GBR health policy development in three main ways. Firstly, it addresses the limited information regarding costs from grazing lands for sediment reductions by providing a quantifiable assessment of opportunity costs for sediment reductions (Pannell 2009; Pannell and Roberts 2010; Rolfe and Windle 2011). Secondly, it provides insights into the benefits of improved GBR health from agricultural pollutant reductions, particularly the benefits of achieving the *Reef Plan* targets. Thirdly, it offers an attempt at completing a cost–benefit analysis to gain an understanding of the net social welfare implications and ability to achieve the *Reef Plan* targets.

10.1 Quantifying the cost trade-offs between sediment exported and opportunity cost for the grazier

With respect to the first research aim of estimating the cost of sediment reductions, this research has presented an analysis combining the biophysical and economic factors to estimate a dollar value per tonne of sediment reduced. This lack of information has been a key criticism of past NRM policies and a key limitation for targeting investments.

Past federal funding initiatives have resulted in a grants mechanism for water quality improvements. To date there has been no direct relationship found between the grant and associated water quality improvements. This research will allow groups such as the Fitzroy Basin Association and NQ Dry Tropics to target investments and more cost effectively allocate funds (Star *et al.* 2012). The bioeconomic modelling provided an understanding of the complexity of grazing enterprises and the heterogeneity that exists within land types, climate zone, business enterprises and starting land conditions. This complexity highlights the

potential for using a range of policy mechanisms to efficiently allocate funding resources. The asymmetric information and heterogeneity that exists means that policy mechanisms such as competitive tenders and auctions may be able to achieve a higher number of more cost-effective investments.

10.2 Estimating the required level of investment in relation to the social benefit

The ability to estimate the community benefit for agricultural pollutant reductions provides insights into the importance that the government should place on achieving improvements in GBR health and achievements through agricultural emissions reductions. Previous studies have predominantly focused on recreational value of the GBR (Kragt *et al.* 2009; Prayaga *et al.* 2010), water quality improvements from pollutant reductions, and policy options (Rolfe and Windle 2010; Rolfe and Windle 2011).

The presentation of agricultural pollutant reductions linked to water quality improvements and subsequent improvements in Reef health helped respondents to be aware of the input changes required, and may have made the survey more comprehensive and believable. It may have also minimised potential for double counting, where people may have expressed values for both improved water quality and improved GBR health without considering that the former is a prior condition of the latter. A limitation of the study is the low level of respondents who were confused by the choice options; this hinders the accuracy of the policy mechanisms required. The survey did not account for the broader population outside of Queensland, given that Rolfe and Windle (2010) did not find scale effects between the capital cities in previous GBR valuation studies highlighting the iconic nature of the asset.

10.3 The need for further policy framework development, which contributes to innovation

The targeting of resources more efficiently and effectively has been a key policy recommendation in reviews for previous NRM programs (Pannell, 2009a, Rolfe *et al*. 2007). Understanding the public and private costs and benefits helps select which policy mechanism is most effective at achieving environmental outcomes (Pannell *et al.* 2008). Similarly, a range of information regarding the importance of the environmental asset to the community, the project risks, adoption of new practices, time lags and costs are all required to fully understand how to prioritise investments and the subsequent policy recommendation. A popular framework for investing in NRM projects has been the natural resource management frameworks where the trade-offs between public and private benefits and costs frame which policy mechanism is the most effective (Pannell *et al.* 2008). The key limitation of the framework is it assumes that the public and private trade-offs are understood, are homogeneous and it does not account for the time required to shift to a new overall environmental position, making its applicability to a specific environmental issue difficult.

For different technologies or innovations there will be different perceived risks and subsequent costs and benefits depending on how the technology or innovation fits within the production system, time taken to see the benefits, and underpinning science. This therefore creates a challenge for GBR NRM organisations to effectively target funds to where the best opportunity is for public and private benefits to be achieved within the supply chain. For example, if there is an innovation that had a small marginal private benefit but is in an area of the production system that can be contracted out, the net private benefit will be realised much faster.

There is often the desire to understand and achieve a level of adoption across a whole industry. Currently across the grazing and sugarcane industries there is the desire to ensure that the landholder achieves a level of best management practice. However, shifting towards these practices has very little private benefits, especially over the short term, presenting a low likelihood of voluntary adoption. These management practice frameworks also has the potential to stifle innovation

in the long term and to create a culture that managers operating in the future will not have to continue to progress, as best management will have already been achieved. In the grazing industry, where adoption of new technologies is slow and profit margins are small, this may hinder long-term industry shifts.

10.5 Future research

Some of the policy implications of this research lead to future research questions and opportunities regarding NRM policy and particularly pollutant reductions for the GBR.

Currently NRM investment frameworks select policy mechanisms based on the private and public trade-offs. However these frameworks are difficult to apply as heterogeneity in costs and benefits and different policy response to this is not accounted for. Information about these trade-offs along with consideration of asymmetric information that occurs regarding private benefits and the actual marginal reduction in the pollutant is required. This therefore requires a new framework to be considered to better allocate resources to ensure measurable outcomes are achieved.

A key aspect of sediment reductions through management practices is the cumulative benefits, both privately and publicly, which have not been explored in this research. It would be expected that there is a threshold point at which the public benefits for improved reef health outweigh the private benefits of sediment reductions. However, there is currently little understanding of the combination of these practices and the subsequent impact both to the landholder and impact on sediment.

This research has solely focused on achieving sediment reductions, and therefore has not considered other potential environmental outcomes of improved land management such as carbon storage, biodiversity outcomes, and potential links to other ecosystems. Grouping multiple environmental outcomes will present different private costs and benefits, and different policy mechanisms are then required.

The limitations highlighted in both the bioeconomic modelling and the choice modelling also present opportunities for future research, such as improving the modelling to account for further paddock scale management and combining more than one land type to better account for the heterogeneity that exists within properties. Testing the choice modelling results in a mail-out survey and with a different choice set design also may ensure the results are more applicable to the broader community values.

10.6 Conclusions

Land base pollutant loads from agriculture have increased to a level many times greater than the natural rate for oceans, coral reefs, lakes and reservoirs globally including the Great Barrier Reef in Australia. The issues and contributions of this thesis are timely as UNESO reviews the status of the GBR world heritage listing and state and federal policy development continues.

The research has contributed to the NRM policy development through completing a cost benefit analysis to estimate the net social benefit of sediment reductions for improved Reef health. To achieve this bioeconomic modelling and a choice experiment were completed, to assess both the costs and the benefits. Although there are a number of contributions to the NRM policy debate there is recognition that further work is required.

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APPENDIX A: EXPERT OPINION CONTRIBUTORS

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APPENDIX B:RECOVERY AND DEGRADATION THRESHOLDS

Degradation and recovery threshold values based on utilisation of green material of total growth by the end of the growing season (30 April) for eight land types in the Burdekin and Fitzroy Basins

Land type	Recovery Threshold % Dry Matter Utilisation	Degradation Threshold % Dry Matter Utilisation
Brigalow blackbutt	22%	40%
Narrow leaved ironbark on mountains and ranges	12%	26%
Narrow leaved woodlands	14%	28%
Coolibah floodplains	18%	35%
Goldfields – red soils	15%	35%
Silver-leaved	15%	35%
Silver-leaved ironbark on duplex	15%	35%
Spotted gum ridges	10%	25%

APPENDIX C: SUMMARY OF LAND TYPE RESULTS

A condition with 0 Trees

Pasture utilisation (% TSDM)	15	20	25	30	35	40	45	50	55	60	65	70
Brigalow blackbutt												
Actual AEs for 5,000ha	706	914	1,110	1,294	1,468	1,639	1,791	1,915	1,895	1,789	1,772	1,805
Net Present Value ($)	22,779	152,026	275,490	387,591	494,878	597,549	691,702	796,955	904,211	812,423	737,105	669,458
Ave Total Sediment Exported (T)	1,055	1,146	1,240	1,342	1,445	1,511	1,666	1,958	3,730	6,536	8,377	9,336
$/T	22	1,421	1,322	1,096	1,037	1,572	605	361	61	-33	-41	-70
cCoolibah floodplains												
Actual AEs for 5,000ha	661	831	953	1,023	1,020	1,046	1,086	1,146	1,198	1,261	1,316	1,387
Net Present Value ($)	223,727	361,210	467,508	530,235	417,689	355,668	216,237	154,556	113,277	32,307	-65,121	-161,452
Ave Total Sediment Exported (T)	663	824	1,237	1,836	2,673	3,488	3,988	4,339	4,661	4,937	5,124	5,229
$/T	338	851	258	105	-134	-76	-279	-176	-128	-294	-522	-916
Gold Fields												
Actual AEs for 5,000ha	732	874	940	907	926	976	1,055	1,107	1,161	1,225	1,282	1,346
Net Present Value ($)	3,315	-173,025	-439,079	-856,711	-1,180,129	-1,578,011	-2,041,281	-2,465,344	-2,859,285	-3,342,675	-3,843,146	-4,498,485
Ave total sediment (T)	6,001	8,288	12,637	19,222	23,171	25,212	26,167	27,888	28,979	30,025	30,887	31,441
$/T	1	-77	-61	-63	-82	-195	-485	-246	-361	-462	-581	-1,181
Narrow Leaved ironbark ranges												
Actual AEs for 5,000ha	787	763	777	869	963	1,048	1,131	1,204	1,271	1,336	1,397	1,456
Net Present Value ($)	-1,446	180,027	226,635	274,795	311,843	356,166	326,801	359,703	334,617	264,622	293,681	266,478
Ave Total Sediment Exported (T)	6,518	23,312	39,376	44,476	47,692	50,822	53,697	56,416	59,007	61,636	63,863	65,813
$/T	0	8	6	6	7	7	6	6	6	4	5	4
Narrow Leaved Ironbark Woodlands												
Actual AEs for 5,000ha	723	895	914	827	883	950	1,026	1,100	1,165	1,230	1,296	1,359
Net Present Value ($)	1,805	47,050	193,120	331,668	337,874	366,975	341,126	317,696	298,671	286,076	257,308	212,905
Ave Total Sediment Exported (T)	2,669	3,422	4,893	5,694	5,933	5,845	5,782	6,015	5,764	6,040	5,837	6,052
$/T	1	60	99	173	26	-331	-407	-101	-76	-46	-141	-206
Silver-leaved Ironbark Nebo												
Actual AEs for 5,000ha	1,316	1,496	1,620	1,672	1,608	1,683	1,780	1,892	1,998	2,076	2,173	2,265
Net Present Value ($)	263,556	444,899	649,556	950,006	856,570	735,415	622,884	456,469	311,045	160,201	-147,515	-219,265
Ave total sediment (T)	1,673	3,891	7,915	14,239	23,352	28,582	32,398	34,664	36,831	39,062	41,459	42,499
$/T	158	82	51	48	-10	-23	-29	-73	-67	-68	-128	-69
Silver-leaved Ironbark Springvale												
Actual AEs for 5,000ha	720	804	757	800	831	866	906	954	1,008	1,036	1,100	1,137
Net Present Value ($)	166,388	349,278	454,088	506,554	473,242	416,012	377,627	292,234	174,170	27,008	-62,279	-223,740
Ave total sediment (T)	4,058	7,213	15,608	19,542	23,023	25,775	27,932	29,564	30,768	31,830	32,974	34,104
$/T	41	58	12	13	-10	-21	-18	-52	-98	-139	-78	-143
Spotted Gum												
Actual AEs for 5,000ha	622	625	646	699	742	801	850	894	937	963	1,012	1,050

Net Present Value ($)	172,908	315,974	456,637	497,902	517,542	560,256	504,541	527,473	480,346	531,887	603,657	717,127
Ave total sediment (T)	9,690	21,986	30,222	35,234	39,101	41,361	43,789	46,097	47,868	49,629	51,169	52,682
$/T	18	12	17	8	5	19	-23	10	-27	29	47	75

A condition with tree basal area

Pasture utilisation (% TSDM)	15	20	25	30	35	40	45	50	55	60	65	70
Brigalow blackbutt												
Actual AEs for 5,000ha	562	727	886	1,039	1,182	1,312	1,432	1,520	1,537	1,449	1,441	1,478
Net Present Value ($)	32,403	134,490	238,876	335,135	419,981	499,472	558,925	557,047	517,121	308,185	183,621	33,375
Ave Total Sediment Exported (T)	1,972	2,106	2,206	2,323	2,485	2,732	3,021	3,851	5,394	8,755	10,403	11,356
$/T	16	762	1,043	824	524	321	206	-2	-26	-62	-76	-158
Coolibah floodplains												
Actual AEs for 5,000ha	361	460	537	580	592	611	652	702	734	763	796	829
Net Present Value ($)	82,889	117,018	108,878	33,049	-57,373	-206,028	-367,888	-452,809	-561,654	-720,087	-838,467	-996,278
Ave Total Sediment Exported (T)	1,587	1,739	2,033	2,610	3,554	4,274	4,433	4,563	4,773	5,109	5,228	5,374
$/T	52	224	-28	-131	-96	-207	-1,016	-656	-517	-472	-999	-1,077
Goldfields												
Actual AEs for 5,000ha	732	874	940	907	926	976	1,055	1,107	1,161	1,225	1,282	1,346
Net Present Value ($)	3,315	-173,025	-439,079	-856,711	-1,180,129	-1,578,011	-2,041,281	-2,465,344	-2,859,285	-3,342,675	-3,843,146	-4,498,485
Ave total sediment (T)	6,001	8,288	12,637	19,222	23,171	25,212	26,167	27,888	28,979	30,025	30,887	31,441
$/T	1	-77	-61	-63	-82	-195	-485	-246	-361	-462	-581	-1,181
Narrow Leaved Ironbark Ranges												
Actual AEs for 5,000ha	178	188	210	235	257	277	298	319	340	358	376	395
Net Present Value ($)	-108,032	-258,833	-385,077	-532,736	-676,209	-819,077	-988,603	-1,120,318	-1,252,420	-1,458,685	-1,581,114	-1,762,441
Ave Total Sediment Exported (T)	46,664	61,306	65,736	68,045	69,824	71,200	71,910	72,352	72,662	72,805	73,228	73,316
$/T	-2	-10	-29	-64	-81	-104	-239	-298	-426	-1,444	-289	-2,077
NLIB woodlands												
Actual AEs for 5,000ha	242	297	317	332	359	375	397	422	446	466	497	518
Net Present Value ($)	-45,571	-76,168	-224,219	-343,295	-479,461	-617,478	-787,776	-914,979	-1,039,129	-1,237,361	-1,341,518	-1,550,043
Ave Total Sediment Exported (T)	15,741	18,827	25,631	30,334	32,774	35,174	36,499	37,478	38,268	38,902	39,231	39,692
$/T	1	60	99	173	26	-331	407	-101	76	-46	141	-206
SLIB -Nebo												
Actual AEs for 5,000ha	601	719	745	725	738	785	840	869	908	942	990	1,015
Net Present Value ($)	163,108	111,574	-76,397	-405,360	-758,284	-1,086,758	-1,462,456	-1,681,784	-1,915,423	-2,250,414	-2,440,821	-2,710,294
Ave total sediment (T)	1,673	3,891	7,915	14,239	23,352	28,582	32,398	34,664	36,831	39,062	41,459	42,499
$/T	12	-11	-20	-26	-47	-91	-162	-65	-119	-242	-124	-140
SLIB Springvale												
Actual AEs for 5,000ha	161	183	208	224	238	253	267	283	300	325	324	339
Net Present Value ($)	-69,929	-188,554	-313,410	-456,809	-579,046	-690,297	-854,748	-975,427	-1,085,773	-1,248,996	-1,373,339	-1,567,808
Ave total sediment (T)	26,638	30,999	33,336	35,490	37,560	38,891	39,957	40,661	41,190	42,065	42,440	42,753
$/T	-3	-27	-53	-67	-59	-84	-154	-172	-209	-231	-229	-621
Spotted Gum												
Actual AEs for 5,000ha	72	76	84	94	103	111	120	129	137	143	152	159
Net Present Value ($)	-12,231	-56,475	-94,788	-138,255	-177,941	-218,033	-268,079	-307,044	-342,472	-400,939	-445,482	-510,593
Ave total sediment (T)	53,757	57,216	58,700	59,457	60,067	60,569	60,745	60,905	60,987	61,077	61,149	61,243
$/T	0	-13	-26	-57	-65	-80	-285	-244	-433	-645	-618	-695

B condition with zero tree basal area

Pasture utilisation (% TSDM)	15	20	25	30	35	40	45	50	55	60	65	70
Brigalow Blackbutt												
Actual AEs for 5,000ha	700	907	1,099	1,276	1,404	1,513	1,639	1,677	1,552	1,538	1,551	1,591
Net Present Value ($)	2,483	130,743	252,742	360,564	467,899	570,014	679,887	740,730	712,465	590,631	505,921	390,422
Ave Total Sediment Exported (T)	1,109	1,205	1,316	1,454	1,867	2,414	2,735	4,015	7,383	9,357	10,649	11,511
$/T	2	1,338	1,103	782	260	187	343	48	-8	-62	-66	-134
Coolibah flood plains												
Actual AEs for 5,000ha	635	771	811	818	838	861	909	957	1,011	1,068	1,124	1,184
Net Present Value ($)	199,631	283,071	302,730	187,498	93,113	-20,042	-187,495	-293,344	-348,551	-496,070	-565,850	-695,522
Ave Total Sediment Exported (T)	788	1,100	1,961	3,050	3,805	4,572	4,983	5,353	5,663	5,918	5,990	6,152
$/T	253	268	23	-106	-125	-147	-408	-286	-178	-578	-971	-801
Gold fields												
Actual AEs for 5,000ha	1,020	1,085	1,159	1,221	1,253	1,316	1,395	1,480	1,570	1,664	1,739	1,832
Net Present Value ($)	76,617	205,943	205,958	67,789	-366,795	-805,318	-1,322,347	-1,742,847	-2,058,271	-2,487,369	-2,967,895	-3,555,111
Ave total sediment (T)	4,102	8,385	12,524	16,679	20,770	23,799	25,938	27,297	28,363	29,280	30,438	31,326
$/T	19	30	0	-33	-106	-145	-242	-309	-296	-468	-415	-661
NLIB ranges												
Actual AEs for 5,000ha	684	632	675	763	852	932	1,008	1,079	1,145	1,202	1,256	1,314
Net Present Value ($)	-10,038	135,109	185,671	191,326	194,292	212,015	148,225	108,395	56,371	-23,126	-97,599	-186,323
Ave Total Sediment Exported (T)	10,719	34,259	47,719	52,039	55,295	58,265	61,011	63,547	65,886	68,900	71,176	73,061
$/T	-1	6	4	1	1	6	-23	-16	-22	-26	-33	-47
NLIBW -daurgina												
Actual AEs for 5,000ha	684	767	713	701	757	837	907	974	1,036	1,097	1,161	1,216
Net Present Value ($)	26,083	101,442	267,836	281,469	295,472	274,015	239,773	204,366	174,182	99,471	76,439	31,502
Ave Total Sediment Exported (T)	3,400	5,110	5,633	5,875	6,031	6,102	6,257	6,203	5,923	6,243	6,086	6,362
$/T	8	44	318	56	90	-304	-221	-660	-108	-233	146	-163
Silver Leaved Ironbark- Nebo												
Actual AEs for 5,000ha	1,137	1,166	1,223	1,294	1,328	1,390	1,484	1,592	1,688	1,761	1,846	1,941
Net Present Value ($)	208,785	447,649	663,083	733,598	599,205	461,685	240,337	42,763	-263,545	-710,802	-975,703	-1,380,204
Ave total sediment (T)	3,793	9,619	17,459	23,376	30,365	35,999	39,199	41,200	43,351	46,141	48,139	49,570
$/T	55	41	27	12	-19	-24	-69	-99	-142	-160	-133	-283
Silver leaved Ironbark-Duplex												
Actual AEs for 5,000ha	146	164	182	197	205	223	239	252	267	297	291	305
Net Present Value ($)	-70,526	-200,985	-315,877	-445,474	-554,334	-660,434	-805,077	-918,420	-1,029,101	-1,215,203	-1,339,027	-1,517,629
Ave total sediment (T)	28,678	33,157	35,926	37,916	40,185	40,970	41,786	42,548	43,008	43,436	44,018	44,309
$/T	-2	-29	-41	-65	-48	-135	-177	-149	-241	-453	-207	-615
Spotted Gum												
Actual AEs for 5,000ha	519	484	529	577	632	685	734	778	816	845	884	922
Net Present Value ($)	155,926	312,020	380,847	406,523	383,376	371,162	370,853	385,811	361,906	331,802	323,749	375,157
Ave total sediment (T)	15,249	31,511	37,945	42,658	45,441	47,796	49,868	51,875	53,764	55,617	57,225	58,598
$/T	10	10	11	5	-8	-5	0	-7	-13	-16	-5	-37

B condition with tree basal area

Pasture utilisation (% TSDM)	15	20	25	30	35	40	45	50	55	60	65	70
Brigalow blackbutt												
Net Present Value ($)	23,855	123,907	218,316	308,791	352,076	405,526	303,916	225,548	74,446	-165,623	-222,173	-388,652
Ave Total Sediment (T)	2,066	2,215	2,345	2,605	3,261	3,940	6,110	8,025	10,767	12,602	13,246	13,823
$/T	12	671	727	348	66	79	-47	-41	-55	-131	-88	-288
Coolibah flood plains												
Net Present Value ($)	65,556	73,439	2,578	-134,929	-233,497	-373,058	-524,989	-625,205	-739,486	-911,256	-1,012,574	-1,182,952
Ave Total Sediment (T)	1,729	1,961	2,507	3,241	3,932	4,866	5,118	5,310	5,523	5,857	5,867	6,105
$/T	38	37	1	-42	-59	-77	-103	-118	-134	-156	-173	-194
Gold fields												
Net Present Value ($)	-93,881	-266,397	-570,263	-961,902	-1,338,152	-1,675,291	-2,115,906	-2,494,846	-2,955,845	-3,353,871	-3,807,132	-4,399,970
Ave total sediment (T)	10,151	14,835	21,998	25,555	28,215	29,315	31,255	32,382	33,202	34,721	35,467	36,244
$/T	-9	-37	-42	-110	-141	-307	-227	-336	-562	-262	-608	-762
Narrow leaved ironbark ranges												
Net Present Value ($)	-144,538	-296,964	-422,608	-573,477	-720,501	-859,318	-1,041,706	-1,184,875	-1,323,482	-1,530,664	-1,654,290	-1,854,165
Ave Total Sediment(T)	55,434	67,778	71,525	73,274	74,564	75,202	75,497	75,846	76,201	76,215	76,480	76,437
$/T	-3	-12	-34	-86	-114	-217	-618	-410	-390	-15,693	-467	-4,717
Narrow leaved ironbark woodlands												
Net Present Value ($)	-9,742	13,207	-47,785	-237,081	-335,101	-425,804	-539,804	-644,596	-775,953	-963,472	-1,061,128	-1,251,508
Ave Total Sediment (T)	22,641	40,105	61,077	76,796	80,796	83,123	84,702	86,021	87,423	88,548	89,098	89,976
$/T	0	0	-1	-3	-4	-5	-6	-7	-9	-11	-12	-14
Silver leaved Ironbark												
Net Present Value ($)	-20,153	-136,422	-397,094	-710,506	-990,251	-1,326,369	-1,678,941	-1,906,018	-2,162,831	-2,456,046	-2,741,941	-3,133,860
Ave total sediment (T)	23,688	34,991	46,158	54,185	58,098	61,622	64,065	66,113	68,771	70,091	70,882	72,389
$/T	-1	-10	-23	-39	-71	-95	-144	-111	-97	-222	-361	-260
Silver leaved ironbark -duplex												
Net Present Value ($)	-70,526	-200,985	-315,877	-445,474	-554,334	-660,434	-805,077	-918,420	-1,029,101	-1,215,203	-1,339,027	
Ave total sediment (T)	28,678	33,157	35,926	37,916	40,185	40,970	41,786	42,548	43,008	43,436	44,018	
$/T	-2	-29	-41	-65	-48	-135	-177	-149	-241	-453	-207	
Spotted Gum												
Net Present Value ($)	-28,102	-71,676	-114,118	-165,644	-207,844	-255,691	-311,253	-352,025	-393,967	-454,949	-498,065	-568,624
Ave total sediment (T)	56,116	59,138	60,170	61,073	61,574	62,029	62,164	62,303	62,419	62,501	62,546	62,598
$/T	-1	-14	-41	-57	-84	-105	-413	-292	-364	-737	-960	-1,352

C condition 0 trees

Pasture utilisation (% TSDM)	15	20	25	30	35	40	45	50	55	60	65	70
Brigalow Blackbutt												
Actual AEs for 5,000ha	664	846	1,000	1,048	976	1,047	1,126	1,173	1,217	1,284	1,354	1,425
Net Present Value ($)	60,800	174,074	265,813	283,675	309,006	339,332	368,736	362,720	291,547	223,812	203,576	138,486
Ave Total Sediment Exported (T)	1,905	2,263	2,918	4,771	8,369	9,440	10,398	11,592	12,644	13,309	13,721	14,020
$/T	32	316	140	10	7	28	31	-5	-68	-102	-49	-217
cCoolibah flood plains												
Actual AEs for 5,000ha	543	609	602	607	657	711	764	823	878	928	986	1,036
Net Present Value ($)	135,907	101,617	-4,970	-146,486	-222,674	-288,255	-416,214	-506,196	-592,709	-785,236	-845,253	-1,027,258
Ave Total Sediment Exported (T)	1,703	2,441	3,709	4,830	5,369	5,709	6,095	6,286	6,492	6,732	6,780	6,913
$/T	80	-46	-84	-126	-142	-192	-332	-469	-421	-803	-1,253	-1,361
Gold Fields												
Actual AEs for 5,000ha	600	692	813	895	971	1,033	1,117	1,199	1,280	1,357	1,435	1,512
Net Present Value ($)	81,693	20,172	-155,783	-432,525	-753,650	-1,051,633	-1,428,518	-1,741,390	-2,017,823	-2,458,369	-2,938,422	-3,486,411
Ave total sediment (T)	15,358	19,333	21,547	24,356	26,907	29,478	30,958	32,047	33,055	34,044	34,762	35,381
$/T	5	-15	-79	-99	-126	-116	-255	-287	-274	-445	-669	-885
NLIB ranges												
Actual AEs for 5,000ha	441	530	600	687	767	842	914	978	1,039	1,096	1,148	1,197
Net Present Value ($)	91,973	121,966	118,727	91,112	37,640	-20,939	-88,439	-134,477	-181,060	-310,671	-338,852	-446,944
Ave Total Sediment Exported (T)	38,463	47,441	56,292	59,950	63,225	66,249	68,523	71,096	73,619	76,235	78,303	80,448
$/T	2	3	2	2	1	0	-1	-2	-2	-4	-4	-6
NLIBW												
Actual AEs for 5,000ha	154	165	186	207	227	248	269	289	307	326	343	362
Net Present Value ($)	193,299	308,800	394,030	463,372	527,106	597,936	633,435	678,408	684,380	672,878	676,098	688,885
Ave Total Sediment Exported (T)	29,511	37,806	43,736	53,103	55,741	57,922	59,702	61,299	62,762	64,140	65,386	66,620
$/T	7	8	9	9	9	10	11	11	11	10	10	10
SLIB nebo												
Actual AEs for 5,000ha	1,137	1,166	1,223	1,294	1,328	1,390	1,484	1,592	1,688	1,761	1,846	1,941
Net Present Value ($)	208,785	447,649	663,083	733,598	599,205	461,685	240,337	42,763	-263,545	-710,802	-975,703	-1,380,204
Ave total sediment (T)	3,793	9,619	17,459	23,376	30,365	35,999	39,199	41,200	43,351	46,141	48,139	49,570
$/T	55	41	27	12	-19	-24	-69	-99	-142	-160	-133	-283
SLIB springvale												
Actual AEs for 5,000ha	375	376	420	472	522	569	609	654	695	731	774	809
Net Present Value ($)	62,764	58,809	-3,080	-58,474	-151,584	-231,710	-411,658	-528,200	-657,728	-832,874	-982,318	-1,233,552
Ave total sediment (T)	18,585	26,741	30,578	32,876	34,551	36,020	37,565	38,681	39,674	40,535	41,418	42,499
$/T	3	0	-16	-24	-56	-55	-116	-104	-130	-203	-169	-307
Spotted Gum no trees												
Actual AEs for 5,000ha	519	484	529	577	632	685	734	778	816	845	884	922
Net Present Value ($)	155,926	312,020	380,847	406,523	383,376	371,162	370,853	385,811	361,906	331,802	323,749	375,157
Ave total sediment (T)	15,249	31,511	37,945	42,658	45,441	47,796	49,868	51,875	53,764	55,617	57,225	58,598
$/T	10	10	11	5	-8	-5	0	-7	-13	-16	-5	-37

C condition with tree basal area

Pasture utilisation (% TSDM)	15	20	25	30	35	40	45	50	55	60	65	70
Brigalow blackbutt												
Actual AEs for 5,000ha	527	670	795	825	834	868	908	941	980	1,022	1,083	1,138
Net Present Value ($)	54,989	109,396	156,224	47,849	-14,151	-62,022	-138,641	-233,558	-293,614	-429,070	-466,716	-586,195
Ave Total Sediment Exported (T)	3,193	3,846	4,539	6,916	9,601	11,747	13,165	14,567	15,361	16,169	16,311	16,663
$/T	17	83	68	-46	-23	-22	-54	-68	-76	-168	-266	-339
Coolibah flood plains												
Actual AEs for 5,000ha	305	364	401	418	435	451	472	503	532	563	592	620
Net Present Value ($)	36,884	-13,135	-119,074	-242,603	-338,484	-470,583	-608,038	-707,670	-803,764	-950,316	-1,045,696	-1,198,820
Ave Total Sediment Exported (T)	2,472	2,993	3,608	4,311	5,060	5,737	6,040	6,288	6,456	6,760	6,851	6,952
$/T	15	-96	-172	-176	-128	-195	-454	-403	-570	-483	-1,043	-1,523
Gold Fields												
Actual AEs for 5,000ha	367	403	464	523	571	612	664	712	756	799	846	888
Net Present Value ($)	-186,151	-418,186	-682,764	-978,820	-1,246,945	-1,521,958	-1,877,852	-2,175,705	-2,487,645	-2,964,791	-3,308,169	-3,798,971
Ave total sediment (T)	23,151	28,213	30,440	32,173	33,798	35,436	36,469	37,422	38,494	39,391	40,114	40,830
$/T	-8	-46	-119	-171	-165	-168	-344	-313	-291	-532	-475	-685
NLIB ranges												
Actual AEs for 5,000ha	123	137	160	181	201	223	243	262	280	297	314	330
Net Present Value ($)	-121,261	-275,304	-407,063	-572,101	-712,955	-852,427	-1,050,035	-1,188,290	-1,319,928	-1,525,750	-1,645,866	-1,856,152
Ave Total Sediment Exported (T)	67,237	75,707	77,271	78,184	78,688	78,823	78,914	78,972	79,005	78,987	79,108	79,136
$/T	-2	-18	-84	-181	-280	-1,038	-2,151	-2,401	-3,993	-11,355	-995	-7,275
NLIBW												
Actual AEs for 5,000ha	302	297	343	382	424	470	514	558	601	641	683	722
Net Present Value ($)	-27,203	-84,735	-187,754	-327,203	-438,819	-547,472	-716,309	-837,021	-967,322	-1,168,215	-1,284,680	-1,478,816
Ave Total Sediment Exported (T)	44,452	72,315	79,228	85,443	88,929	90,439	91,856	92,879	93,681	94,875	95,166	95,768
$/T	-1	-1	-2	-4	-5	-6	-8	-9	-10	-12	-13	-15
SLIB nebo												
Actual AEs for 5,000ha	289	333	363	412	446	477	513	545	574	605	634	662
Net Present Value ($)	-109,555	-239,774	-444,509	-669,911	-906,679	-1,105,726	-1,373,338	-1,563,605	-1,778,642	-2,102,003	-2,339,361	-2,707,932
Ave total sediment (T)	47,846	55,831	63,177	65,656	68,675	71,525	73,312	74,984	76,685	77,871	78,908	79,892
$/T	-2	-16	-28	-91	-78	-70	-150	-114	-126	-272	-229	-375
SLIB springvale												
Actual AEs for 5,000ha	126	138	152	172	176	190	199	211	219	244	238	250
Net Present Value ($)	-196,995	-296,605	-383,256	-486,253	-569,147	-670,150	-796,203	-891,541	-989,919	-1,112,584	-1,222,125	-1,364,286
Ave total sediment (T)	31,852	36,199	38,973	39,839	42,114	42,913	44,016	44,619	45,329	45,973	46,269	46,447
$/T	-6	-23	-31	-119	-36	-127	-114	-158	-139	-191	-370	-799
Spotted Gum												
Actual AEs for 5,000ha	48	52	58	65	71	78	85	92	100	105	113	120
Net Present Value ($)	-24,166	-60,449	-100,674	-150,144	-193,028	-237,349	-295,582	-336,212	-378,140	-438,204	-477,195	-542,702
Ave total sediment (T)	58,576	61,016	62,140	62,792	63,279	63,597	63,662	63,728	63,770	63,777	63,821	63,857
$/T	0	-15	-36	-76	-88	-139	-895	-614	-997	-9,422	-882	-1,839

APPENDIX D: CHOICE MODELLING SURVEY

What priority do you believe that the QLD and federal government should give to the following areas?

Please rank the following from 1 to 4 where 1 is the most important and 4 is the least important.
Click on the boxes in the order that you want them ranked.
You can deselect the boxes by clicking on them again to change your selection

- ☐ Education
- ☐ Law enforcement
- ☐ Environment
- ☐ Public health

- ☐ Yes – Vey confident
- ☐ Yes – Mildly confident
- ☐ No

Aside from the Great Barrier Reef, have you visited any other marine parks or coral reefs in the following areas?

- ☐ Nigaloo and Capes Marine Park in Western Australia
- ☐ In other parts of Australia
- ☐ Overseas
- ☐ None of the above

Below is a list of possible actions that could be undertaken to reduce the pressures on the GBR. Keeping in mind the impacts on relevant industries, please score each one from (**1**) NOT important to (**5**) VERY important.

Reduce the impacts of coastal residential development	1	2	3	4	5
Reduce the impacts of coastal infrastructure development	1	2	3	4	5
Reduce the impacts from industrial development	1	2	3	4	5
Improve water quality runoff from cattle grazing	1	2	3	4	5
Improve water quality runoff from cropping and irrigation	1	2	3	4	5
Reduce the impacts of recreational fishing	1	2	3	4	5
Reduce the impacts of commercial fishing	1	2	3	4	5
Reduce the impacts of tourism	1	2	3	4	5
Reduce the impacts of greenhouse gas emissions	1	2	3	4	5
Increase controls over shipping	1	2	3	4	5

What is your gender?

☐ Male

☐ Female

Which of the following age groups do you belong to?

☐ 18-30 years

☐ 31-35 years

☐ 36-45 years

☐ 46-60 years

☐ 61-75 years

☐ Over 75

The pressure of land based activities on the Great Barrier Reef

The Great Barrier Reef is one of the most diverse ecosystems in the world, and has been classified as a World Heritage area. However, there are many pressures on the system from increased human activity, climate change and natural causes such as cyclones.

This survey looks at a particular aspect of reef health.

We will first ask a couple of questions, followed by some information on the subject and then some more questions.

The pressure of land based activities on the Great Barrier Reef

The majority of the Great Barrier Reef (GBR) is in good condition however areas closer to the coast are not as healthy as they once were. There are a number of issues which impact reef health such as poor water quality from increased sediments and nutrients.

To start with ... some quick questions on your knowledge of the GBR and its catchments:

1. Which of the following do you think best describes what has happened to the health of the GBR over the past ten years? (Please tick one box)

   Declined health ☐

   Improved health ☐

   Stayed much the same ☐

   I don't know ☐

2. From the factors below which do you believe cause the greatest adverse pressure on the GBR? (Please rank from the greatest effect =1 to the least effect =4)

   Climate change ☐

   Over fishing ☐

   Nutrient run-off ☐

   Sediment run-off ☐

3. What would your motivation be to improve the health of the GBR? (Please tick one box)

To maintain recreational fishing ☐

To ensure use for future generations ☐

To maintain the tourism industry ☐

To visit it myself ☐

There are a number of issues that are relevant to the GBR but for this study we are focusing on the following:

1. Reef health and water quality, and

2. The activities to achieve improvements in water quality.

There are many pressures which can lead to poor reef health but a significant contributor is land based activities.

Land- based activity: Activities that occur on agricultural and grazing lands and also in urban and industrial activities impact the water quality that enters into rivers and streams that empty into the GBR.

Water quality is affected by sediment and nutrients. 80 per cent of this sediment and nutrients exported to the GBR is from grazing (Productivity Commission 2003; Wilkinson & Bartley 2010).

Nutrients can best be described as trace elements from fertilisers and soil, and sediment is best described as small soil particles which are carried through the water.

The impacts of water quality on the GBR

1. Reduces the diversity of corals
2. Increases the macro algae which is a reef weed
3. Affects nearly 23% of reefs where water quality guidelines are exceeded

Addressing the problem

- Improvements could be made by reducing sediment and nutrient run off
- It is estimated that for the inner reef area improvements in water quality could;
  - Increase hard and soft corals by about 25%
  - Reduce reef weeds such as macro algae by about 39% (Fabricius & De'ath 2004).

Improved water quality → Better water clarity → Improved reef health

Targets:
The state and federal government have created a plan for water quality improvement. Targets are to reduce from current levels:

- sediment by 20% by 2020 and
- nutrient by 50% by 2020.

Current annual levels of sediment are 14 million tonnes (due to human activity) + 3 million tonnes (from natural processes)= 17million tonnes

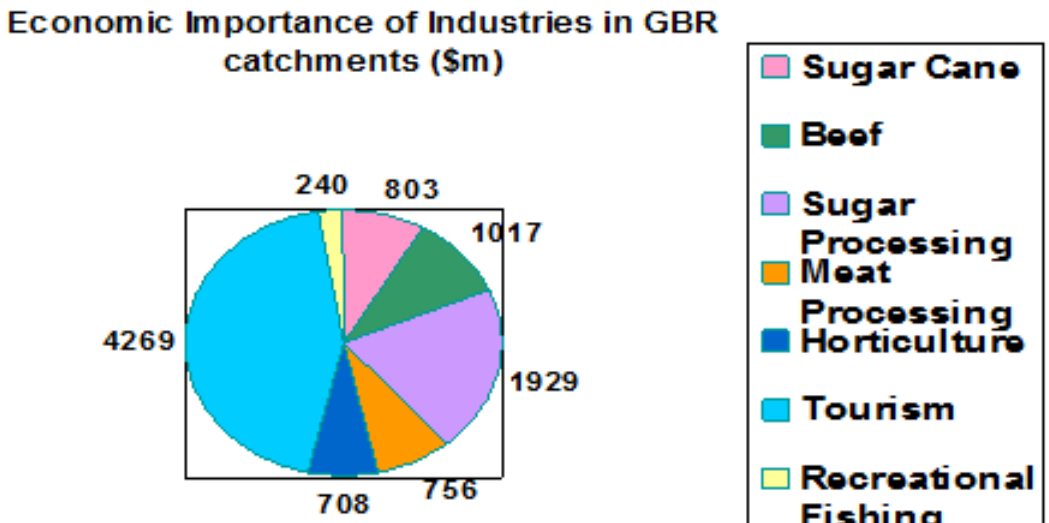
The Fitzroy and Burdekin catchments together contribute 8.8 million tonnes per year, which is approximately 352 Olympic swimming pools full of sediment.

Current levels of total load of dissolved nitrogen are 17,000 tonnes (due to human activity) + 14,000 tonnes (natural process)= 31,000 tonnes per year

The focus of this survey is on improving water quality from land-based activities.

Areas of improvement:
There are three main agricultural industries in the GBR catchments grazing, sugar cane, and horticulture. Grazing (80% of catchment) and sugar cane production (1 %) utilise the highest amount of land area in the GBR catchments (Carroll *et al.* 2011). There may need to be changes to management and production in these industries to achieve improved water quality.

Grazing, sugar cane and horticulture contribute significantly to the economy in the GBR regions, as do the sugar processing, meat processing, tourism and recreational fishing.



Source: Productivity Commission (2003)

Making sediment and nutrient reductions from these industries will involve different costs.

Figure 1. Great Barrier Reef Catchments.
The green area is the approximate area where sugar cane is grown. The majority of the remaining area is used for grazing.

Sugar Cane Production (area marked in Figure 1):

The production of sugar cane uses different applications of fertilizers, chemicals and nutrients.

Trace quantities of nutrient and small amounts of soil are then washed out into the reef and are predominately found in the in-shore and mid-zone reefs (shore to greater than 26km from the coast).

This impacts negatively on coral species and macroalgae

Grazing:
The production of cattle can reduce the amount of grass and pasture species that cover the ground. If the ground is relatively bare this can result in large amounts of soil getting washed out in rainfall events onto the near shore (within 25km from the coast).
This impacts the onshore sea grasses and reduces the light available for coral growth.

- An overall reduction in nutrients would result in an improvement in overall reef health.
- An overall reduction in sediment (soil) movement would result in improvements in inshore reef health.

To reduce nutrient run-off cane growers can take up practices such as reduced nutrient application, reduced spray chemicals, reduced tillage, controlled traffic and general improvements to improve soil and nutrient run-off. To achieve this new technologies are required with sugar cane growers having large costs to purchase this equipment.

Figure 2. Improving nutrient application to more precise methods like this will reduce nutrient run-off

- To reduce sediment graziers can improve grazing management to increase ground cover and reduce the bare soil that can be washed away.

Figure 3. Improving ground cover in grazing areas like this will reduce sediment run-off

While landholders are already proactive, financial support will be needed to generate changes quickly. The government already has some programs, but larger changes will require more funding.

Guide to survey Questions

In this survey we want to know what you think of different management options for improving water quality into the GBR. To do this we present some scenarios to choose between.

Current predictions are for a decline in GBR health over the next 50 years from adverse water quality, climate change and other direct pressures. This decline may be reduced with better water quality.

Information in this survey represents the best available scientific data to date.

Extra management would cost money which may be paid for by:

- Increased taxes levied by state and federal governments.
- Increased rate payments to the local council.
- Increased prices for goods and services such as food as farmers meet higher environmental standards.

Some other things to consider:

Also you may want to consider that there are ***other World Heritage Areas in Australia*** that may require additional funding for management.

There are ***other important environmental issues*** and Australian or international icons that you may wish to support.

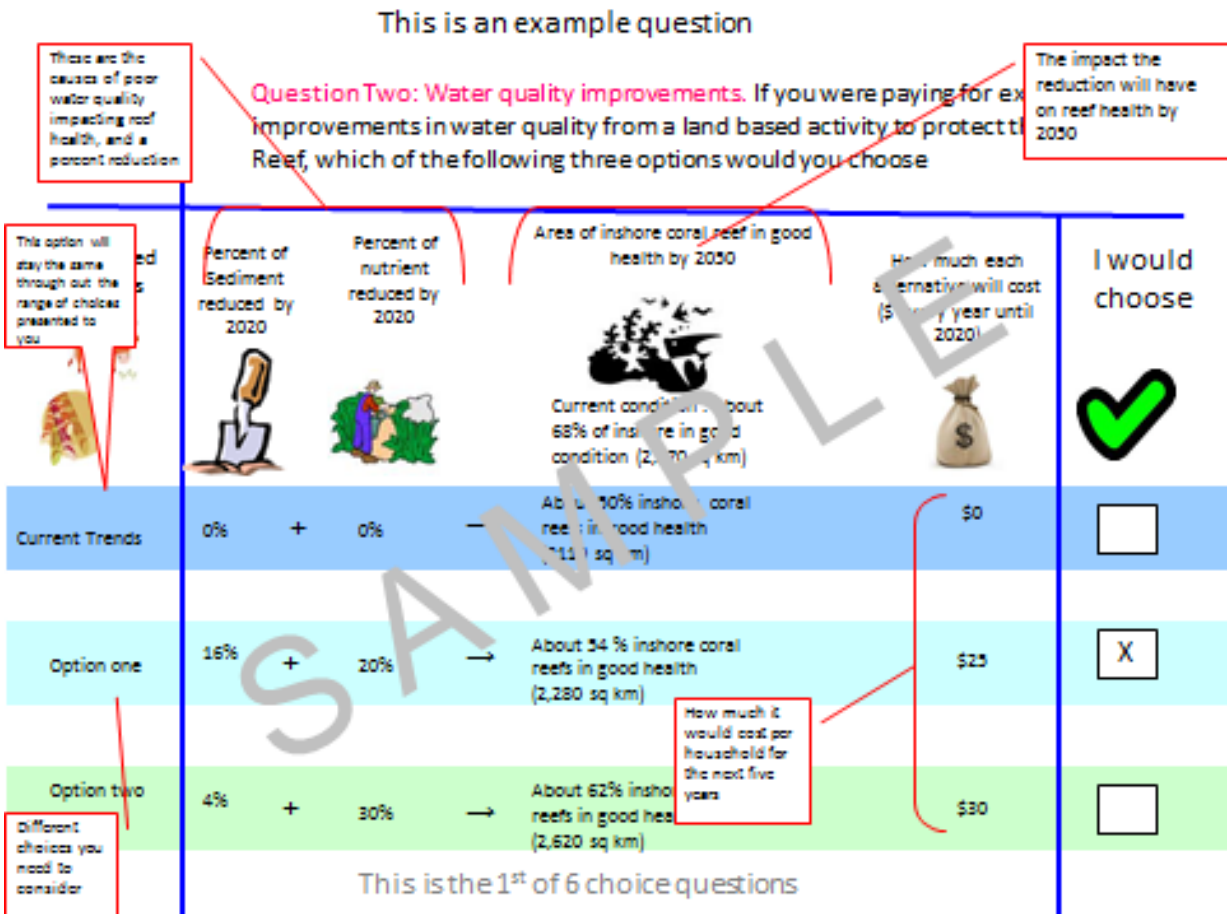
You should consider ***your income and other expenses***. The cost is an annual amount, per household until 2020.

The next page is an example question, there will be six like this on the following pages.

These scenarios are hypothetical but are based on ***current scientific knowledge*** about what might happen.

Please make your choices as if they were ***real***.

The choices on each page ***may look very similar, but they do differ***. Please treat each page separately.


This is an example question
These are the causes of poor water quality impacting reef health, and a percent reduction
Question Two: Water quality improvements. If you were paying for ex
improvements in water quality from a land based activity to protect t
Reef, which of the following three options would you choose
The impact the reduction will have on reef health by 2050
This option will stay the same through out the range of choices presented to you
Percent of Sediment reduced by 2020
Percent of nutrient reduced by 2020
Area of inshore coral reef in good health by 2050
Current condition: about 68% of inshore in good condition (2,
km)
Hold much each alternative will cost ($ per year until 2020)
I would choose
Current Trends
0%
+
0%
–
About 50% inshore coral reefs in good health
sq km)
$0
Option one
16%
+
20%
→
About 54 % inshore coral reefs in good health (2,280 sq km)
$25
X
How much it would cost per household for the next five years
Option two
4%
+
30%
→
About 62% inshore coral reefs in good health (2,620 sq km)
$30
Different choices you need to consider
This is the 1st of 6 choice questions
SAMPLE

Question One: Water quality improvements. If you were paying for extra improvements in water quality from a land based activity to protect the Great Barrier Reef, which one of the following three options would you choose

Land based activities	Percent of Sediment reduced by 2020		Percent of nutrient reduced by 2020		Area of inshore coral reef in good health by 2050 *Current condition : About 68% of inshore in good condition (2,870 sq km)*	How much each alternative will cost ($ every year until 2020)	I would choose
Current Trends	0%	+	0%	→	About 50% inshore coral reefs in good health (2110 sq km)	$0	☐
Grazing	16%	+	20%	→	About 54 % inshore coral reefs in good health (2,280 sq km)	$20	☐
Sugar cane	4%	+	30%	→	About 62% inshore coral reefs in good health (2,620 sq km)	$30	☐

This is the 1st of 6 choice questions

Question Two: Water quality improvements. If you were paying for extra improvements in water quality from a land based activity to protect the Great Barrier Reef, which of the following three options would you choose

Land based activities	Percent of Sediment reduced by 2020		Percent of nutrient reduced by 2020		Area of inshore coral reef in good health by 2050 *Current condition : About 68% of inshore in good condition (2,870 sq km)*	How much each alternative will cost ($ every year until 2020)	I would choose
Current Trends	0%	+	0%	→	About 50% inshore coral reefs in good health (2110 sq km)	$0	☐
Grazing	12%	+	18%	→	About 50 % inshore coral reefs in good health (2,280 sq km)	$20	☐
Sugar cane	6%	+	32%	→	About 60% inshore coral reefs in good health (2,620 sq km)	$30	☐

This is the 2nd of 6 choice questions

Question Three: Water quality improvements. If you were paying for extra improvements in water quality from a land based activity to protect the Great Barrier Reef, which of the following three options would you choose

Land based activities	Percent of Sediment reduced by 2020		Percent of nutrient reduced by 2020		Area of inshore coral reef in good health by 2050 Current condition : About 68% of inshore in good condition (2,870 sq km)	How much each alternative will cost ($ every year until 2020)	I would choose
Current Trends	0%	+	0%	→	About 50% inshore coral reefs in good health (2110 sq km)	$0	☐
Grazing	8%	+	10%	→	About 27 % inshore coral reefs in good health (2,280 sq km)	$13	☐
Sugar cane	2%	+	15%	→	About 31% inshore coral reefs in good health (2,620 sq km)	$15	☐

This is the 3rd of 6 choice questions

Question Four: Water quality improvements. If you were paying for extra improvements in water quality from a land based activity to protect the Great Barrier Reef, which of the following three options would you choose

Land based activities	Percent of Sediment reduced by 2020		Percent of nutrient reduced by 2020		Area of inshore coral reef in good health by 2050 Current condition : About 68% of inshore in good condition (2,870 sq km)	How much each alternative will cost ($ every year until 2020)	I would choose
Current Trends	0%	+	0%	→	About 50% inshore coral reefs in good health (2110 sq km)	$0	☐
Grazing	12%	+	15%	→	About 15 % inshore coral reefs in good health (2,280 sq km)	$15	☐
Sugar cane	3%	+	23%	→	About 38 % inshore coral reefs in good health (2,620 sq km)	$23	☐

This is the 4th of 6 choice questions

Question Five: Water quality improvements. If you were paying for extra improvements in water quality from a land based activity to protect the Great Barrier Reef, which of the following three options would you choose

Land based activities	Percent of Sediment reduced by 2020		Percent of nutrient reduced by 2020		Area of inshore coral reef in good health by 2050 Current condition : About 68% of inshore in good condition (2,870 sq km)	How much each alternative will cost ($ every year until 2020)	I would choose
Current Trends	0%	+	0%	→	About 50% inshore coral reefs in good health (2110 sq km)	$0	☐
Grazing	32%	+	3%	→	About 43 % inshore coral reefs in good health (2,280 sq km)	$20	☐
Sugar cane	8%	+	13	→	About 50% inshore coral reefs in good health (2,620 sq km)	$24	☐

This is the 5th of 6 choice questions

Question Six: Water quality improvements. If you were paying for extra improvements in water quality from a land based activity to protect the Great Barrier Reef, which of the following three options would you choose

Land based activities	Percent of Sediment reduced by 2020		Percent of nutrient reduced by 2020		Area of inshore coral reef in good health by 2050 Current condition : About 68% of inshore in good condition (2,870 sq km)	How much each alternative will cost ($ every year until 2020)	I would choose
Current Trends	0%	+	0%	→	About 50% inshore coral reefs in good health (2110 sq km)	$0	☐
Grazing	16%	+	20%	→	About 54 % inshore coral reefs in good health (2,280 sq km)	$25	☐
Sugar cane	4%	+	30%	→	About 62% inshore coral reefs in good health (2,620 sq km)	$30	☐

This is the 6th of 6 choice questions

Q.12 Do you work or volunteer in any of the following fields?

Please tick all the relevant boxes

- ☐ Environmental planning and policy
- ☐ Environmental conservation
- ☐ Sugarcane or ethanol industry
- ☐ Grazing or beef industry
- ☐ Marine industries
- ☐ Commercial fishing
- ☐ Tourism

Considering the choices you have just made, please score the following statements from (1) STRONGLY AGREE to (5) STRONGLY DISAGREE

I am confident that I made the correct choices	1	2	3	4	5
I understood the information in the questionnaire	1	2	3	4	5
I needed more information than was provided	1	2	3	4	5
I found the choice options to be credible	1	2	3	4	5
I found the choice options confusing	1	2	3	4	5
Cost was not important in the choices I made	1	2	3	4	5

In this final section we would like to ask you a few questions to make sure the people we are surveying come from a wide range of backgrounds.

Do you have any children?

Yes ☐ No ☐

What is the highest level of education you have obtained or are obtaining?

☐ Primary only

☐ Junior/Year 10

☐ Secondary/Year 12

☐ Diploma or trade certificate

☐ Tertiary degree

☐ Other (please specify)

Over the years, when you have heard about proposed projects where there is a conflict between development and the environment, have you tended to:

☐ Favour preservation of the environment more frequently

☐ Favour development more frequently

☐ Favour development and environmental preservation equally

Do you have confidence in Government agencies to enforce conservation measures

☐ Yes – Vey confident

☐ Yes – Mildly confident

☐ No

How would you rate your knowledge of the issues addressed in this survey on a scale of **1** to **10** from **(1)** having NO knowledge to **(10)** having EXTENSIVE knowledge.
Rating: _____

To the best of your knowledge please indicate the total **weekly** income (**before taxes**) that you and your spouse (if applicable) currently earn.

- ☐ less than $499 per week ($25,999 per year)
- ☐ $500 – $799 per week ($26,000 – $41,599 per year)
- ☐ $800 – $1199 per week ($41,600 – $62,399 per year)
- ☐ $1200 – $1999 per week ($62,400 – $103,999 per year)
- ☐ $2000 or more per week ($104,000 per year)

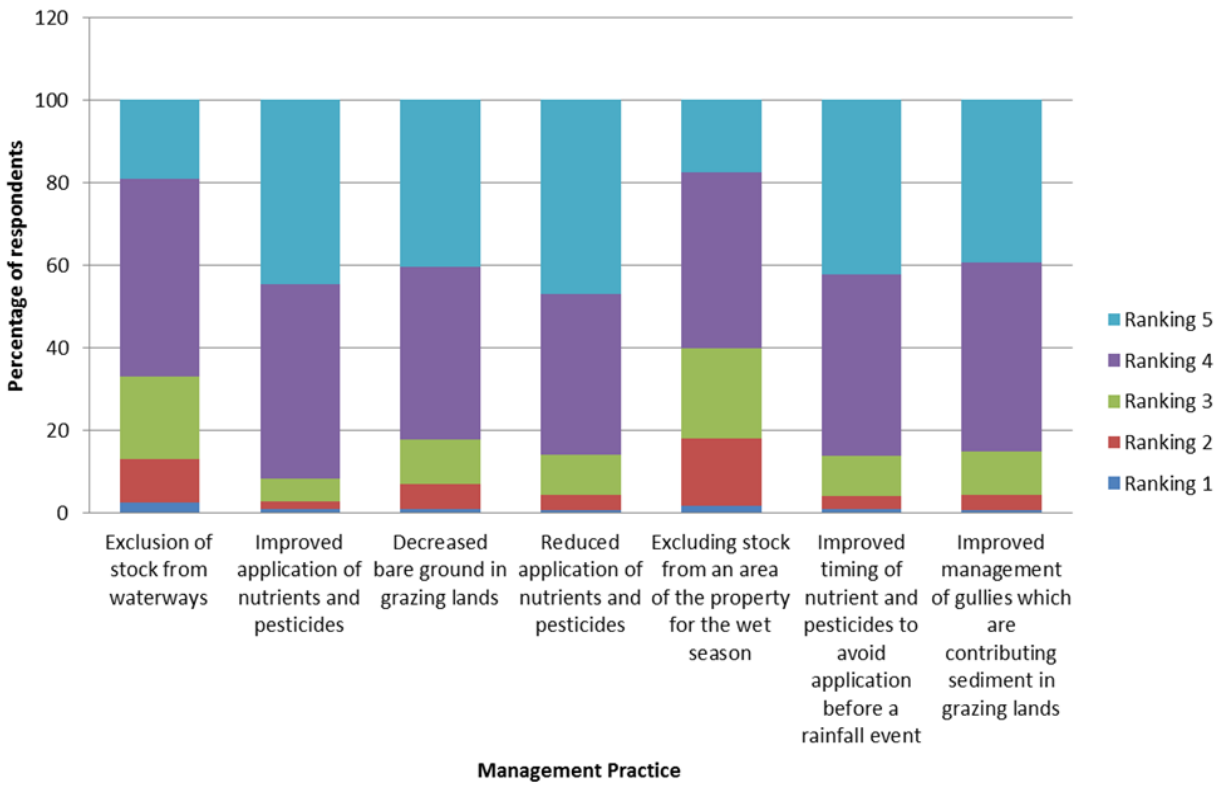
APPENDIX E: FOLLOW UP QUESTIONS FROM THE CHOICE MODELLING EXPERIMENT

Ranking of confidence, credibility and understanding

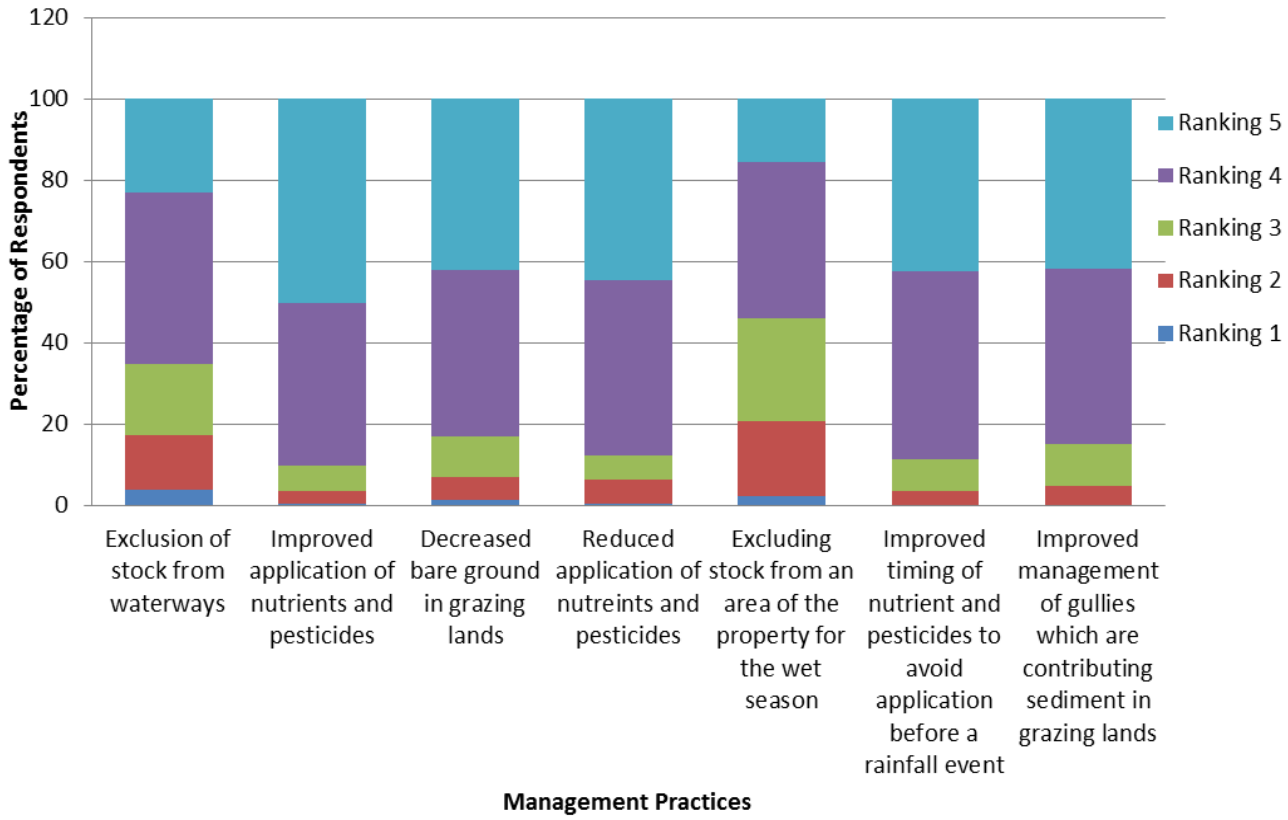
Labelled with water quality path Confidence the correct choices were made ranked from 1 (strongly agree) to 5 (strongly disagree)	1	2	3	4	5
I am confident that I made the correct choices	22	33	20	19	6
I understood the information in the questionnaire	26	31	15	21	6
I needed more information than was provided	5	26	20	34	15
I found the choices to be credible	11	32	27	21	4
I found the choice options confusing	8	25	23	29	15
Cost was not important in the choices I made	11	21	16	30	21

Labelled no water quality path

Confidence the correct choices were made ranked from 1 (strongly agree) to 5 (strongly disagree)	1	2	3	4	5
I am confident that I made the correct choices	22	36	22	16	3
I understood the information in the questionnaire	23	39	16	17	5
I needed more information than was provided	9	27	24	27	14
I found the choices to be credible	15	32	23	26	4
I found the choice options confusing	10	23	17	35	15
Cost was not important in the choices I made	15	21	13	30	21



Ranking of management practices for the labelled with water quality path sample



Ranking of management practices for the labelled no water quality path sample

Percentage of respondents ranking components of the Reef

To rank (from 1 not important) to 5 (very important) the various components of the Reef which they thought were important					
Ranking	1	2	3	4	5
Water Quality Path					
Coral Reefs	0	1	3	21	75
fish	0	1	3	29	66
seagrass	1	3	7	34	55
marine turtles	1	1	4	28	66
dugongs and dolphins	0	2	4	25	70
seabirds	1	4	8	35	53
No Water Quality Path					
Coral Reefs	0	1	1	22	76
fish	0	0	3	30	67
seagrass	0	3	6	32	58
marine turtles	0	1	3	27	69
dugongs and dolphins	0	1	2	26	70
seabirds	0	4	8	32	56

Percentage of respondents ranking their confidence, credibility, and information

Unlabelled with water quality path					
Confidence the correct choices were made ranked from 1 (strongly agree) to 5 (strongly disagree)					
	1	2	3	4	5
I am confident that I made the correct choices	24	36	18	15	7
I understood the information in the questionnaire	31	29	15	17	7
I needed more information than was provided	12	27	17	31	12
I found the choices to be credible	15	33	24	23	5
I found the choice options confusing	10	22	15	35	19
Cost was not important in the choices I made	15	22	10	35	18
Labelled with water quality path					
I am confident that I made the correct choices	22	33	20	19	6
I understood the information in the questionnaire	26	31	15	21	6
I needed more information than was provided	5	26	20	34	15
I found the choices to be credible	11	32	27	21	4
I found the choice options confusing	8	25	23	29	15
Cost was not important in the choices I made	11	21	16	30	21

Un-labelled with water quality path					
The importance of the following management practices from 1 (not important) to 5 (very important)					
	1	2	3	4	5
Exclusion of stock from waterways	3	13	16	49	18
Improved application of nutrients and pesticides	1	4	3	45	48

Decreased bare ground in grazing lands	2	4	7	44	44
Reduced application of nutrients and pesticides	1	4	5	41	49
Excluding stock from an area of the property for the wet season	5	15	19	40	21
Improved timing of nutrient and pesticides to avoid application before a rainfall event	1	4	7	42	46
Improved management of gullies which are contributing sediment in grazing lands	1	4	8	47	40
Labelled with water quality path					
Exclusion of stock from waterways	3	10	20	48	19
Improved application of nutrients and pesticides	1	2	5	47	45
Decreased bare ground in grazing lands	1	6	11	42	40
Reduced application of nutrients and pesticides	1	4	10	39	47
Excluding stock from an area of the property for the wet season	2	16	22	42	18
Improved timing of nutrient and pesticides to avoid application before a rainfall event	1	3	10	44	42
Improved management of gullies which are contributing sediment in grazing lands	1	4	10	46	39

Unlabelled with water quality path					
To rank (from 1 not important) to 5 (very important) the various components of the Reef which they thought were important					
Coral Reefs	1	1	1	23	74
fish	1	2	3	27	68
seagrass	1	3	3	31	62
marine turtles	0	1	3	26	69
dugongs and dolphins	1	1	2	24	73
seabirds	1	3	5	34	57
labelled					
Coral Reefs 3	0	1	3	21	75
fish	0	1	3	29	66
seagrass	1	3	7	34	55
marine turtles	1	1	4	28	66
dugongs and dolphins	0	2	4	25	70
seabirds	1	4	8	35	53