Incorporating socio-economic considerations into systematic conservation planning

PhD thesis submitted by
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Abstract

A primary focus of conservation efforts has been the establishment of protected areas, both by governments and conservation non-government organizations, to meet conservation commitments. While most research to date has neglected considerations other than biodiversity, there is a growing recognition that the socio-economic context in which conservation actions are ultimately applied influences the success of implementation. This is reflected by increasing prominence in the literature of considerations of conservation costs. However, despite the increased focus on cost-efficiency in conservation, there is still a lack of critical guidance on techniques for estimating costs and the implications for conservation outcomes of using untested surrogates for actual costs. The inclusion of appropriate cost measures in conservation planning is not only of academic interest but is relevant to ensuring that conservation plans accurately reflect the social context and provide relevant policy advice to government and conservation practitioners. The current knowledge gaps include appropriate methodologies for cost estimation, ways of costing multiple conservation actions, and consideration of costs that vary between groups of stakeholders. These gaps must be filled to ensure that conservation plans translate into effective on-ground actions that are adequately funded and supported by local communities.

The primary objective of my thesis is to identify the appropriate costs of conservation given particular socio-economic goals and the likely associated conservation actions, such as acquisition of land for national parks or engaging stakeholders in private land conservation agreements. My thesis research explores these issues for three different types of conservation planning exercises and then develops and applies rigorous economic methods for estimating the costs of conservation. The three kinds of conservation planning exercises are:

1. Acquiring land in a well defined land market for extension of an existing protected area system, when acquisition costs are a key consideration;
2. Protecting areas in the absence of a well defined market, when opportunity costs are a better measure of the costs of conservation;
3. Conserving land on private property through changes in land management that involve incremental changes in management costs.

The second objective of my thesis is to incorporate costs to multiple stakeholders and account for multiple conservation strategies and stakeholder preferences. Most conservation plans will involve engaging with a range of stakeholders and implementing different conservation strategies, ranging from acquisition to stewardship programs with private landholders. To accurately reflect the true costs associated with conservation, the variable costs to different stakeholders associated with different strategies must be accounted for explicitly.

My third objective is to incorporate uncertainties in my cost estimates into robust budgeting and decision making. Studies to date have not addressed how uncertainties associated with cost assumptions can drive budget estimates and shape conservation outcomes. Uncertainties arise from many factors which remain unexplored in the literature. In addition, given an understanding of the uncertainties associated with cost estimates, there is no guidance in the literature on how to make robust decisions, that is, decisions that still give an acceptable outcome despite uncertainty in parameters. Robust decisions require better accounting for uncertainty in the decision making process.

In Chapters 2 and 3, I examine the most common application of conservation planning in which areas are selected for full protection in a fully developed land market and therefore the cost of conservation is the acquisition cost of land at market value. I use the expansion of the protected area system in Queensland as a case study for this planning context. The government has committed to adding ~12 million ha to the reserve system, ~4 million ha of which will be in national parks, bringing the total area to 20 million ha by 2020. But there have been no details released on explicit conservation objectives, financial allocations, or formal budgeting. I estimate the costs associated with acquisition of land for national parks by first estimating sales value for all properties with a geographically weighted regression model. I then use this model, and varying values for several key factors, to incorporate uncertainties in my cost estimates into robust budgeting and decision making. I use two methods to explore uncertainty: sensitivity analysis and information-gap modelling.

In Chapter 2, I explore the challenge of budgeting *a priori* for protected area expansion in the face of uncertainty, specifically considering the future expansion of protected areas in

Queensland, Australia. I use Marxan to estimate the costs of potential reserve designs with data on actual land value, market value (including land and improvements), transaction costs, and tenure status. With scenarios, I explore three sources of budget variability: size of biodiversity objectives; subdivision of properties; and legal acquisition routes varying with tenure. Depending on the assumptions made, the estimated budget varies from $214 million to $2.9 billion. I demonstrate how a rigorous analysis can inform discussions about the expansion of systems of protected areas, including the identification of factors that drive budget variability over time.

In Chapter 3, I again use the Queensland protected area expansion as a case study and apply an info-gap model as a second method for addressing uncertainty. In contrast to Chapter 2, in which I use Marxan to select comprehensive sets of properties, this chapter applies a common prioritization strategy by calculating benefit-cost ratios and targeting those properties with the highest ratio value. In recent applications of BCRs, surrogate measures of the full benefits and costs of conservation projects, such as measuring area of vegetation types or using land value as a surrogate for acquisition cost, have been used. In this chapter I demonstrate the potential limitations of applying BCRs with surrogate measures. I compare the performance of BCRs based on two surrogate measures of both benefits and costs with the performance of prioritization based only on benefits. In addition, I explore the robustness of my BCR allocation, defined as the proportion of properties misallocated for investment due to uncertainties about the benefits and costs of projects, using an information-gap model (info-gap). I find that BCR allocations are more efficient than benefit allocations, and protect more regional ecosystems at a minimum level of 10%, and are generally more robust than benefit-only allocations.

The second conservation planning context that I examine is the case of selecting areas for full protection in regions where markets are not fully developed and therefore opportunity costs are a better measure of the full costs of conservation. In Chapters 4 and 5, I consider opportunity costs in marine and terrestrial contexts. In Chapter 4, I consider the planning region of Kubulau district, Vanua Levu, Fiji Islands, where the community is considering extending or re-configuring the marine protected area (MPA) network. I present a novel method for calculating the opportunity costs to fishers from their displacement by MPAs. I model opportunity costs as a function of food fish abundance and probability of catch, based on gear type and market value of

species. I also develop a profit model to investigate the effect of restricted access to transport on costs to fishers. I then include the opportunity cost model in Marxan to examine potential MPA configurations for Kubulau District, Fiji Islands, to minimize costs to local communities. I identify optimum areas for protection in Kubulau with: (a) the current MPA network locked in place; and (b) a clean-slate approach.

In Chapter 5, I use the Mbaracayu Forest Biosphere Reserve in Paraguay as a case study. The biosphere reserve has been identified as a high priority area for conservation interventions compared to other similar regions in Paraguay. I chose the biosphere reserve as my study region because of the availability of high-resolution data on costs and vegetation types. In this chapter I partition land costs into three distinct opportunity costs to smallholder agriculture, soybean agriculture, and ranching. I demonstrate that opportunity costs to single stakeholder groups can be inaccurate measures of true opportunity costs and can inadvertently and disproportionately displace conservation costs to groups of stakeholders who have not been considered. Additionally, I examine how spatial correlations between costs as well as size of conservation objectives affect the performance of opportunity costs to single stakeholder groups as surrogate measures of true opportunity costs. I conclude that planning with opportunity costs to single stakeholder groups can result in cost burdens to other groups that could undermine the long-term success of conservation. Thus, an understanding of the spatial distributions of opportunity costs that are disaggregated to groups of stakeholders is necessary to make informed decisions about priority conservation areas.

In Chapters 6 and 7, I consider the third conservation planning context: the case of selecting areas, not for full protection, but for participation in incentive programs that might only partially reduce production or require only marginal changes to management regimes. In Chapter 6, I estimate the potential costs of a stewardship program in the Daly River catchment, Northern Territory, which would underwrite the cost difference between routine land management and the additional requirements of conservation management. Based on survey responses from landholders, I first assess the current costs of land management in the catchment and use regression to identify key drivers of spatial variation in both routine land management costs and total land management costs (defined as land management costs to meet objectives for both

routine property management and conservation). I then estimate the additional costs of conservation management over and above routine land management at an average of $1.99 per ha. Much of the cost-effectiveness of stewardship payments would come from their ability to leverage the costs of routine land management already met by landholders.

In Chapter 7, I apply the cost estimates developed in Chapter 6 but also look at additional design criteria for a stewardship program in the Daly Catchment including the probability of landholders participating in the program. I use choice modelling to estimate the probability of participation in two legal arrangements - conservation covenants and management agreements - based on payment level and proportion of property required for management under the agreement. I then spatially predict landholders' probability of participating at the resolution of individual properties and incorporate these spatial predictions into conservation planning software to examine the potential for the stewardship program to meet conservation objectives. I find that there is a tension between planning for a cost-effective program and planning for a program that targets properties with the highest probability of participation.

Overall, my thesis highlights the importance of stating explicit socio-economic objectives. Explicit objectives are at the heart of systematic conservation planning; however, objectives associated with social and economic values are often neglected. The recent rush to include socio-economic costs in conservation planning has brought to the forefront the importance of considering the social context for conservation, and the lack of established methods for accounting effectively for costs underlies much of the current debate in the literature. Of the studies that have included cost metrics, relatively few link them to explicit objectives and most have not been based on rigorous economic methods. My thesis provides techniques for rigorously estimating costs in three broad types of situations that have to be addressed in conservation planning, while also accounting for uncertainties in cost estimates.

Publications associated with this thesis

Publications

Improving social acceptability of marine protected area networks: a method for estimating opportunity costs to multiple gear types in both fished and currently unfished areas. **Adams, V.M.**, Mills, M., Jupiter, S.D., Pressey, R.L. Biological Conservation 144: 350-361. (Chapter 4)

Opportunity Costs: Who really pays for conservation? **Adams, V.M.**, Pressey, R.L., Naidoo, R. Biological Conservation 143: 439–448. (Chapter 5)

An info-gap model to examine the robustness of cost-efficient budget allocations. **Adams, V.M.**, Pressey R.L. pp. 971-979. ICVRAM 2011: 1st international conference on vulnerability and risk assessment and management. University of Maryland, College Park. (Chapter 3)

How much does it cost to expand a protected area system? Some critical determining factors and ranges of costs for Queensland. **Adams, V.M.**, Segan, D.B., Pressey, R.L. PLoS ONE, 6, e25447. doi:25410.21371/journal.pone.0025447. (Chapter 2)

Applications and limitations of benefit-cost ratios in conservation planning. **Adams, V. M.**, and R. L. Pressey. Conservation Letters, *In Review*. (Chapter 3)

Estimating land management costs: The first step in designing a stewardship program for the Northern Territory. **Adams, V.M.**, Pressey, R.L., Stoeckl, N. Biological Conservation, *In Review.* (Chapter 6)

Estimating landholders' probability of participating in a stewardship program and implications for spatial conservation priorities. **Adams, V.M.**, Pressey, R.L., Stoeckl, N. Biological Conservation, *In Review.* (Chapter 7)

Reports

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Conference Presentations

V.M. Adams, M. Mills (presented), S.D. Jupiter, R.L. Pressey. 2011. Improving social acceptability of marine protected area networks: a method for estimating opportunity costs to multiple gear types. International Marine Conservation Congress, Victoria, British Columbia, Canada.

V.M. Adams, R.L. Pressey. 2011. An info-gap model to examine the robustness of cost-efficient budget allocations. International Conference on Vulnerability and Risk Analysis and Management (ICVRAM), Hyattsville, Maryland, USA.

V.M. Adams, R.L. Pressey, N. Stoeckl. 2010. Applying choice modelling to design payment programs for off-reserve conservation in the Northern Territory, Australia. Ecological Society of Australia, Canberra, Australia.

V.M. Adams, R.L. Pressey, N. Stoeckl. 2010. Designing payment programs for off-reserve conservation in the Northern Territory, Australia. International Congress for Conservation Biology, Edmonton, Alberta, Canada.

V.M. Adams, D.B. Segan, R.L. Pressey. 2009. Extending a protected area system: How do we make wise conservation investments? 10th International Congress of Ecology, INTECOL, Brisbane, Australia.

V.M. Adams, D.B. Segan, R.L. Pressey. 2009. Marine opportunity costs: a method for calculating opportunity costs to multiple stakeholder groups. 10th International Congress of Ecology, INTECOL, Brisbane, Australia.

V.M. Adams, D.B. Segan, R.L. Pressey. 2009. Marine opportunity costs: a method for calculating opportunity costs to multiple stakeholder groups. Fiji Islands Conservation Science Forum, Suva, Fiji Islands.

V.M. Adams, R.L. Pressey. 2008. How much does it cost to expand protected areas? Australian Protected Area Congress 2008, Sunshine Coast, Australia.

V.M. Adams, R.L. Pressey, R. Naidoo. 2008. Opportunity Costs: who really pays for conservation. Society for Conservation Biology, Chattanooga, Tennessee, USA.

Other publications produced during my candidature

Publications

Designing, implementing and managing marine protected areas: emerging trends and opportunities for coral reef nations. Ban, N.C., **Adams, V.M.**, Almany, G.R., Ban, S., Cinner, J., McCook, L.J., Mills, M., Pressey, R.L., White, A. Journal of Experimental Marine Biology and Ecology, *In press.*

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Chapter 1 General Introduction

There is growing recognition that the socio-economic context in which conservation actions are ultimately applied influences the success of their implementation. However, there is still a lack of critical guidance on techniques for estimating costs and the implications for conservation outcomes of using untested surrogates for actual costs. In this chapter I provide an introduction to the limitations of the body of literature that incorporates economic costs into conservation planning. I also provide an outline for my thesis, its objectives and structure.

Socio-economic Costs of Conservation: A systematic conservation planning approach

With increasing demands on natural resources such as clean water, food and fuel, it is becoming critical that we manage ecosystems to sustain the supply of these goods and services. A primary focus of conservation efforts to date has been the establishment of protected areas to meet the goals of agencies or non-government organizations or to meet global commitments through international mandates such as the Convention on Biological Diversity (IUCN 2009; UNEP 2002). Countries are increasingly establishing new protected areas to complement their existing networks[1] and so ensure protection of a representative set of all biodiversity features, including ecosystems, species and populations. The field of conservation planning focuses on the task of selecting areas for protection that contribute the most to conservation objectives at the least cost or within a set budgetary constraint (Margules & Pressey 2000).

It is widely recognized that conservation features are variable across space at broad scales (e.g. biodiversity hotspots or global 200 priorities Brooks et al. 2006) and within regions (e.g. Pressey 2004). Over the last 25 years or so, technical abilities to design systems of protected areas based on biological data have improved (Margules & Pressey 2000), but most research to date has neglected considerations other than biodiversity. This is despite the increasing recognition that the socio-economic context in which conservation actions ultimately take place has a large role in the determining the effectiveness of conservation management (Naidoo et al. 2006). This increased focus on the socio-economic context of conservation decisions is demonstrated by the addition of five stages to the systematic conservation planning framework of Margules and Pressey (2000), the new stages being focused predominantly on the social, economic and political context for the technical aspects of planning (Pressey & Bottrill 2009).

Naidoo et al. (2006) recently reviewed progress in including economic costs into conservation planning and highlighted key factors that make economic costs a central consideration. They also identified remaining research questions such as how to estimate and include economic costs in conservation planning. Conservation costs can be separated into five components: acquisition, management, opportunity, transaction, and damage (Naidoo et al. 2006). Acquisition costs are

[1] I use protected areas and reserves interchangeably throughout my thesis to refer to areas under protection meeting the IUCN class I-VI categories of protection.

the costs of acquiring a parcel of land and are therefore relevant to terrestrial conservation planning but much less so in the marine context. In addition, in the case of communally owned land acquisition costs are less relevant than in markets where land rights are assigned to a single individual. Acquisition costs can be estimated from historic land valuation data or, in developing countries where records can be less reliable, through the estimation of opportunity costs. Opportunity costs are costs of forgone opportunities and should reflect purchase value because they represent, in principle, the highest-value extractive use for a piece of land. I recognize that opportunity costs might be reflected in management and transaction costs. However, I conform to established usage in the conservation literature which focuses on opportunity costs from lost production (hereafter simply referred to as 'opportunity costs'). Opportunity costs are important in both marine and terrestrial planning and can be used to reflect the social costs of conservation actions to stakeholders by considering different extractive uses by stakeholder groups.

Management costs are those associated with the maintenance of a conservation program and can be broken down into fixed costs, which are independent of the amount of conservation effort, and variable costs, which are proportional to the type or amount of conservation management. Management costs can be a significant portion of total conservation costs and are therefore often considered in developing conservation plans. Transaction costs are those associated with negotiating an economic exchange, for example the legal and negotiation costs of purchasing a parcel of land from a private landholder for reservation. These costs can be substantial, particularly when negotiations last for several years; however, they are often difficult to estimate because they depend upon characteristics of individual landholders and the kinds of conservation arrangements being put in place. Damage costs are those associated with damages to economic activities from conservation management. For example, wildlife exiting a reserve might damage neighboring crop lands or livestock. Much like transaction costs, damage costs can be difficult to estimate and have seldom been recorded.

Cost data have been used in conservation planning during the process of selecting potential protected areas, by shifting selections toward those areas that are relatively inexpensive, while still achieving all conservation objectives (e.g. Polasky et al. 2001; Richardson et al. 2006; Stewart & Possingham 2005). Including costs in the selection of areas can save subsequent

extensive reconfiguration of initial plans when areas selected when ignoring costs prove to have less expensive replacements. Case studies have focused on acquisition, opportunity and management costs. A full accounting of conservation costs would include consideration of all five components, but this can be difficult or impossible because of limited data. The relative importance of these costs varies across terrestrial and marine environments, with acquisition costs being the most common type of cost data considered in terrestrial studies to date and opportunity costs the predominant cost data featured in marine studies (Ban & Klein 2009). In general, cost data should be carefully aligned with explicit cost objectives to ensure that the desired cost savings are delivered. For example, if the objective is to minimize purchase costs for the government, then acquisition costs would be the most appropriate measure. In comparison, if the objective is to minimize long-term costs to the management agency, then management costs would be a more accurate measure.

Systematic conservation planning can be described with an operational model in which data are gathered and summarized, explicit objectives are set, objectives and data are incorporated into a decision support tool, priority areas are selected, and conservation actions are then allocated to areas during implementation (Figure 1.1). As shown in Figure 1.1, because of the relative difficulty of estimating damage and management costs a priori, the ability to integrate and assess conservation costs depends on initial estimations and corrections at multiple stages in the planning process.

Figure 1.1 emphasizes that the first step in the systematic conservation planning process is stating explicit objectives for both conservation and economic goals. These objectives will then be translated into data layers that reflect the conservation and socio-economic features of concern and quantitative targets should be assigned to reflect potentially qualitative objectives. At the initial estimation stage it is likely that it will only be possible to estimate acquisition, opportunity and transactions costs because management and damage costs will depend on the extent and spatial configuration of areas selected for protection. The data would then be used in conjunction with a conservation decision support tool (I use Marxan throughout my thesis) to select potential reserve designs. Once potential configurations have been selected, management and damage costs can be estimated and the total costs of conservation can be assessed and

reserve designs can be altered iteratively if need be to reduce total cost estimates. After a final desktop assessment of potential reserves is completed, the conservation plan would then be used to guide on-ground conservation actions, which is likely to involve significant stakeholder engagement and reconfiguration of selected areas due to implementation constraints and opportunities that are revealed through this engagement (Knight & Cowling 2007; Pierce et al. 2005). In addition, implementation often occurs over a series of years. Therefore, over this time-frame, data may need to be re-assessed as on-ground information informs and updates previous desktop estimates, or as new data are collected, such as landholders' willingness to participate in conservation actions.

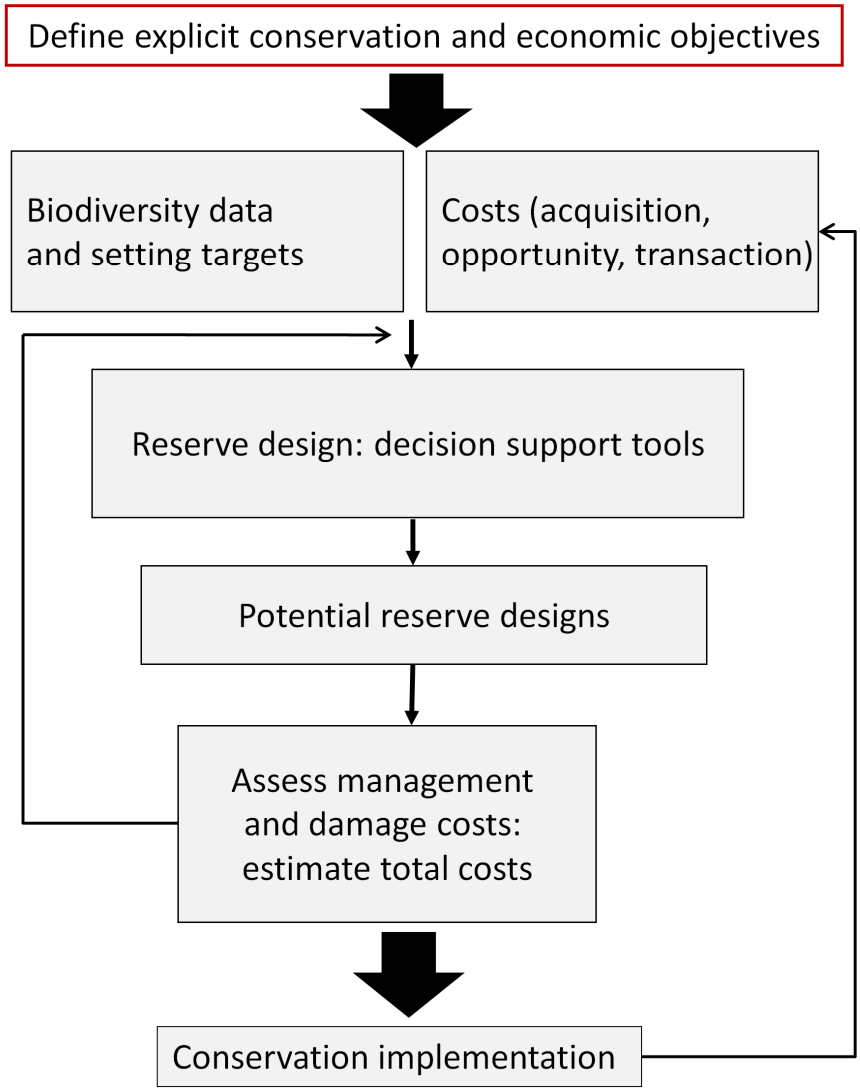


Figure 1.1 Operational model for systematic conservation planning from data scoping through conservation implementation.

Literature trends and research gaps

I identified all studies that included socio-economic costs in conservation planning with a Web of Science search ("cost" or "efficient" or "socio-economic" and "conservation" or "reserve" and "planning" or "site selection" or "prioritization" or "design"). I reviewed the 93 studies identified by my search and recorded the types of conservation costs considered, the cost estimation techniques used, and other relevant socio-economic variables considered, such as landholder characteristics. Based on these 93 studies, I examined trends in the number of annual publications that included socio-economic costs and trends in types of costs considered. I also compared the numbers of studies covering terrestrial and marine environments (Figure 1.2).

Following the seminal publication by Ando et al. (1998), which demonstrated that cost-effective selection strategies achieve the same species coverage at lower costs compared to solutions that seek to minimize the total area of selected sites, there has been a rapid increase in publications incorporating costs into conservation planning (Figure 1.2). As discussed by Ban & Klein (2009), the development of marine conservation planning has lagged behind terrestrial conservation planning. The first publication to include socio-economic costs in marine conservation planning was in 2003, 7 years after the first terrestrial publication that considered costs. To date, marine studies account for only 18% of all studies considering socio-economic costs in conservation planning.

Despite the increased focus on cost-efficiency in conservation (Figure 1.2), there is still a lack of critical guidance on cost estimation techniques and the implications of failing to use rigorous economic data for conservation planning (Naidoo et al. 2006; Polasky 2008). In a review of the literature, Newburn et al. (2005) addressed the question "Do conservation biologists currently offer effective methods to prioritize spending for conservation programs on private lands?". While their review of the literature focused on private lands specifically, their conclusions regarding the incorporation of economic factors into conservation planning are applicable more generally. They concluded that the literature still provided inadequate methods and rigorous guidance on rigorous derivation and application of cost data and highlighted trends in research gaps which I have summarized below (Gaps 1 and 2).

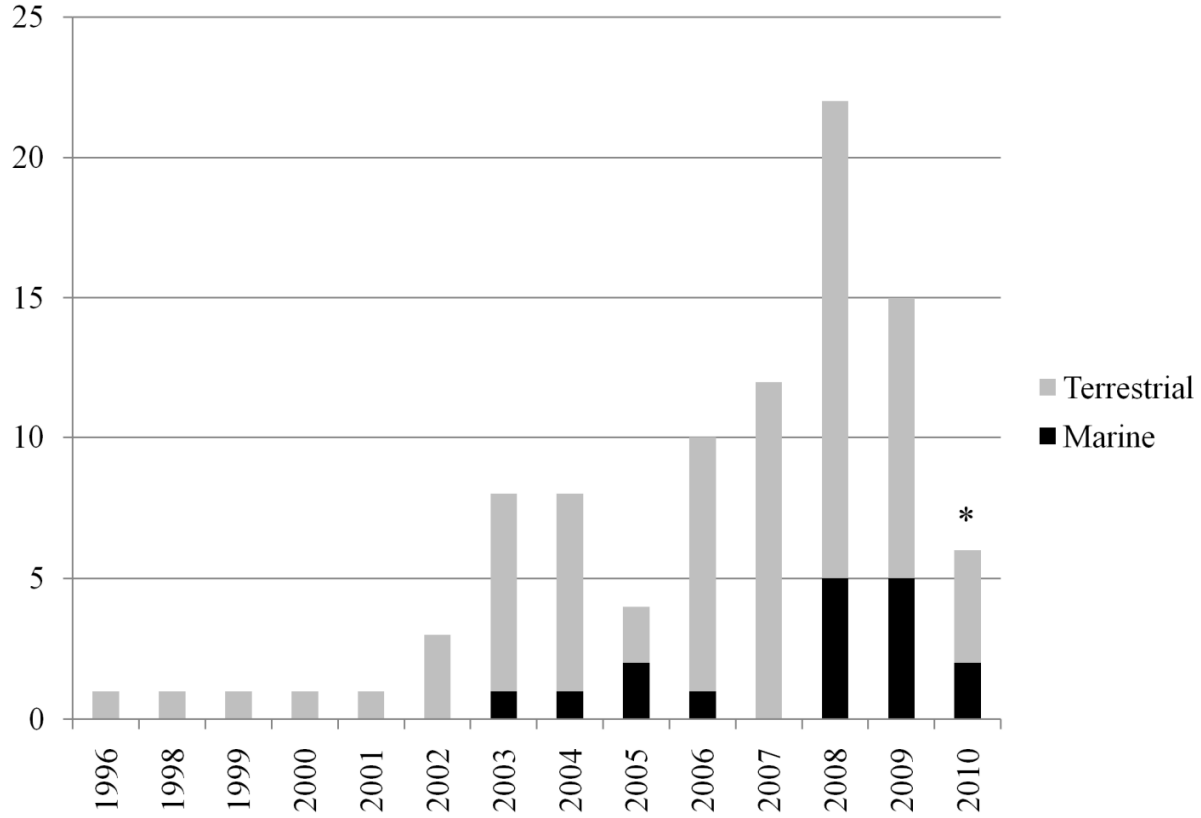


Figure 1.2 Number of publications that include socio-economic costs in conservation planning, by year, classified as marine or terrestrial studies. *This literature review was conducted in July 2010 and thus records for 2010 are incomplete.

I have complemented these with additional research gaps identified in my review of the literature (Gaps 3 and 4). The four important research gaps are:

1. The lack of appropriate methodologies, either developed uniquely to answer conservation problems or adapted from standard economic methods, to estimate components of conservation costs. For example, hedonic modeling is a standard economic method for estimating land values. However there is only one application of this method in the conservation literature. In other cases, where data have been derived with standard methods or from standard data sources such as land valuation data bases, there is still a strong tendency to use aggregate measures rather than fine-resolution metrics (for example averaging land values across regions instead of using individual property

values). The tendency to use aggregate measures may be in part due to conservation planners using standardized planning units rather than property boundaries. In addition, available land valuation data often have missing values or valuations at variable dates, so conservation planners who are unfamiliar with standard economic methodologies for estimating missing values choose to aggregate spatially instead.

2. The persistent focus of conservation planning on selecting priority areas for strict reservation, thereby neglecting other mechanisms, especially those applicable to private land, into decision making. Strict reservation is only one of many conservation options available and it is regularly complemented by off-reserve conservation on private lands. There is a need to account for private land conservation explicitly which means understanding the costs to private land holders. This includes using property boundaries in planning exercises to represent actual land holdings, understanding legal routes for acquisition or covenanting dependent on land tenure, and estimating variable costs associated with different types of private conservation such as opportunity and management costs.
3. The failure to explicitly link cost metrics to explicit socio-economic objectives. Explicit socio-economic objectives will reflect the social context of the planning region and thus are likely to reflect specific objectives regarding different stakeholder groups in study regions and might also reflect different conservation strategies. Therefore, it is important to understand how costs relate to socio-economic dimensions, in particular how to account for variable impacts to multiple groups of stakeholders on whom conservation costs fall differently.
4. The failure to address the impact of uncertainties associated with economic costs on financial estimates of achieving conservation objectives or the configurations of selected priority areas. Understanding how we take into account uncertainty in our estimates has two associated questions that remain unanswered: (a) How do we report on the uncertainties associated with our financial estimates? and (b) How do we make robust decisions given the uncertainties associated with our estimates?

I summarized the key research gaps identified by Newburn et al. (2005) and measured the percentage of studies covered by my review that addressed these issues prior to 2005 and after

2005 (Table 1.1). I categorized the studies as 'prior to' and 'after' Newburn et al. (2005) to determine whether the trends observed in 2005 have changed with the increased number of publications over the last five years. The only substantial change is a decrease in the percentage of studies using aggregated mean or median land values across large areas (Table 1.1). This indicates that the research gaps identified by Newburn et al. (2005) are not a dominant focus of studies that include socio-economic costs in conservation planning. Accordingly, little headway is being made to provide effective methods for prioritizing conservation spending based on sound economic methods.

As indicated by Table 1.1, the costs being included in conservation planning are still rudimentary cost estimates with more than a third of the studies using aggregated mean or median land values rather than implementing standard economic methods to estimate values of individual parcels. At the time of their study, Newburn et al. (2005) noted that no studies had implemented standard economic methods such as hedonic models, the most common economic technique for land valuation. In my review of the literature, only one study has implemented hedonic models for land valuation (Newburn et al. 2006). Additionally, relatively few studies include more than one component of conservation costs.

Table 1.1 A review of key trends in 93 studies identified in my search of the primary literature. I have measured percentage of studies 'Prior to' and 'After' Newburn et al. (2005) to analyze whether the focus of conservation planning studies has changed in the last five years.

Characteristic	Prior to	After
Incorporating costs based on land valuation data or standard economic methodologies	36%	35%
Estimating costs at fine resolution (e.g. single land parcels) versus aggregating to mean or median costs across large areas such as regions	56%	67%
Considering boundaries of individual land parcels	12%	13%
Considering private land conservation strategies	16%	12%

Thesis objectives

The primary objective of my thesis is to identify the appropriate costs of conservation given stated socio-economic goals and the likely associated conservation actions for three different types of conservation planning exercises and then to develop and apply rigorous economic

methods for estimating those costs. I achieve this by examining three types of planning exercises and estimating the appropriate costs: 1. Acquiring land in a well defined land market for extension of an existing protected area system (acquisition costs); 2. Protecting areas where there is not a well defined market and opportunity costs are a better measure of conservation costs; and 3. Conserving land on private property through changes in land management (incremental changes in management costs). The second objective of my thesis is to use cost estimates to help fill the other research gaps identified. In particular, I address how to incorporate costs to multiple stakeholders and how to account for multiple conservation strategies and stakeholder preferences. My third objective is to incorporate uncertainties in cost estimates into robust budgeting and decision making.

Objective 1: Develop cost metrics for inclusion in conservation planning based on standard economic methods

Consistent methods for estimating management and opportunity costs are still lacking in the studies reviewed. Naidoo & Adamowicz (2006) and Naidoo & Iwamura (2007) provided a method for estimating opportunity costs in terrestrial environments. However, this method has not been applied in other studies. Similarly, Balmford et al. (2003) provided a global study estimating management costs of terrestrial parks. A similar method has only been applied at a local scale by one study (Frazee et al. 2003). It is clear that further development and applications of these methods are needed, and rigorous economic methods such as those presented in these studies should be commonplace in studies rather than the exception. Furthermore, most studies to date have only considered one component of conservation costs, which fails to provide planners with a full picture of the financial costs associated with conservation actions. Consistent methods for accounting for all conservation costs are needed so that more rigorous financial estimates can be constructed for conservation plans.

Objective 2: Examine the impacts of socio-economic dimensions, such as accounting for tenure structures, stakeholder values and preference, and differential costs to stakeholders, on conservation costs and decisions.

The cost measures included in the planning process should be directly aligned with the cost objectives. Conservation plans will likely be implemented more effectively by working with a

range of conservation strategies, ranging from acquisition to stewardship agreements with private landholders, and requiring engagement with multiple stakeholders. The more information that is known a priori about stakeholder preferences or 'values' the better our estimates of costs of conservation can be. This is because actual costs will be impacted not only by market values but also by the perceived costs of conservation that influence stakeholders' willingness to participate in conservation agreements. In addition, the costs of conservation are likely to vary between stakeholder groups. Thus, an understanding of the spatial distributions of opportunity costs that are disaggregated to groups of stakeholders is necessary to make informed decisions about priority areas for conservation. To date, there has been relatively little research on variable costs between stakeholder groups or on stakeholder preferences for different conservation actions on their land. As these types of information are obtained, the more likely conservation planners are to achieve cost-efficient strategies.

Objective 3: Examine potential sources of uncertainty in cost estimates and assess the impacts of these uncertainties on financial budget estimates.

To date there has been some research on how to prioritize under constrained budgets, but we still know relatively little about how much it will cost, in total, to achieve conservation objectives. The lack of studies and methodologies for estimating a full financial accounting of protected area expansion remains a hurdle in applying conservation science to achieve real-world outcomes. In addition to the difficulties of estimating financial budgets for conservation, studies have not addressed how uncertainties associated with cost assumptions can shape budget estimates and conservation outcomes. Uncertainties arise from many sources, but the main factors that remain unexplored in the literature are: legal acquisition routes dependent on tenure; whether properties can be subdivided; and size of biodiversity objectives. An example of the first source of uncertainty is on freehold (private) land in Queensland where areas can be established either through acquisition at market value or voluntary establishment of a nature refuge which has no associated acquisition cost. Therefore, the costing of a parcel of land depends on its tenure and assumptions about the expected acquisition route, in turn depending on the preferences of landholders. No framework exists for budgeting for conservation actions while including a complete accounting of costs and associated uncertainties. A better accounting of conservation costs could help bridge the planning-implementation gap by helping to lobby for adequate funds.

In addition, given an understanding of the uncertainties associated with cost estimates, there is no guidance in the literature for how to account for this uncertainty to make robust decisions. It is critical to understand not only the potential uncertainties associated with cost estimates, but also how to make robust conservation decisions regardless of these uncertainties.

Thesis Outline

This thesis is presented as a series of chapters formatted for publication in peer-reviewed journals. Figure 1.3 shows the overall structure of the thesis. The six data chapters are structured as three sets of two chapters, with each set addressing a different socio-economic context and different type of cost accounting. The first chapter of each set addresses objective 1 by applying economic methods to estimate conservation costs. The second chapter in each set applies these costs in conservation planning to address objectives 2 and 3. Authorship of chapters for publication (Chapters 2-7) is shared with members of my thesis committee, Bob Pressey (Chapters 2-7) and Natalie Stoeckl (Chapters 6-7) as well as several contributing co-authors: Daniel Segan (Chapter 2), Morena Mills (Chapter 4), Stacy Jupiter (Chapter 4), and Robin Naidoo (Chapter 5). Several government agencies provided spatial data and I have identified and cited the custodians of data within the relevant chapters. Tables and figures are shown throughout the text and additional supporting methods and figures are provided in the appendices.

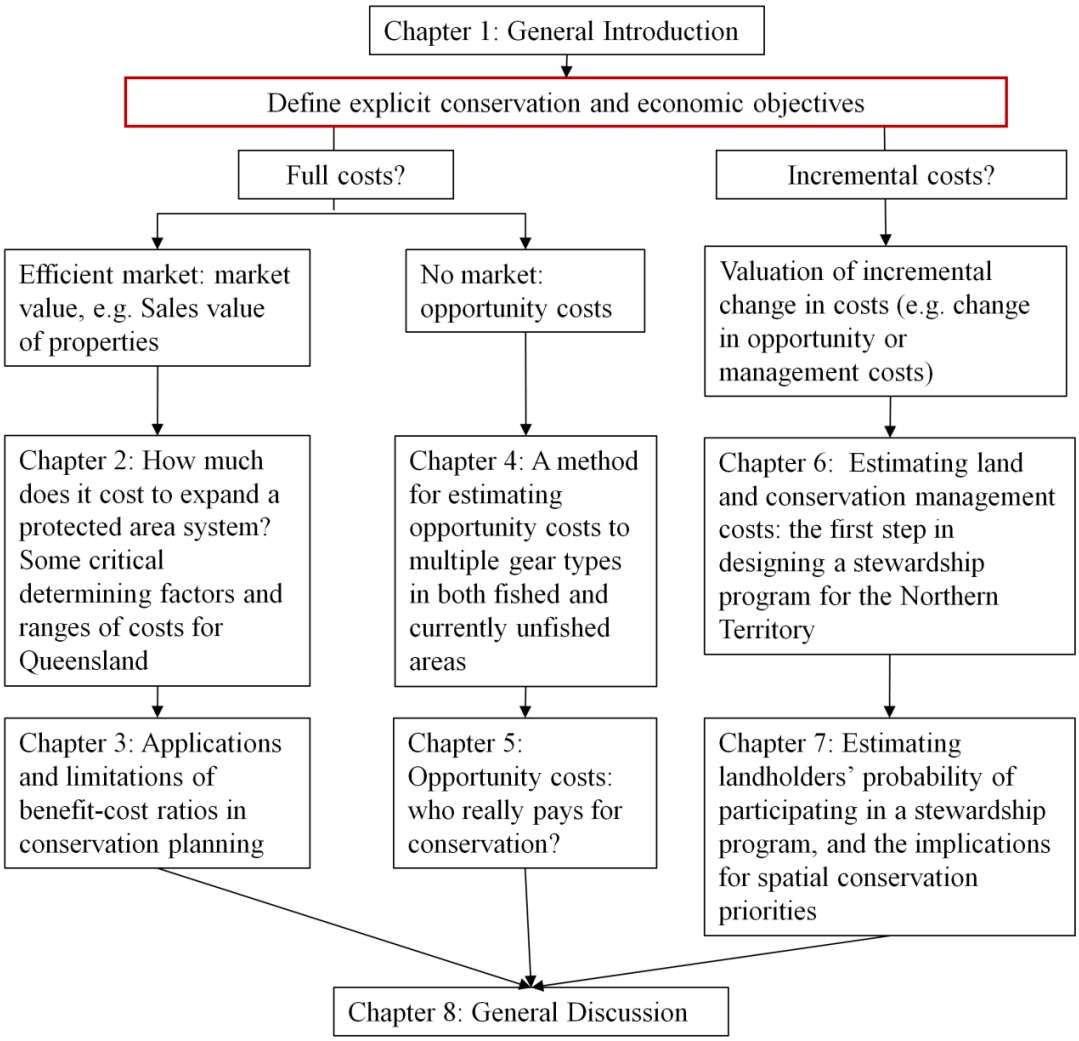


Figure 1.3 Chapter structure for this thesis.

Chapter 1 (this chapter) provides an introduction to the limitations of the body of literature that incorporates economic costs into conservation planning.

Chapter 2 conducts a sensitivity analysis on the financial budget for expanding Queensland's protected area estate, focusing on three sources of budget variability: legal acquisition routes dependent on tenure; subdivision of properties; and size of biodiversity objectives. Spatial data were provided under license by the Queensland Government Department of Environment and Resource Management. Daniel Segan assisted in compiling all spatial data and formatting the data for use in the conservation planning software Marxan. I conducted the analysis and wrote

the chapter, and Daniel Segan and Bob Pressey assisted in the interpretation of results and editing.

Publication:

How much does it cost to extend a protected area system? Some critical determining factors and ranges of costs for Queensland. Adams, V.M., Segan, D.B., Pressey, R.L. PLoS ONE, In Review.

Chapter 3 applies an information-gap model to explore the trade-offs in cost efficiency and robustness of budget allocations when benefit-cost ratios are used to allocate funds. The mathematical development of the information-gap model is published in the peer reviewed conference proceedings for the 2011 International Conference on Vulnerability, Risk Assessment and Management (ICVRAM 2011). The application of the info-gap model to expanding Queensland's protected area estate has been submitted for publication in a peer reviewed journal. Spatial data were provided under license by the Queensland Government Department of Environment and Resource Management. I conducted the analysis and wrote the chapter and Bob Pressey assisted in the interpretation of results and editing.

Publications:

An info-gap model to examine the robustness of cost-efficient budget allocations. Adams, V.M. and Pressey R.L. 2011. ICVRAM 2011: 1st International Conference on Vulnerability and Risk Assessment and Management, April 11-13, 2011, University of Maryland, College Park, pp.971-979.

Applications and limitations of benefit-cost rations in conservation planning. Adams, V. M., and R. L. Pressey. Conservation Letters, In Review.

Chapter 4 develops and applies a method for estimating opportunity costs to multiple stakeholders in a marine environment. Stacy Jupiter collected the field data in Kubulau district, Vanua Levu, Fiji Islands. Morena Mills helped conduct analysis of the catch per unit effort data collected by Stacy Jupiter. I conducted the analysis and wrote the chapter, and Stacy Jupiter, Morena Mills and Bob Pressey assisted with model interpretation and editing.

Publication:

Improving social acceptability of marine protected area networks: a method for estimating opportunity costs to multiple gear types in both fished and currently unfished areas. Adams, V.M., Mills, M., Jupiter, S.D., Pressey, R.L. 2011. Biological Conservation 144: 350-361.

Chapter 5 estimates the impacts on total costs as well as costs to other stakeholders when conservation planning only includes opportunity costs to a single stakeholder group. Robin Naidoo developed the spatial data used. I conducted the analysis and wrote the chapter, and Robin Naidoo and Bob Pressey assisted with interpreting the results and editing.

Publication:

Opportunity Costs: Who really pays for conservation? Adams, V.M., Pressey, R.L., Naidoo, R. 2010. Biological Conservation 143: 439–448.

Chapter 6 develops a model for land management costs in the Daly catchment, Northern Territory and uses this model to estimate the costs of a conservation stewardship program which would underwrite the difference between routine property management and conservation management. Spatial data were provided under license by the Northern Territory Department of Natural Resources, Environment, Arts and Sports. I collected all social data used to inform the models by mail surveys and in-person interviews. I conducted the analysis and wrote the chapter, and Natalie Stoeckl and Bob Pressey assisted in the survey design, model interpretation and editing.

Publication:

Estimating land management costs: The first step in designing a stewardship program for the Northern Territory. Adams, V.M., Pressey, R.L., Stoeckl, N. Biological Conservation, In Review.

Chapter 7 uses the model from Chapter 6 in conjunction with stakeholder preferences for different stewardship agreements to design a spatially explicit conservation plan that accounts for multiple conservation actions. Spatial data were provided under license by the Northern Territory Department of Natural Resources, Environment, Arts and Sports. I collected all social data used to inform the models by mail surveys and in-person interviews. I conducted the

analysis and wrote the chapter, and Natalie Stoeckl and Bob Pressey assisted in the survey design, model interpretation and editing.

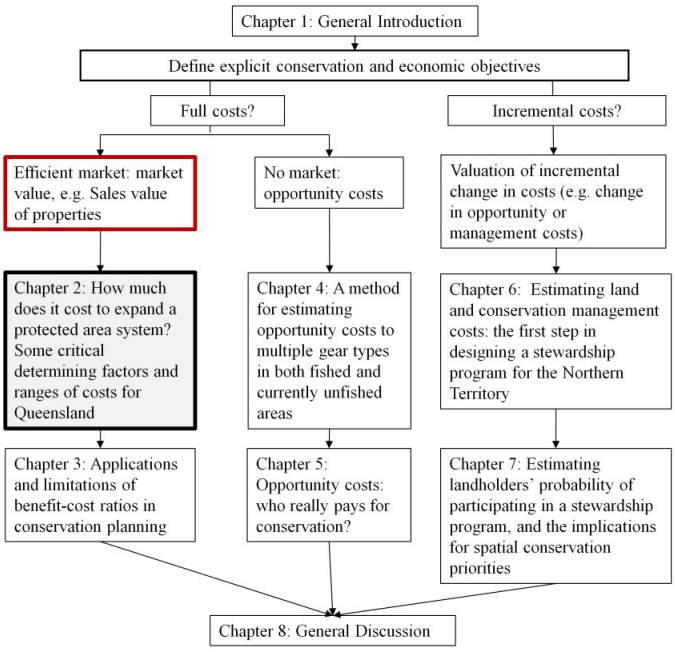
Publication:

Estimating landholders' probability of participating in a stewardship program, and the implications for spatial conservation priorities. Adams, V.M., Pressey, R.L., Stoeckl, N. In Review.

Chapter 8 provides a summary of the previous chapters and a discussion of the contributions made to advance the cross-disciplinary field of conservation planning with economic considerations. In this chapter, I also highlight the opportunities and constraints of addressing the research gaps identified in Chapter 1 and discussed through Chapters 2 – 7.

Chapter 2 How much does it cost to expand a protected area system? Some critical determining factors and ranges of costs for Queensland[1]

Many governments have recently gone on record promising large-scale expansions of protected areas to meet global commitments such as the CBD. As systems of protected areas are expanded to be more comprehensive, they are more likely to be implemented if planners have realistic budget estimates so that appropriate funding can be requested. Estimating financial budgets *a priori* must acknowledge the inherent uncertainties and assumptions associated with key parameters, so planners should recognize these uncertainties by estimating ranges of potential costs. I explore the challenge of budgeting *a priori* for protected area expansion in the face of uncertainty, specifically considering the future expansion of protected areas in Queensland, Australia. The government has committed to adding ~12 million ha to the reserve system, bringing the total area protected to 20 million ha by 2020. I used Marxan to estimate the costs of potential reserve designs with data on actual land value, market value, transaction costs, and land tenure. With scenarios, I explored three sources of budget variability: size of biodiversity objectives; subdivision of properties; and legal acquisition routes varying with tenure. Depending on the assumptions made, my budget estimates ranged from $214 million to $2.9 billion. Estimates were most sensitive to assumptions made about legal acquisition routes for leasehold land. Unexpected costs (costs encountered by planners when real-world costs deviate from assumed costs) responded non-linearly to inability to subdivide and percentage purchase of private land. I demonstrate how a rigorous analysis can inform discussions about the expansion of systems of protected areas, including the identification of factors that influence budget variability.



[1]Adams, V. M., D. B. Segan, and R. L. Pressey. 2011. How much does it cost to expand a protected area system? Some critical determining factors and ranges of costs for Queensland. PLoS ONE, 6, e25447. doi:25410.21371/journal.pone.0025447.

Introduction

International mandates such as the Convention on Biological Diversity have become prominent in debates about extending protected areas (UNEP 2002). Countries that sign the CBD commit to effectively protecting a portion of all their ecosystems. A recent study assessed progress towards the goal of protecting 10% of each ecoregion by 2010 and found that half of the world's ecoregions had not met this target (Jenkins & Joppa 2009). Around the Conference of the Parties 2010, there have been notable political promises for expanded protected areas. For example, in 2008, the Democratic Republic of Congo announced it would double its protected area extent to 30 million ha (Herkenrath 2008). More recently, the European Union promised to protect at least 20% of land by 2020, also doubling its current protected area estate (European Parliament 2010). Unfortunately there is often a gap between political promises and the actual funding available to achieve them. This could reflect the pervasive underfunding of conservation activities globally (James et al. 1999a). However, the shortfall in funding could also derive from the lack of comprehensive financial estimates of large-scale expansions of protected areas.

Constraints on funding for conservation have motivated global financial analyses to estimate the costs of conservation commitments (Balmford et al. 2003; Naidoo & Iwamura 2007). However, these have rarely been complemented by fine-scale estimates of the spatially variable conservation costs based on mapping of the ecosystems or other features that should be represented (but see Ferraro 2002; Frazee et al. 2003). Many planning studies have demonstrated that conservation objectives can be met more cheaply with data on spatially variable costs. However, these studies rely on many untested assumptions about factors influencing cost estimates. For example some studies apply global scale estimates to local scale problems (Carwardine et al. 2009; Kark et al. 2009) or assume acquisition of only native vegetation within properties (Carwardine et al. 2009; Klein et al. 2009c). To estimate the actual costs of expanding protected areas, planners must move beyond single estimates based on dubious assumptions. Several frameworks for considering uncertainty, both ecological and financial, have been proposed (Langford et al. 2009; Wilson et al. 2009) and the uncertainty of cost estimates associated with specific conservation actions has been assessed (e.g. Knoke et al. 2008; McCarthy & Lindenmayer 2007). However, there has been no systematic exploration of the uncertainties associated with estimating the cost of expanding protected areas.

Table 2.1 Factors known or likely to affect the acquisition costs of protected areas. The three factors considered directly in this study are shaded and bold. Factors in italics were considered indirectly through legal acquisition routes and subdivision of properties. Studies with asterisks estimated the effects on establishment costs as the number and/or total extent of conservation areas, which are likely to translate into effects on financial costs in all or most regions.

Factor	Notes	References
Amount of biodiversity data	More complex data increase the total extent of conservation areas required to achieve conservation objectives because of imperfect spatial correlations between features.	(Pressey et al. 2003; Pressey & Logan 1995; Rodrigues & Gaston 2001; Warman et al. 2004)*
Rarity and nestedness of species occurrences	Higher rarity of species (less spatial co-occurrence of species) increases the total extent of conservation areas required to represent them. Higher nestedness of species (more spatial co-occurrence of species) reduces the total extent of conservation areas required to represent them.	(Pressey et al. 1999; Rodrigues & Gaston 2001)*
Size of biodiversity objectives	Larger conservation objectives for features such as species and vegetation types increase the total extent and total cost of conservation areas needed to achieve them.	(Justus et al. 2008; Warman et al. 2004)* (Adams et al. 2010)
Size of planning units (considered with landholder willingness to subdivide property)	Smaller planning units require smaller total extents of conservation areas to achieve the same conservation objectives because they lead to less over-representation of objectives.	(Pressey & Logan 1998; Rodrigues & Gaston 2001)*
Spatial variability in costs of planning units	Efficiency gains of including costs in the planning process are strongly related to the relative variability of conservation costs.	(Pressey et al. 1999)* (Adams et al. 2010; Naidoo et al. 2006)
Spatial correlation between biodiversity values and costs	Efficiency gains of including costs in the planning process are strongly related to the correlation between conservation costs and benefits.	(Adams et al. 2010; Babcock et al. 1997; Naidoo et al. 2006)
Connectivity of conservation areas	Grouping planning units so that they achieve objectives for connectivity (e.g. compactness, alignment to provide movement corridors) increases the total extent of conservation areas required to achieve other conservation objectives such as representation of species and vegetation types.	(Nicholls & Margules 1993; Stewart & Possingham 2005)
Uncertainty about establishment costs of individual planning units	The actual establishment costs (e.g. opportunity or acquisition costs) of all planning units are seldom or never known with certainty, particularly across large regions. Typically, these costs must be estimated with surrogates (e.g. agricultural potential) or modeled from a limited number of data points (e.g. sales prices).	No studies have explicitly considered uncertainty of cost estimates, but several studies have developed frameworks for considering uncertainties (Langford et al. 2009; Wilson et al. 2009)
Legal acquisition routes for protection of different tenures	Depending on the tenure of land parcels, different legal routes are probably available for placing the parcel under protection (e.g. conservation easement or nature reserve programs for freehold land; stewardship requirements and payment programs for leasehold land). The total costs of achieving conservation objectives will vary strongly between different legal routes.	(Messer 2006)
Landholder willingness	Landholders vary in their inclination to engage with conservation organizations. Issues include willingness to sell, willingness to negotiate portions of properties to be sold (i.e. willingness to subdivide property for sale), and willingness to participate in nature refuge or conservation management programs.	(Knight et al. 2010; Messer 2006)

I identified ten factors likely to affect the cost of expanding protected areas (Table 2.1). Some of these relate to biological variables and are relatively well understood; but socio-political factors are also likely to be important determinants of costs. This study examines how three socio-political factors affect the expected cost of expanding protected areas in Queensland, Australia (Figure 2.1).

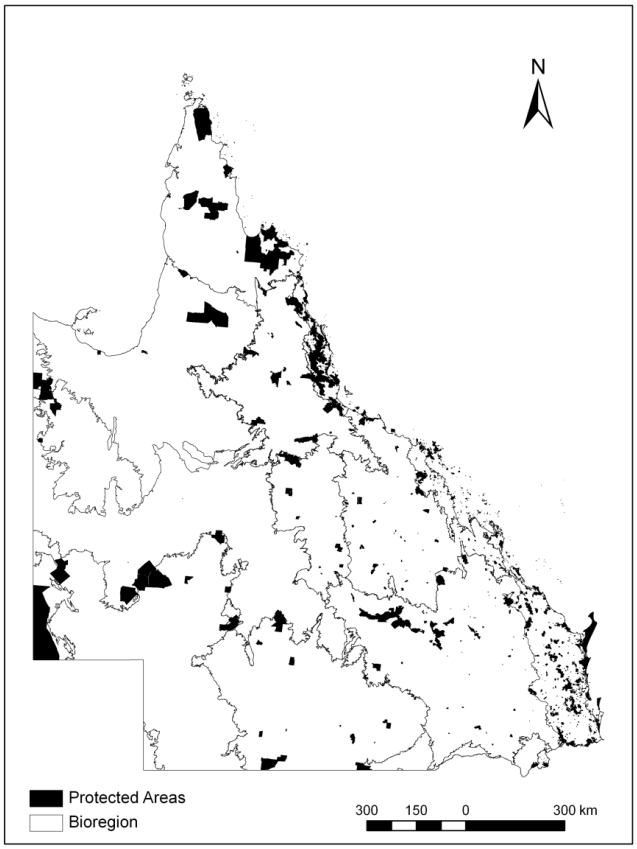


Figure 2.1 The state of Queensland, protected areas, and bioregions.

In 2008, the Queensland government promised to add ~12 million ha to the network of protected areas, 4 million ha of which would be acquired for national parks, bringing the total estate to 20 million ha by 2020 (Anna Bligh Premier of Queensland 2008). There is flexibility in how this promise could be fulfilled, and different approaches will have different financial implications. The current 7.6 million ha of national parks were acquired over 100 years. The promised expansion therefore represents an unprecedented rate of addition of protected areas.

The primary objective of my study was to conduct a financial analysis of how much it will cost to expand the Queensland protected area estate, accounting for uncertainties around assumptions that make a range of potential costs more useful than a single figure. I therefore used a sensitivity analysis to examine two aspects of uncertainty in financial estimates: 1. the possible range of financial budgets required, depending on different assumptions made, termed "expected costs" and 2. the possible "unexpected costs" given departures from assumptions in the face of real-world constraints. I focused my study on three factors: the size of biodiversity objectives; willingness to subdivide properties; and legal acquisition routes dependent on tenure (Table 2.2). I selected these factors to reflect socio-economic assumptions that involve considerable uncertainty in the expansion of protected areas in Queensland.

Table 2.2 Factors (3) included in this study, associated variables (4) used in my calculations, and ranges of values to indicate uncertainties. The full factorial design required 200 scenarios to consider all combinations of values (2x2x5x10).

Factor	Variable considered	Range of uncertainties
Biodiversity objectives	Size of objectives for regional ecosystems	10%/1,000 ha or scaled objectives (2 values)
Landholder willingness to subdivide property	No subdivision / subdivision	No subdivision requires acquisition of entire property. Subdivision allows for acquisition of only remnant vegetation (2 values)
Legal acquisition routes for protection of different tenures	Freehold acquisition routes	0-40% of properties purchased (in 10% increments), with the remainder placed in Nature Refuge (5 values)
	Leasehold acquisition routes under the Delbessie Agreement	0-90% of properties purchased (in 10% increments), 5% of property leases under terminal lease renewal, with the remainder placed in Nature Refuge (10 values)

I used sensitivity analysis because the promised rate of expansion of the protected area estate is unprecedented and not reflective of previous acquisitions. It was therefore not possible to derive accurate financial estimates from historic data. Historically, parks have not been located strategically, but rather in response to political imperatives, such as the Wet Tropics World Heritage Area, or based on ad hoc responses to availability of land and lengthy negotiations with landholders. Furthermore, future expansion of protected areas in Queensland will involve recent legal initiatives with uncertain applications. One of these initiatives is the Nature Refuge program which allows landholders to voluntarily place portions of their properties under

conservation covenants. The covenants are attached to land titles in perpetuity, stipulate nature conservation as the primary use and constitute IUCN category VI protected areas. Nature refuges therefore contribute to the national reserve system and to meeting global commitments such as the CBD. However, NatureAssist, which involves competitive bidding by landholders to support management of Nature Refuges, provided its first round of funding only in 2007. It is unclear whether the last 5 years of funding will reflect future funding, or whether this will be sufficient incentive to engage many more landholders. A second recent legal initiative is the Delbessie Agreement, legislated in the last 5 years and untested in implementation. The Delbessie Agreement is a framework of legislation, policies and guidelines supporting the environmentally sustainable, productive use of rural leasehold land for (DERM 2007). The lessees with properties identified as having conservation value can enter into a Nature Refuge agreement and be rewarded with a 10-year lease extension. Alternatively, they can elect to have their properties acquired. The percentage of lessees choosing either option is very difficult to estimate. Given the Government's commitment to a rapid expansion of protected areas in Queensland and these two relatively untried legal instruments, I set out to explore the effects on costs of uncertainties around several influential variables.

Methods

Planning region

The study region included the whole state of Queensland, Australia, with a total area of 185 million ha of which approximately 8 million ha is protected (Figure 1.2). I implemented my sensitivity analysis by estimating the total financial cost of the protected area expansion for incremental changes in four variables, associated with my three key factors (Table 2.2). I selected the four variables and appropriate values for them based on expert interviews conducted with the Queensland Department of Environment and Resource Management (DERM). I then used Marxan (Ball et al. 2009), a reserve design tool, to estimate the total financial costs for each scenario (see Reserve Design section). Marxan is a commonly used conservation planning software that uses a simulated annealing algorithm to find good solutions to selecting a set of areas, in this case properties, that meet the constraint of achieving representation targets at a minimum "cost". The objective function can include costs of areas, total boundary length of areas, and penalties for failing to meet conservation objectives. For each scenario, I used Marxan's best run (the run with the smallest objective function) and calculated the total area

selected and total expected costs. To ensure that I explored ranges of each variable while holding all other variables constant, I used a full factorial design resulting in 200 reservation scenarios (Table 2.2). Details on methods for each of the variables are below.

Biodiversity objectives

Based on conversations with DERM I developed two different methods for setting conservation objectives that reflected the department's thinking and minimum policy requirements for the Commonwealth Government to achieve 10% protection of all bioregions. I do not endorse these objectives as ecologically adequate, but present them here to illustrate their effect on costs of protected areas.

At the core of Australia's biodiversity conservation strategy are the goals for protected areas to be comprehensive, adequate, and representative. Comprehensiveness refers to the need to sample all bioregions; adequacy relates to the persistence of biodiversity; and representativeness indicates how well ecosystems are sampled within bioregions (ANZECC 1997). To address both comprehensiveness and representativeness, I used the Queensland Herbarium's regional ecosystem mapping of remnant vegetation at 1:50,000 to identify my conservation features (DERM 2009). Regional ecosystem mapping is the most comprehensive, fine-scale data on vegetation formations available for Queensland. Classification of regional ecosystems considers bioregional boundaries, vegetation structure, geology, landform, and soil (Neldner et al. 2005; Sattler & Williams 1999). There are thirteen bioregions (Figure 2.1) and over 1300 regional ecosystems mapped across Queensland. I used the estimated pre-clearing extent of each regional ecosystem for setting biodiversity objectives (for rationale, see Pressey et al. 2003), although my selections of new areas were based on remnant native vegetation.

The first method for defining objectives ("10%/1,000 ha") was derived from minimal policy requirements (National Objective and Targets for Biodiversity Conservation 2001-2005 2001; UNEP 2002). I used a base objective of 10% of the estimated pre-clearing extent of each regional ecosystem. If this percentage was less than 1,000 ha, I set the objective to 1,000 ha. If the pre-clearing extent was less than 1,000 ha, I set the objective to the pre-clearing extent (i.e.

100%). I then expressed these objectives as ha of remnant vegetation. If the remnant area of a regional ecosystem was smaller than its objective, I trimmed the objective to the remnant area.

The second method ("scaled objectives") used a power function to scale objectives based on extent of vegetation. This method reflects the fact that a very extensive ecosystem might not need 10% protection to ensure long-term viability, whereas a small, heavily cleared ecosystem could need a much larger percentage protected. If the pre-clearing extent was less than 1,000 ha, the objective was set to pre-clearing extent (i.e. 100%). If the pre-clearing extent was 1,000 ha or larger, I used the following equation with a power value of p=0.5:

$$t_k = \frac{x_k^p}{\sum_1^K x_k^p}$$

where t_k is the objective for regional ecosystem k, expressed as a proportion, p is the power, and x is the pre-clearing extent of regional ecosystem k. More extensive regional ecosystems (representing larger proportions of total pre-clearing vegetation) therefore had larger objectives, but the power function produced a diminishing rate of increase in objectives with increasing pre-clearing extent. I multiplied each objective by pre-clearing extent to express it in ha and, if necessary, trimmed it to total remnant area.

Objectives for the 10%/1,000 ha method totalled about 12.5 million ha compared to about 10.6 million ha for scaled objectives. However, the methods differed more importantly in their objectives for individual regional ecosystems. Compared to the 10%/1,000 ha objectives, scaled objectives gave larger values to regional ecosystems with smaller pre-clearing extents and smaller values to regional ecosystems with larger pre-clearing extents (Figure 2.2).

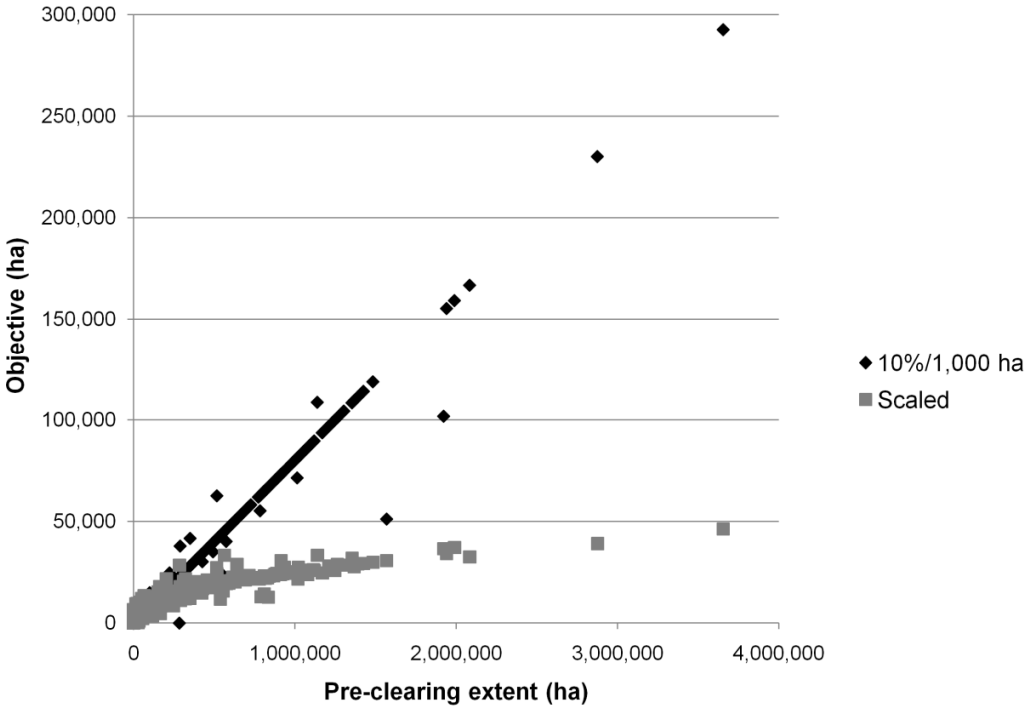


Figure 2.2 Objectives for regional ecosystems plotted against estimated pre-clearing extents for both methods.

Subdivision of properties

Willingness of landholders to subdivide properties for sale is likely to be highly variable and difficult to estimate *a priori* for large numbers of properties managed by people with no previous contact with DERM. Acquisition of whole properties is more expensive and will often involve purchase of land without native vegetation. Acquisition only of native vegetation is less expensive and more targeted. To bound my estimates of costs, I compared two subdivision scenarios, one without subdivision and one involving subdivision to protect only remnant native vegetation. In scenarios with subdivision, I assumed homogeneous cost across each property (below) and pro-rated the cost for remnant vegetation. In places, my subdivision assumption will underestimate costs, with protection of only unrealistically small patches of native vegetation. However, it reflects assumptions made in the literature (Carwardine et al. 2008; Klein et al. 2009a).

Legal acquisition routes

I explicitly considered the multiple acquisition routes for freehold (privately owned) land (Table 2.3). Freehold land can be purchased by the state at market value. Alternatively, the owner can retain the property with a voluntary Nature Refuge covenant on all or part of it, with no acquisition cost. The two acquisition routes for freehold land mean that estimates of establishment costs are uncertain, depending on assumptions about willingness of landholders to sell or negotiate Nature Refuges. Additionally, the Queensland Government's commitment to an eight-fold increase in participation in the Nature Refuge program might not reflect the actual willingness of landholders. Based on the relative cost of freehold, leasehold and state land, I assumed that the government would purchase at most 40% of freehold land at market price and the remainder would be negotiated as Nature Refuges (Table 2.2). This reflects the fact that freehold land can be very expensive and many freehold properties are unsuitable as national parks because they are small and heavily cleared with only fragments of native vegetation remaining. This reasoning reflects DERMs approach, so my range of acquisition assumptions (0-40%) likely reflects the scenarios considered by government.

Table 2.3 Pathways of land into the Queensland protected area system and associated costs in relation to tenure.

Pathway into protected area system	**Cost of acquiring**	**Transaction cost**	**Annual management costs**
Freehold voluntary purchase to create new park	Market value	$20,000 per sale for coastal properties, $15,000 elsewhere	$8.12 per ha
Leasehold voluntary purchase to create new park	Market value	$20,000 per sale for coastal properties, $15,000 elsewhere	As above
Leasehold Future Conservation Area (FCA): terminal 30-year lease with transfer to parks system at expiry	Value of improvements (difference between market and unimproved land value)	$20,000 per sale for coastal properties, $15,000 elsewhere	As above
State Forests or other State land transfer to parks system	None	None	As above
Freehold converted to Nature Refuge by covenant	None	$20,000 per sale for coastal properties, $15,000 elsewhere	$3.82 per ha
Leasehold converted to Nature Refuge by covenant	None	$20,000 per sale for coastal properties, $15,000 elsewhere	As above

I also considered the multiple acquisition routes for leasehold land (land owned by the State and leased for set periods for agricultural or grazing) (Table 2.3). Under the Delbessie Agreement (DERM 2007), landholders with leases currently up for renewal identified as conservation priorities have three options: immediate sale at market value; 30-year terminal lease with payments for improvements at the end of this term; and renewal of lease with a covenanted Nature Refuge on part of the property. If the landholder elects to sell immediately, the Queensland government is required to purchase the property at market value. While conversations with DERM indicate that 30% of leaseholders are expected to sell, with the remainder negotiating renewed leases with Nature Refuges, this assumption is untested. Little is known about leaseholders' responses to the Delbessie Agreement, so the establishment costs associated with leasehold purchases and renewals are highly uncertain. I therefore explored acquisition levels up to 90% at market price, with the remainder negotiated as Nature Refuges (Table 2.2).

For each level of assumed acquisition of freehold and leasehold properties, I calculated the expected cost of properties as: $x*costNR+y*costpurchase$, where x and y are the percentages assumed for Nature Refuge and acquisition, respectively, and *costNR* and *costpurchase* are the costs of Nature Refuge and acquisition, respectively. For both freehold and leasehold land, I avoided *a priori* allocation of properties to individual acquisition routes to avoid idiosyncratic correlations between cost and particular examples of regional ecosystems.

Planning units and conservation costs

Legal properties are the units with which managers implement conservation actions, but previous analyses of conservation costs have rarely used cadastral boundaries. To more accurately estimate the costs and extent of land needed to meet objectives, I used legal property boundaries to define planning units (NRW 2008a). My subdivision scenarios involved selection of properties but costing only of native vegetation.

For each property, I used data on tenure, unimproved land value, and sales prices (NRW 2008a, b, c) to estimate the cost of acquisition, considering all acquisition routes (Table 2.3). Dates of

land valuation and sales data varied, so I adjusted all values to 2008 dollars using published annual interest rates (RBA 2009). I estimated market values from recent sales of properties in Queensland from 2000-2008 with hedonic modelling (Newburn et al. 2006) (see Appendix 1 for details). I considered a standard ordinary least squares (OLS) model as well as a geographically weighted regression model (for comparison of results see Appendix 1 for details). I tested the OLS residuals for spatial autocorrelation using the Moran I Statistic, which rejected the null hypothesis that there was no spatial autocorrelation ($p<0.01$). I therefore modelled sales value with geographically weighted regression in ArcGIS 9.3, which analyses spatially variable relationships between the dependent and independent variables (for full details of variables considered see Appendix 1).

For my geographically weighted regression, I first considered the entire state. However, because coastal properties in Queensland have different characteristics to those elsewhere (for example, average size of coastal properties is 1/40 that of others), analysing the entire state led to local multi-collinearities in coastal properties. I therefore applied geographically weighted analysis for coastal properties and the remainder of properties separately. For coastal properties, the only predictor without strong local correlations was *log(cleared area, ha)*. The local R^2 for coastal properties was lower than for others because of the lack of predictors available to capture potential for coastal development (adjusted $R^2 = 0.688$). For the remainder of properties, predictors were *log(land value per ha)*, *log(cleared area, ha)*, *log(soil, ha)* and *log(distance to nearest town, km)* (adjusted $R^2=0.904$).

The spatially variable coefficients for *log(cleared area, ha)* and *log(land value per ha)* and the final predicted sales values are in Figure 2.3. The coefficient for *log(cleared area, ha)* is of interest because this was the only predictor used across the entire state (including coastal areas). The coefficient for *log(land value per ha)* is of interest because land value is typically the only type of cost data used to estimate acquisition costs in other academic studies for Australia (Carwardine et al. 2008; Klein et al. 2009a). The final sales values are easily interpreted for the state. Noticeable low-cost regions along the coast corresponded to defence properties. High-cost inland properties followed the major inland highway and clustered around agricultural and mining towns.

For each property in each scenario, I calculated the total cost (*TC*) as an expected value based on the respective percentage acquisition assumptions, using the following equation:

$$TC = x * costNR + y * costpurchase + x * managementNR + y * managementNP + transaction$$

where *x* and *y* were the percentages assumed for Nature Refuge and acquisition, respectively, *costNR* and *costpurchase* were the costs of Nature Refuge and acquisition, respectively, *managementNR* and *managementNP* were the one-year increase in management costs for Nature Refuge and national park, respectively, and *transaction* was the transaction cost (Table 2.3). Transaction costs were based on estimates by DERM. Management costs were based on gross hectares added, current average annual management costs of $8.12 per ha of national park (from the most recent available financial expenditures at the time of analysis, 2006-7), and average annual NatureAssist cost for Nature Refuges of $3.82 per ha (from the most recent available financial expenditures at the time of analysis, 2007). For properties in the expanded protected area system, I considered management costs for a single year with no discounting.

Reserve design

I accounted for the contribution to objectives of existing protected areas by locking them into the solutions in all scenarios. I used Marxan to design additional reserves that met the remaining portions of all objectives while minimizing total costs (Ball et al. 2009). For each of my 200 reservation scenarios, I ran Marxan with 100 repeat runs and no configuration constraints (boundary length modifier or BLM set to 0). Spatial design criteria should reflect differences in regional ecosystems, population density, and management objectives, so applying universal criteria across the entire state of Queensland would have been inappropriate.

Analysis of reserve design solutions

I compared the scenarios by recording for each, from the 'best' solution (with the smallest objective function across 100 repeat runs), the total extent of selected areas, excluding existing reserves, and their total cost.

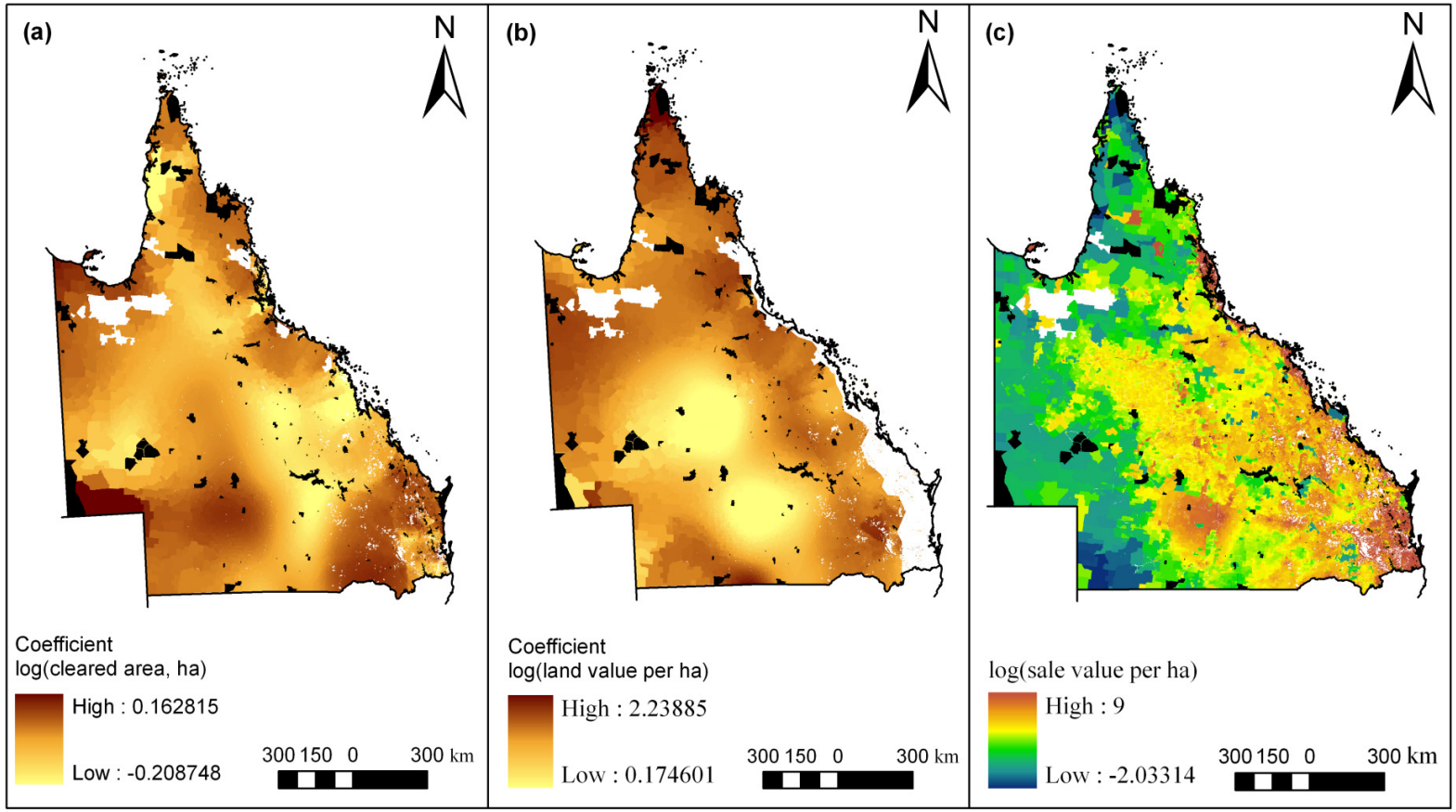


Figure 2.3 Maps of spatially variable coefficients based on geographically weighted regression conducted for coastal and non-coastal regions of Queensland. Current protected areas are shown in black for reference. White areas in the north-west have no vegetation mapping. (a) Coefficient of log(cleared area, ha) from the geographically weighted regression for coastal and non-coastal regions. White areas in the south-east are properties without remnant vegetation and are excluded from my analysis. (b) Coefficient of log(land value per ha) from the geographically weighted regression for non-coastal regions. The additional, continuous white area along the eastern seaboard is coastal Queensland, excluded from this model. North of this excluded region is Cape York Peninsula, considered separately in the model. (c) Predicted log(sale value per ha) for properties with remnant vegetation. White areas in the south-east are properties without remnant vegetation and are excluded from my analysis.

I used a two-step process to assess the sensitivity of cost estimates to values of my variables. First, I calculated the total cost that would apply if all assumed values of key variables held true ("expected cost"). Expected costs would apply, for example, if areas were selected on the assumption that no subdivision of properties would be possible and, after negotiation with landholders, this proved to be the case. I recorded the expected cost for each run and regressed expected cost against my four variables: percentage acquisition route for freehold land (percentage, 0-40), percentage acquisition route for leasehold land (percentage, 0-90), type of objective (binary, 10%/1,000 ha or scaled), and subdivision (binary, yes or no). Second, I calculated the cost of each scenario with different amounts of deviation from scenario assumptions ("unexpected costs"). For example, areas might be selected and total costs estimated on the assumption that 50% of leasehold properties would be purchased. If the percentage requiring purchase was actually 60%, then 10% of properties expected to be inexpensive would prove to be otherwise and unexpected costs would apply. The increase in total cost would be greater than if 60% acquisition had been expected initially, which would have shifted selections more towards properties with lower acquisition costs. I calculated both the absolute and percentage differences between expected and unexpected costs.

This approach to estimating sensitivity to key variables is important because *a priori* assumptions about values of variables, no matter how well informed, will always be inaccurate to some extent. My approach "commits" the agency to configurations of new reserves based on assumptions holding, even if the assumptions prove incorrect. In reality, the agency would revise configurations accordingly, although it might not be able to revise its financial estimates so easily. Nonetheless, my approach indicates how inaccurate cost estimates can be if values of key factors deviate from expected.

Results

For each of my 200 scenarios I used Marxan (Ball et al. 2009) to select properties to meet my conservation objectives and calculated the total area selected and total expected costs. I then calculated the potential unexpected costs for the areas selected in each scenario given certain deviations from assumptions. The total additional area of land required to achieve objectives was much larger than the government's promised 12 million ha. The minimum area added to the reserve system across all scenarios was 18 million ha and the maximum was 29 million ha with an average of 23 million ha. In all scenarios, the full extent of available State land (~2.3

million ha) was selected by Marxan due to its low cost compared to other tenures. Total costs ranged from $214 million to $2.9 billion, with larger costs associated with larger percentage acquisition assumptions.

I identified the scenarios most likely to reflect the government's commitment to acquire 4 million additional ha of national park, which I also interpreted as 33% of the total expanded area required to meet objectives. Given an average of 23 million ha of additional protection needed, adding only 4 million ha to the national park estate would mean that covenants on private land would be relied upon heavily to meet conservation objectives. On the assumption that the government will commit to acquire 33% of the required land to meet conservation objectives, the area needing acquisition actually ranged from about 6 to 10 million ha. The total expected costs of scenarios involving an additional 4 million ha or 33% acquisition for national parks ranged from $250 million to $1.6 billion. The minimum cost estimate of $250 million was based on the 10%/1000 ha objectives and assumed property subdivision, 30% freehold acquisition, and 0% leasehold acquisition. However, given the uncertainties about implementation of the Delbessie Agreement and the potential need to purchase larger amounts of leasehold land, the maximum expected cost could rise to $2.3 billion (based on the scaled objectives with no subdivision, assuming 0% freehold acquisition and 90% leasehold acquisition).

Total expected costs were 50-80% larger for scaled objectives which required ~17% more area (Figure 2.4). For both sets of objectives, the expected costs of subdivision scenarios were 5-30% lower than those without subdivision (Figure 2.4). Biodiversity objectives interacted with percentage acquisition assumptions to influence expected costs. Expected costs responded linearly to increasing percentages of both freehold and leasehold land purchased for both sets of objectives (Figure 2.4). In the multiple regression of total cost against all factors, objectives and subdivision had the largest effects (Table 2.4). On average, scaled objectives were about $489 million more expensive than 10%/1,000 ha objectives. Subdivision of properties for purchase of only remnant vegetation reduced total costs on average by $225 million. Percentage purchase of leasehold land had almost three times the effect on total cost as percentage purchase of freehold land.

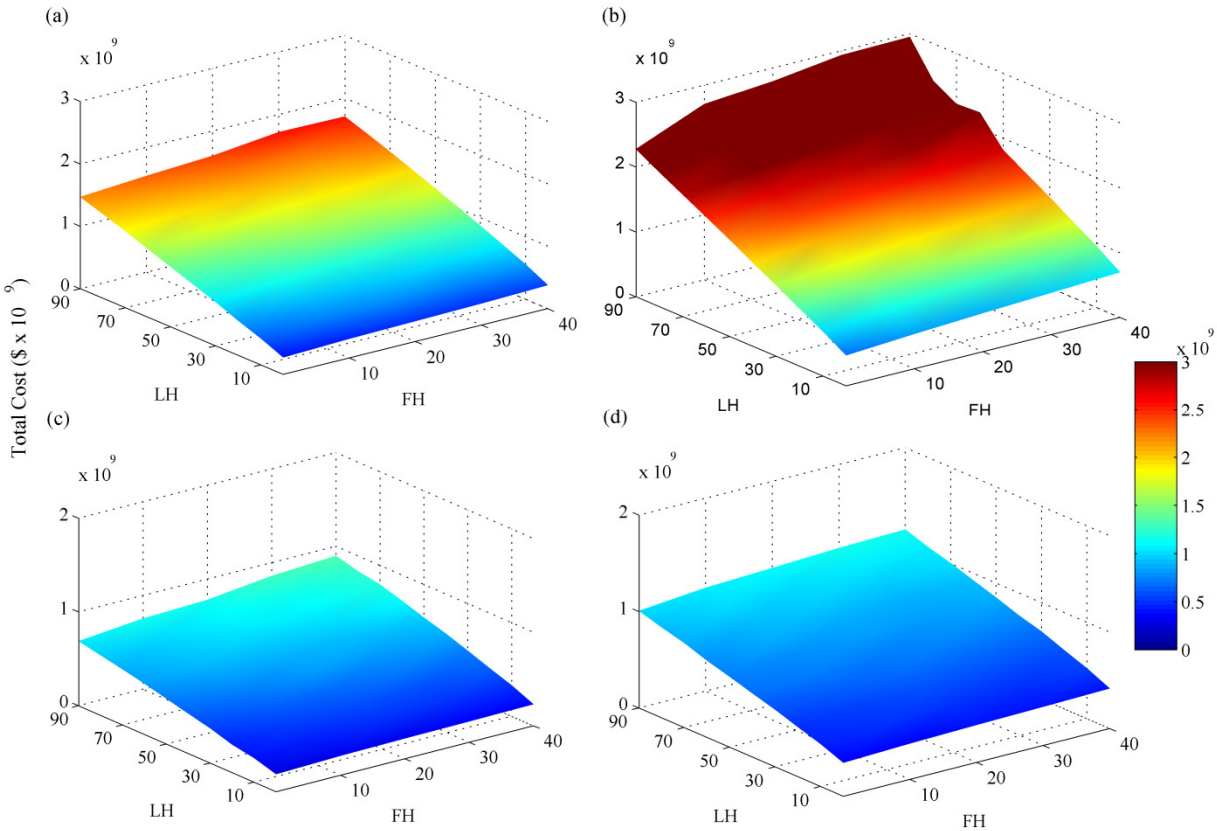


Figure 2.4 Total cost (billions of Australian dollars) as a function of variable percentages of leasehold and freehold land purchased. Expected total costs are plotted on the z-axes, percentages of leasehold (LH) purchased are on the y-axes, and percentages of freehold (FH) purchased are on the x-axes. (a) 10%/1,000 ha objectives and no subdivision of properties; (b) scaled objectives and no subdivision of properties; (c) 10%/1,000 ha objectives and subdivision of properties; (d) scaled objectives and subdivision of properties.

Table 2.4 Multiple regression model of total cost. All variables are highly significant ($p < 0.001$). Coefficients represent the dollar change in total cost.

		Independent Variables (coefficient, *t*)				
	Intercept	pct_{LH} [a]	pct_{FH} [b]	Subdivision	Scaled objectives	Overall R^2
	242 124 199,	1 538 538 297,	455 236 918,	-225 607 405,	489 342 331,	
Total cost	104.13	518.9	75.60	-132.46	287.3	0.949

[a] pct_{LH} Percent leasehold assumption was expressed in proportional form (i.e. 10% coded as 0.10). Therefore, the coefficient indicates that a 10% increase in leasehold purchase gives a dollar change in cost of 1 538 538 297 x 0.1 or 153 835 829.

[b] pct_{FH} Percent freehold assumption was expressed in proportional form (i.e. 10% coded as 0.10). Therefore, the coefficient indicates that a 10% increase in freehold purchase gives a dollar change in total cost of 455 236 918 x 0.1 or 45 523 691.

Predictably, unexpected ability to subdivide properties reduced costs and unexpected inability to subdivide properties increased costs, but the responses were non-linear across scenarios (Figure 2.5). The largest reductions in costs when assumed inability to subdivide proved incorrect were for 40% freehold purchase and 0% leasehold purchase for the 10%/1,000 ha objectives, and 40% freehold purchase and 90% leasehold purchase for the scaled objectives (Figure 2.5a,b). Similar combinations of purchases produced the largest increases in costs when assumed subdivision was not possible. The largest increases occurred for 40% freehold purchase and 20% leasehold purchase for the 10%/1,000 ha objectives, and 40% freehold purchase and 80% leasehold purchase for the scaled objectives (Figure 2.5c,d).

The sensitivity of total costs to unexpected 10% increases in purchases of leasehold land increased with larger expected percentage purchases of freehold land and decreased with larger expected percentage purchases of leasehold land (Table 2.5). However, the sensitivity was larger for the scaled objectives, with a maximum increase of $279 million compared to $217 million for the 10%/1,000 ha objectives. The opposite trend applied to unexpected 10% increases in purchase of freehold land. Sensitivity decreased with larger expected percentage purchases of freehold land and increased with larger expected percentage purchases of leasehold land (Table 2.6). Average increases in costs were higher for the 10%/1,000 ha objectives for 0% freehold purchase but, for all other freehold percentage purchase scenarios, average increases in cost were higher for the scaled objectives.

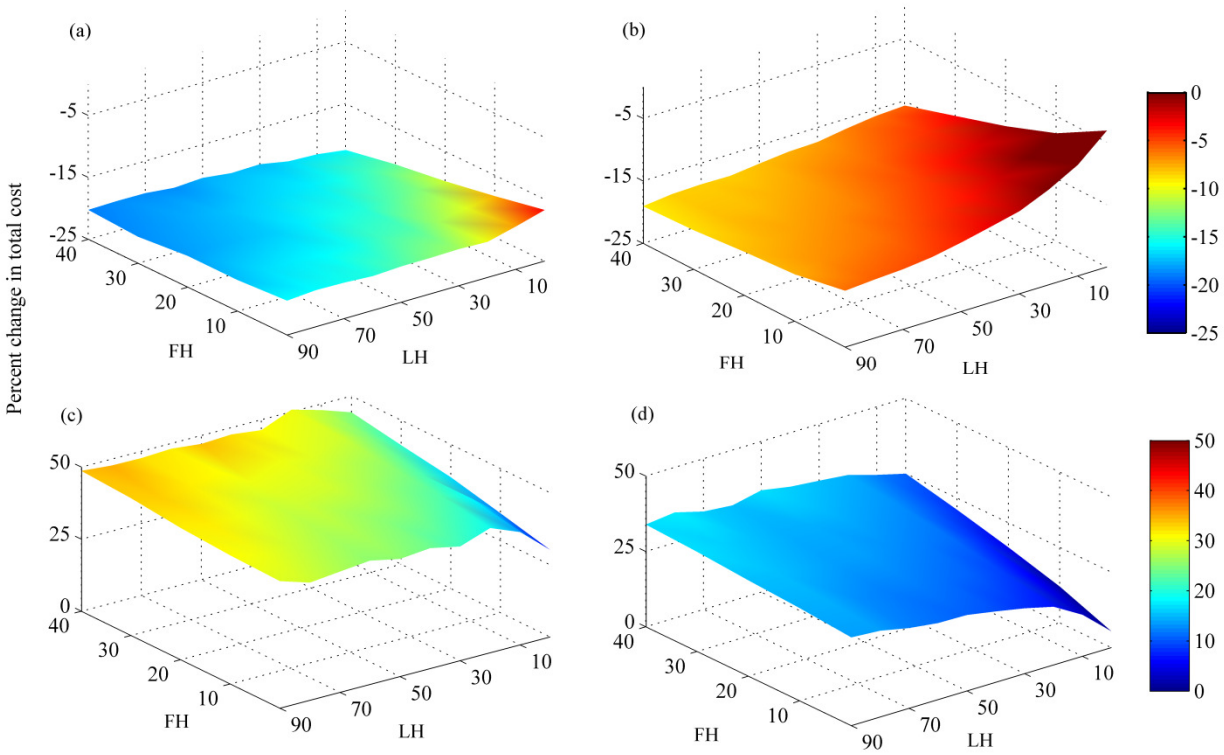


Figure 2.5 Percentage change in total cost due to unexpected subdivision conditions as a function of variable percentages of leasehold and freehold land purchased. Percentage deviations from expected total costs are plotted on the z-axes, percentage purchases of leasehold (LH) are on the y-axes, and percentage purchases of freehold (FH) are on the x-axes. (a) Percentage reduction in cost if all properties can be unexpectedly subdivided under the 10%/1,000 ha objectives and no subdivision assumption; (b) Percentage reduction in cost if all properties can be unexpectedly subdivided under the scaled objectives and no subdivision assumption; (c) Percentage increase in cost if all properties are unexpectedly impossible to subdivide under the 10%/1,000 ha objectives and subdivision assumption; (d) Percentage increase in cost if all properties are unexpectedly impossible to subdivide under the scaled objectives and subdivision assumption.

Table 2.5 Sensitivity to 10% change in expected purchase of leasehold land for both objectives (10%/1,000 ha on left and Scaled on right), holding all other assumptions constant. Sensitivity is expressed as the increase (AUD$ million) in total cost for an unexpected 10% increase in purchase of leasehold. The expected percentages of leasehold (LH) and freehold (FH) purchase that serve as baselines for the increases are given as rows and columns, respectively. For example, dollar values in the row corresponding to 20% leasehold correspond to an unexpected need to purchase 30% of leasehold land. No changes in expected purchases of freehold land apply here. (a) assuming no subdivision of properties; (b) assuming subdivision and purchase only of remnant vegetation.

(a)

	10%/1,000 ha objectives						**Scaled objectives**				
LH, FH	0	10	20	30	40	LH, FH	0	10	20	30	40
0	186	210	216	218	217	0	249	266	276	279	279
10	152	163	172	175	177	10	212	223	228	234	233
20	141	147	155	159	162	20	202	209	213	219	219
30	138	142	146	152	155	30	201	206	210	212	217
40	135	138	142	147	150	40	198	202	206	209	210
50	134	137	140	143	145	50	199	202	205	208	210
60	133	135	138	140	144	60	197	199	202	205	241
70	133	135	136	139	142	70	196	199	201	203	206
80	132	133	135	135	138	80	195	198	200	202	204
90	130	130	133	143	136	90	196	231	233	242	237

(b)

	10%/1,000 ha objectives						**Scaled objectives**				
LH, FH	0	10	20	30	40	LH, FH	0	10	20	30	40
0	133	153	156	159	160	0	179	193	197	204	204
10	116	124	142	133	137	10	160	167	184	175	178
20	109	113	119	122	126	20	158	163	165	170	172
30	108	110	114	119	121	30	155	160	162	165	168
40	105	107	110	113	116	40	155	159	160	163	165
50	105	107	109	123	113	50	155	157	160	174	167
60	103	104	106	107	110	60	153	155	157	158	159
70	103	103	105	107	108	70	152	154	155	157	158
80	101	102	103	105	106	80	153	154	156	157	185
90	102	102	103	105	106	90	152	153	155	156	157

Table 2.6 Sensitivity to 10% change in expected purchase of freehold land for both objectives (10%/1,000 ha on left and Scaled on right), holding all other assumptions constant. Sensitivity is expressed as the increase (AUD$ million) in total cost for a 10% increase in purchase of freehold. The expected percentages of leasehold (LH) and freehold (FH) purchase that serve as baselines for the increases are given as rows and columns, respectively. So, for example, dollar values in the column corresponding to 20% freehold correspond to an unexpected need to purchase 30% of freehold land. No changes in expected purchases of leasehold land apply here. (a) assuming no subdivision of properties; (b) assuming subdivision and purchase only of remnant vegetation.

(a)

	10%/1,000 ha objectives						**Scaled objectives**				
LH, FH	0	10	20	30	40	LH, FH	0	10	20	30	40
0	137	23	19	19	21	0	117	58	54	51	51
10	142	38	25	22	23	10	139	67	59	56	56
20	157	44	30	25	26	20	141	72	66	58	58
30	156	51	39	30	27	30	147	79	68	65	59
40	158	53	42	33	30	40	148	84	71	66	64
50	154	57	45	38	35	50	149	89	75	68	63
60	156	60	46	41	36	60	150	89	77	70	66
70	157	57	50	42	37	70	152	88	81	74	67
80	161	63	48	47	42	80	150	89	82	76	71
90	161	64	51	39	44	90	146	92	82	71	70

(b)

	10%/1,000 ha objectives						**Scaled objectives**				
LH, FH	0	10	20	30	40	LH, FH	0	10	20	30	40
0	88	14	12	11	11	0	79	34	30	28	27
10	106	24	18	12	12	10	91	41	39	32	30
20	103	32	19	16	14	20	96	45	39	33	32
30	113	36	26	17	16	30	96	46	42	37	33
40	116	37	27	23	19	40	92	49	44	39	36
50	111	38	31	27	22	50	98	53	43	43	37
60	115	39	31	28	25	60	106	54	46	42	40
70	114	43	34	28	27	70	99	55	48	42	41
80	107	42	35	29	30	80	101	58	48	45	39
90	107	43	38	32	29	90	106	60	50	46	43

Discussion

While governments commonly identify lofty multi-year conservation goals, it is uncommon for them to estimate the required costs. When financial estimates are released, they might be serious underestimates, reflecting the pervasive problem of under-funding conservation. The Queensland government estimated the cost of its commitment of 20 million ha of protected areas by 2020 at $120 million (Anna Bligh Premier of Queensland 2008). Recognizing that my cost estimates include increases in funding for management while the government estimate was strictly for acquisition, the figure of $120 million is still well below my expected cost range (Figure 2.4). My lowest total estimate was $214 million and required several extremely optimistic assumptions, including that all land would be acquired through Nature Refuges, and that remnant vegetation would always be subdivided out of properties to more efficiently achieve objectives. In the range of scenarios that best matched the political promise by the Queensland government to acquire 4 million ha or 33% of the expanded area, the minimum cost estimate was $250 million.

Cost variability

Total costs across my 200 scenarios varied by an order of magnitude from $214 million to $2.9 billion. All of the factors tested influenced this variation, with objectives and subdivision the most important. Cost was linearly influenced by percentage leasehold acquisition, which is to be expected because of the linear increments in percentage of properties acquired. However, when I considered unexpected costs, they responded non-linearly to unexpected inability to subdivide and percentage purchase of freehold. This is because freehold properties are small, much more expensive per ha than leasehold properties and have smaller proportions of native vegetation than leasehold properties. The wide range of cost estimates, the influence of underlying assumptions, interactions between assumptions, and non-linear responses make *a priori* estimates and general rules of thumb difficult to derive.

Many of the effects of key factors are difficult to anticipate without the kinds of analyses presented here. For example, scaled objectives raised costs by 50-80% despite the targeted area being about 2 million ha smaller. There were two reasons. First, rarer regional ecosystems had larger proportional objectives with the scaled method and these ecosystems were more expensive to protect per unit area. The Pearson correlation between log(regional

ecosystem area) and log(regional ecosystem average cost per ha) was -0.117 ($p<0.001$). The second reason was that larger scaled objectives reduced spatial flexibility for representing rarer regional ecosystems, providing few or no alternatives to more expensive properties. The costs of conservation can therefore be more sensitive to objectives for individual biodiversity features than to aggregate area goals such as percentages of a state.

Dealing with uncertainty about costs

The amplitude of unexpected increases and reductions in costs could help to guide what assumptions are the most conservative. I define financially conservative assumptions here as those that safeguard against large cost increases while allowing for potential financial windfalls. I measured uncertainty by estimating how expected costs were changed when assumptions did not hold true. The largest deviations from expected costs occurred when very low acquisition assumptions (~0-20%) proved inaccurate. When acquisition assumptions were low, property values were effectively smoothed to obscure spatial variability in purchase costs. This can result in selection of expensive properties and expose planners to large unexpected costs if landholders are less receptive to Nature Refuges than assumed. Selections of properties under larger acquisition assumptions recognized variability in costs and avoided expensive properties, where there were choices, so unexpected costs represented smaller increases when assumptions did not hold (Table 2.6). Therefore, a conservative approach would use larger acquisition assumptions to allow the selection algorithm to avoid expensive properties where possible.

Unexpected costs or savings resulting from inaccurate subdivision assumptions indicated that potential costs far outweighed potential savings. When subdivision unexpectedly occurred after selection of areas, costs were reduced by only ~15%. In contrast, when subdivision unexpectedly did not occur, costs increased by ~45%. A conservative approach – assuming no subdivision - therefore resulted in only a small loss of efficiency but avoided a large financial risk. Unexpected increases or reductions in costs related to subdivision also interacted with assumptions about percentages of freehold and leasehold land that would be acquired (Figure 2.5), so potential increases were up to 50% in my analyses.

Lessons for Queensland

Expanding the Queensland protected area system could incur a wide range of financial costs, depending on biodiversity objectives and socio-political conditions encountered. While this

variation might be reduced by estimating plausible bounds for key factors, a single exact estimate cannot be provided because there will always be uncertainty associated with *a priori* assumptions. The narrowed range of scenarios based on the acquisition promise of 4 million ha or 33% still resulted in costs varying by an order of magnitude due to unavoidable uncertainty around the responses of leaseholders to the Delbessie Agreement. Because the Queensland government is required under this Agreement to immediately purchase leasehold properties with conservation value, I explored the full range of leasehold percentage purchases, resulting in a wide cost range. Even long experience with acquisition of individual properties, sometimes involving protracted negotiations, does not necessarily equip an agency to accurately cost a massive expansion of protected areas across many hundreds of properties, involving many hundreds of landholders who have not previously dealt with agency officers, while implementing new, and largely untested, legislation.

To avoid undesirable surprises in Queensland, planners should not assume that large percentages of land will be protected through Nature Refuges at low cost to the government. Similarly, they should use conservative assumptions about subdivision. Assumptions of larger percentages purchased and smaller proportions of properties subdivided will cause the selection algorithm (in my case, Marxan) to avoid expensive properties with extensive clearing of native vegetation. While increasing expected budgets, this would avoid large unexpected increases in costs. The selected properties could then be analyzed in detail with respect to remnant vegetation and attitudes of landholders to narrow the range of expected costs, targeting specific properties for subdivision and acquisition routes.

Narrowing the range of potential costs of an expanded protected area system in Queensland requires a better understanding of landholders' interests in selling their properties in whole or part, negotiating leases, or participating in the Nature Refuge program, and how these interests vary geographically and by land use. Likely levels of participation in conservation programs as well as potential costs can be determined through local-scale experiments such as tendering processes or closed-bid auctions (Hajkowicz 2007; Stoneham et al. 2003). This in-depth analysis would, however, be very difficult and costly across Queensland. Reducing variation in financial estimates might also be helped by decision rules such as only considering properties with at least 50% remnant native vegetation (Messer 2006).

Particular care should also be given to setting interim versus long-term objectives, especially for the small regional ecosystems that can influence costs and for any regional ecosystems threatened by further reductions in extent. Achieving longer-term, scientifically defensible targets, which might have high financial costs, in the context of continuing, incremental depletion of native vegetation, requires an explicit strategy for scheduling conservation actions (Pressey et al. 2004), not evident in current Queensland policy.

General conclusions

Although my analysis focused on Queensland, the same factors are likely to affect the cost of conservation in other parts of the world. My findings are generic, in that the estimated costs of expanding protected area systems to meet policy goals or political commitments are likely to vary widely and to be highly sensitive to assumptions about influential factors.

In my study, subdivision and conservation objectives were particularly important in influencing the costs of conservation for very different reasons. The ability to subdivide properties dramatically affected the amount of land required to meet the conservation objectives. This result is general and not context-specific and has been noted previously in relation to the total extent of selected areas (Pressey & Logan 1998; Rodrigues & Gaston 2001). Conservation planners should be aware of the sensitivity in their spatial selections and subsequent financial estimates when assuming that landholders will be willing to subdivide properties. Conservation objectives affected the amount of land acquired, the total cost of achieving conservation objectives, and the spatial flexibility in achieving those objectives. The spatial options available to meet objectives are context-specific, but other studies have found effects of conservation objectives on total area and costs (Justus et al. 2008; Warman et al. 2004).

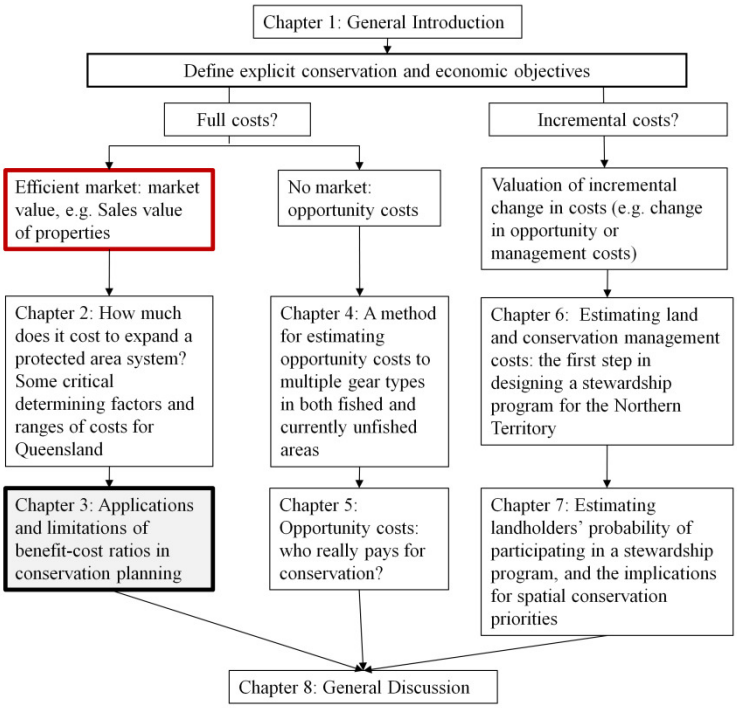
Comparisons of the potential political pathways for acquiring and protecting land have been largely neglected in the literature. However, with the recent increase in conservation programs on private lands (Claassen et al. 2008; Rissman et al. 2007; Stoneham et al. 2003), the cost differences between the traditional acquisition route and alternative approaches to protection are likely to become more apparent. This means that, when budgeting for conservation, assumptions regarding how lands are protected will become increasingly important. The factors I selected provide important insights into the types of impacts that assumptions can have on estimated financial costs. However, conservation planners should

evaluate the context-specific assumptions they make based on the potential of those assumptions to change conservation requirements. This means that factors not discussed here might be important in other studies.

While conservation planners cannot be expected to have infallible foresight into the accuracies of their assumptions, they can be expected to be transparent about their uncertainties (Bottrill et al. 2008). Given the global under-funding of conservation (James et al. 2001) and recent commitments to extensive, but uncosted, expansions of protected area systems, planners must be forthright in providing realistic estimates and avoiding underestimates that could compromise both their credibility and conservation outcomes.

Chapter 3 Applications and limitations of benefit-cost ratios in conservation planning [1]

Benefit-cost ratios (BCRs) have been applied to conservation decisions to promote cost-efficiency and transparency. However, published studies have not acknowledged how uncertainties associated with use of surrogates or variable estimates of benefits and costs can affect decisions. To address these issues, I first review several unresolved issues associated with BCR in conservation prioritizations. I then use the promised expansion of the Queensland protected area estate to quantitatively demonstrate how different surrogate measures of benefits and costs, such as areas of vegetation types for benefit or land values for acquisition cost, change budget allocations. Second, I analyse the effects of uncertainties associated with BCR by applying an information-gap (info-gap) model to explore the robustness of my BCR allocations of priority areas. I define robustness as the proportion of properties selected in error due to uncertainties about benefits and costs. My model shows how uncertain I can be while still selecting a portfolio of conservation areas that performs above a critical threshold of misallocations. I discuss the implications of my info-gap model for managers implementing BCR prioritizations on the ground.



[1] Adams V.M., Pressey R.L. (In Review) Applications and limitations of benefit-cost rations in conservation planning. *Conservation Letters*.

Introduction

Conservation budgets are limited so, to maximize conservation outcomes, prioritization strategies have simultaneously considered costs and benefits of conservation actions (Ando et al. 1998; Ferraro 2003a). To promote transparency and cost-efficiency, benefit-cost ratios (BCRs) have been recommended to managers (Bottrill et al. 2008; Fuller et al. 2010; Joseph et al. 2009b). BCRs are fundamentally based on principles of total economic valuation in which the goal is to maximize surplus, given by benefits minus costs, with total valuation requiring complete accounting of the net present value of benefits and costs. In conservation applications, in which costs and benefits are typically expressed in non-monetary metrics, ranking of alternatives by BCR is equivalent to maximizing the benefit minus cost.

However, in conservation applications, benefits and costs have been defined only partially which can result in mis-allocations if the partial measures do not accurately capture total benefits and costs (Adams et al. 2010). Several studies have examined how different formulations of benefit measures can affect the portfolios selected and have identified ways of ensuring that a robust portfolio is selected when biophysical attributes are uncertain or difficult to measure (Ferraro 2003b, 2004). Benefits have been defined, for example, in relation to the contribution to environmental benefit objectives such as water quality (Ferraro 2003a), the probability of persistence of a species (Joseph et al. 2009b), or the extent of vegetation types in land parcels (Fuller et al. 2010) (see Table 3.1 for a summary of BCR applications). While these definitions are explicit, they might reflect only the direct benefits to an organization or individual rather than considering the additional objectives of diverse stakeholders, relative perhaps to ecosystem services, recreation and livelihoods. In addition, in the case of selecting land parcels to meet representation objectives, benefit formulations can be particularly troublesome. If outcomes are measured in relation to achievement of objectives but benefits do not incorporate these objectives (Fuller et al. 2010), then the BCR ranking procedure cannot optimally select land parcels based on the requirements of meeting the objectives.

There is a parallel limitation of using surrogates for costs in BCR applications for conservation planning, although the implications of cost surrogates on planning outcomes has not been explicitly examined in the literature to date. Typically, conservation applications of BCR have considered only specific costs, such as managing species (Joseph et al. 2009b) or acquiring land (Fuller et al. 2010), and not accounted fully for the direct costs to an

organization or broader social costs. Partial accounting of benefits and costs has four limitations: 1. the broader socio-economic context is neglected; 2. the ratio loses its pure value-for-money meaning; 3. trade-offs in selecting similarly ranked projects are not explicit; and 4. the effects of uncertainties in data have not been considered.

Table 3.1 Summary of applications of benefit-cost ratios.

Application	Purpose	Example	References
Theoretical exploration of conservation decision making	Examination of which decision making criteria result in optimal conservation strategies	Comparison of areas or projects selected for conservation investment using prioritizations based on benefits only and benefit-cost ratios	(Babcock et al. 1997; Ferraro 2003a; Juutinen & Monkkonen 2007; Newburn et al. 2005)
Benefit-cost analysis	Analysis of benefits and costs associated with different conservation actions or land uses	Justification of conservation action (such as a biodiversity corridor or sustainable use area) in preference to other uses of the same areas	(Burger et al. 2006; Gunawardena & Rowan 2005; Mmopelwa 2006; Naidoo & Adamowicz 2005; Rasul & Thapa 2006; Sinden 2004)
Ranking of actions	Selection of conservation actions based on ranking of BCRs to achieve "biggest bang for the buck"	Ranking of easement contracts, ranking of species management projects, defining priorities for generic conservation actions	(Azzaino et al. 2002; Babcock et al. 1997; Balmford et al. 2003; Bottrill et al. 2008; Ferraro 2003a, b, 2004; Fuller et al. 2010; Joseph et al. 2009b; Naidoo & Iwamura 2007; Williams et al. 2003)

Neglecting the broader socio-economic context

Studies based on partial estimates of the five components of costs described by Naidoo et al. (2006) – acquisition, opportunity, management, transaction and damage costs - are unlikely to be informed by true costs. Table 3.2 is a full listing of the types of costs associated with establishing a national park. Most studies have considered only acquisition costs because these are seen as the largest outlays by organisations (Fuller et al. 2010; Klein et al. 2009a). This approach fails, however, to acknowledge societal preferences, costs to stakeholders, and long-term costs of managing reserves. These other costs might outweigh acquisition costs and, if considered, alter the spatial pattern of conservation priorities. Additionally, partial accounting neglects potential cost savings through tourism and entry fees. At least some of the reserves located to maximize these savings might be cost-neutral in the long term (Thur 2010).

Table 3.2 Summary of "positive" and "negative" costs of establishing a national park. Positive costs are accrued by the agency establishing the park. Negative costs are income accrued by the agency and therefore represent cost savings.

Costs	Description/comment	Data used for estimation	References
Positive			
Acquisition	Market value of property	Sales data and infrastructure valuation	Sales data are rarely used. Land value is more commonly used in conservation planning (Carwardine et al. 2008; Fuller et al. 2010; Klein et al. 2009a). Records of land values are maintained relatively consistently for tax purposes. Sales values are only recorded when properties are sold. However, land value underestimates acquisition cost by different amounts within and between regions (Adams et al. 2011).
Transaction	Time costs and legal fees	Historical transaction costs	Transaction costs are typically included as a fixed amount, set arbitrarily (Carwardine et al. 2008; Fuller et al. 2010; Klein et al. 2009a). However, transaction costs are likely to vary by factors such as tenure, land use, and preferences of landholders.
Management	Establishment (one-time capital outlays such as visitor infrastructure)	Infrastructure valuation	In addition to acquisition of properties, properties being acquired for national parks require infrastructure to support both park management and visitor use. These large initial costs are likely to vary widely between properties, depending on the location, condition, and intended use of the new parks. Establishment costs are typically neglected by planning studies (Carwardine et al. 2008; Fuller et al. 2010; Klein et al. 2009a).
	Maintenance (recurrent annual expenditures)	Historical management expenditures estimated from budgets, or predicted expenditures based on characteristics of reserves	Management costs are variable between reserves depending on configuration, internal infrastructure, and landscape context. Ideally, management costs would be modeled to relate costs to these variables (Frazee et al. 2003; Wilkie et al. 2001). In practice, most studies that consider management costs use average per ha costs or previous models, many of which will not be appropriate for the region being studied.
Damage	Costs associated with damages to adjacent economic activities, arising from conservation programs.	Difficult to expect or estimate a priori, unless there are analogous examples in the same region	For example, loss of income can be estimated from damaged crops and livestock from wild animals living in protected areas adjacent to human settlements (Woodroffe et al. 2005).

Costs	Description/comment	Data used for estimation	References
Opportunity	Costs to stakeholders in terms of forgone revenues from alternate uses of resources. These costs might be reflected in costs to conservation organizations through acquisition costs or payments to stakeholders to compensate for forgone revenues.	Can be spatially mapped with standard valuation techniques	Standardized techniques have been developed for estimating opportunity costs (Carwardine et al. 2008; Naidoo & Adamowicz 2006; Naidoo et al. 2008). However, relatively little work has been done on incorporating opportunity costs to different stakeholder groups to ensure that conservation decisions distribute costs equitably (Adams et al. 2010).
Negative			
Revenues	Tourism revenue and visitor fees can be important sources of funds for managing national parks.	Tourism revenue and visitor statistics	(Becker 2009; Carr & Mendelsohn 2003)
Willingness to pay through taxes	Willingness of people to pay taxes or levies to fund maintenance of national parks. Willingness might be higher for parks that have obvious societal value, such as those close to urban centers or with iconic national status.	Willingness to pay surveys	(Jacobsen & Thorsen 2010; Naidoo & Ricketts 2006)

Losing the meaning of value for money

In the formal use of BCRs, the ratio is based on complete accounting and has an explicit monetary meaning concerning value for money: any project with a BCR greater than or equal to one would be accepted. However, when using non-monetary units, BCRs can only be used to judge the relative value of projects by ranking them. For example, Fuller et al. (2010) used presence of vegetation types as a surrogate for benefits and land value as a surrogate for market value and ranked land parcels to prioritize acquisitions for expanding the national reserve system (Fuller et al. 2010). In these cases, the critical threshold value, beyond which funding is not allocated, is budget-driven. Another aspect of losing the meaning of value for money is that, if many projects have similar BCRs, then ranking them based on partial benefits and costs might obscure those that actually provide the best value. For example, Ferraro (2003b) demonstrated that different benefit formulations result in different 'optimal' portfolios. In these situations, the set of parcels included in all portfolios might be considered high priorities for investment because they are robust to different benefit measures (in the case of Ferraro 2003b different scoring methods).

Considering trade-offs: are ten cheap projects the same as one expensive project?

The decision space for selecting projects based on relative benefits and costs can be divided into four quadrants (Figure 3.1). Regardless of whether BCRs are based on total net present value (Naidoo & Adamowicz 2005) or surrogates (Fuller et al. 2010), the relative ratios of projects in the quadrants remain the same. If the budget is not fully spent on projects in quadrant I (high benefit, low cost), then attention would shift to projects in quadrants II and III. However, the use of surrogates, and the loss of the formal meaning of BCRs of 1 or more, complicates decisions in these parts of the decision space. Based on surrogate measures, does a project in quadrant II, say with a ratio of 10/10, have the same priority as ten smaller projects in quadrant III, each with ratios of 1/1? A more nuanced assessment than BCRs is needed to answer this question, perhaps to understand the broader socio-economic context of projects (e.g. the superefficiency criterion proposed by Andersen & Petersen 1993 using a data envelope analysis). For example, important considerations that distinguish projects with similar ratios might include whether the projects could attract external funding, such as tourism to national parks, or the complementarity of projects in contributing to regional objectives.

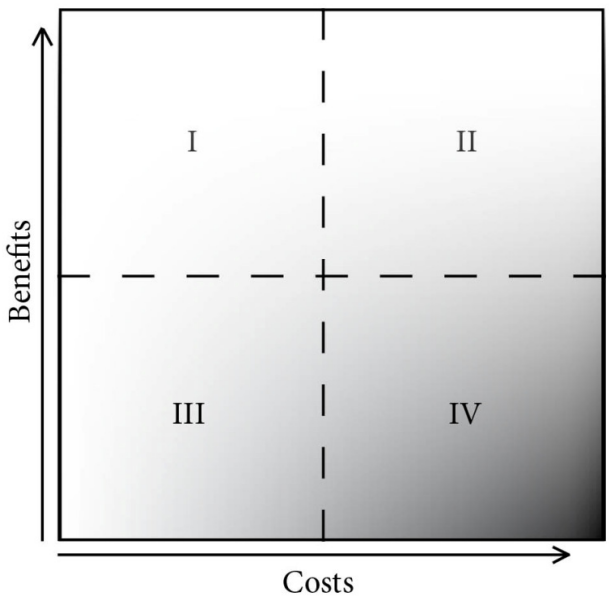


Figure 3.1 Benefit-cost decision space. When benefits are high and costs are low, the B/C ratio is high (quadrant I) and projects with BCRs in this area would be funded. When benefits are low and costs are high, the B/C ratio is low (quadrant IV) and these projects would not be funded. When costs are high but benefits are high or when benefits are low and costs are low (quadrants II and III), conservation planners might choose to invest in these areas or projects by ranking them based on their BCR values. Alternatively, planners could consider values associated with projects that are not embedded in BCRs, such as complementarity in meeting explicit conservation objectives or qualitative benefits such as the preferences of local communities. While investments in quadrants I and IV are clear-cut, those in quadrants II and III might require closer consideration and involve explicit trade-offs. Choices are likely necessary, for example, as to whether a single costly project in quadrant II contributes more to objectives than several projects in quadrant III with an equivalent total cost.

Robustness of cost-efficient decisions: explicitly accounting for uncertainty in variables

Uncertainties in data and the associated robustness of decisions have not been discussed by studies using BCRs for conservation prioritization. Because BCRs are ratios of benefits to costs, the uncertainties associated with both components will be compounded and could therefore involve more uncertainty than allocation strategies based solely on maximising benefits (Babcock et al. 1997; Ferraro 2003a). While BCRs might result in more efficient allocation of resources than benefits alone, it is important to consider the implications of uncertainty explicitly. Two sources of uncertainty relate to data resolution (Pressey & Logan 1995; Rodrigues & Gaston 2001) and the informativeness of surrogates (Bode et al. 2008; Grantham et al. 2010).

In this chapter, I use quantitative analyses to explore two of these limitations of BCRs for conservation prioritization, specifically as they influence the selection of properties to meet a

commitment to expand the Queensland protected area system. First, I ask how alternative partial definitions of benefits and costs change selections of areas. This is the first time that the effect of partial definitions of costs on rankings has been explored. Second, given a particular definition of benefits and costs, I ask about the robustness to uncertainty of a selection strategy based on BCR rankings, using info-gap modeling. For both questions, I use benefit-only prioritization, ignoring costs, as a comparison with BCR prioritization to determine whether BCRs are more efficient and robust than benefits alone.

Methods

Study region, data sets, and allocation strategies

The study region (Figure 3.2) included the whole state of Queensland, Australia, with a total area of 185 million ha. About 8 million ha is in the protected area estate. In 2008, the Queensland government promised to add ~12 million ha to the network, 4 million ha of which would be acquired for national parks, bringing the total estate to 20 million ha by 2020 and requiring an estimated $120 million (Anna Bligh Premier of Queensland 2008). In light of this commitment, I explored the potential selections of properties (single or combined ownership parcels) using BCRs with four data sets. I considered all unprotected properties in Queensland with remnant vegetation (n=278,442).

For each property, I calculated the benefit, b_i, using the definition of Fuller et al. (2010):

$$b_i = \sum_{j=1}^{m} a_{ji} / r_j$$

where m is the number of vegetation types, a_{ji} is the area of vegetation type j in parcel i, and r_j is the proportion of vegetation type j remaining since clearing occurred. I defined the cost c_i for each property to be the acquisition cost to the government:

$$c_i = p_i + t_i$$

where p_i is the price of the property (below) and t_i is the transaction cost, assumed to be $15,000 for all properties.

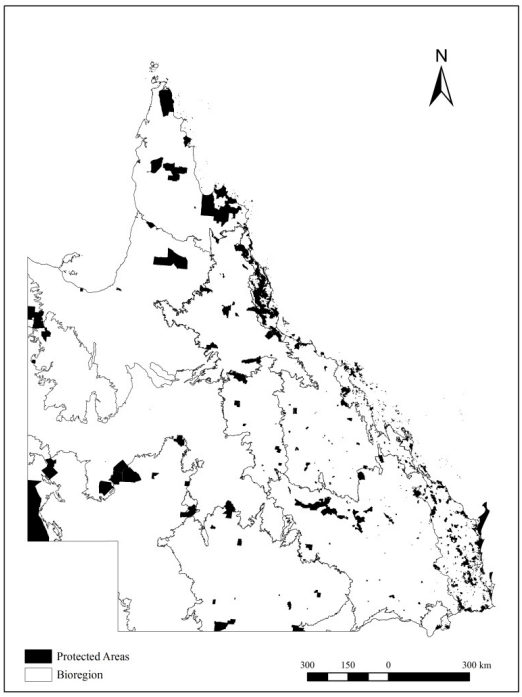


Figure 3.2 The state of Queensland, protected areas, and bioregions.

Data set 1: Benefit b_i based on broad-scale vegetation mapping (60 vegetation types across Australia, 52 of which occur in Queensland) (NVIS 2010). Cost c_i based on land value per property (NRW 2008c) as a surrogate of p_i (Carwardine et al. 2008; Fuller et al. 2010; Klein et al. 2009a).

Data set 2: Benefit b_i based on the 52 broad-scale vegetation types (NVIS 2010). Cost c_i based on estimated sales values (see Chapter 2 for details), reflecting more accurate estimates of p_i.

Data set 3: Benefit b_i based on mapping of about 1300 regional ecosystems (DERM 2009). Cost c_i based on land value per property (NRW 2008c).

Data set 4: Benefit b_i based on mapping of about 1300 regional ecosystems (DERM 2009). Cost c_i based on estimated sales values (see Chapter 2 for details).

I assumed a budget of $120 million to reflect the government's commitment. For each data set, I calculated the benefit, cost and BCR for each property and used two selection strategies: 1. rank properties from highest to lowest BCR; and 2. rank properties from highest to lowest b_i (benefit only). For each strategy, I selected the highest ranked properties until the budget was fully exhausted based on estimated sales value as the more realistic cost. I chose this selection

strategy to critically examine the utility of BCRs applied in the way recommended in a recent study (Fuller et al. 2010). In subsequent work, it would be possible to consider complementarity in the definition of benefits and to formulate the selection method as an optimization problem constrained by achieving representation objectives.

I examined correlations between the different cost and benefit measures because previous studies have demonstrated that the efficiency gains of including costs in prioritization are strongly related to the spatial correlations between costs and benefits and their relative spatial variability (Babcock et al. 1997; Ferraro 2003a). I mapped the properties selected under the two strategies for the four data sets and recorded parameters of each set of selections: total extent and number of properties, number of regional ecosystems, by threat category, with at least 10% protection (as a benchmark for CBD commitments, UNEP 2002), and total cost. I also calculated the spatial correlations between the 8 sets of selected properties (Pearson coefficient of all 278,442 properties, with a value of 1 for included and 0 otherwise).

Info-gap model case study

I used info-gap to measure the robustness of both BCR and benefit-only prioritizations. Info-gap is a decision-theory framework that evaluates the robustness of decisions by testing how far a parameter or function can deviate from the estimated value while still producing an acceptable performance (Ben-Haim 2006). I applied the info-gap model modified from Adams and Pressey (2011) to the selection of properties to supplement the Queensland protected area system. I did this for the two strategies (BCR and b_i) applied to data set 4 (regional ecosystem mapping, sales value), assuming a budget of $120 million.

My robustness measure was the percentage of total properties selected for acquisition that were excluded or included in error. Effectively, had I known the true values of benefits and costs, robustness was the sum of: 1. the percentage of selected properties that I would not have purchased in favour of investing elsewhere; and 2. the percentage of properties not selected that should have been (Adams & Pressey 2011). I selected all properties above a certain BCR or b_i, defined as θ, and calculated the robustness, D_c, for a range of uncertainty values, α (mathematical details in Appendix 2). I plotted α against D_c, to produce a robustness curve for each strategy.

Results

Characteristics of data sets

Costs as land values were weakly but positively correlated with both measures of benefit (Table 3.3a). Costs as sales values were not correlated with benefits. However, benefits were less spatially variable (coefficient of variation c_{NVIS}=1801.30 and c_{RE}=1614.82) than costs (coefficient of variation c_{land}=4111.95 and c_{sale}=31216.30). The two measures of cost were weakly correlated. The two measures of benefit were strongly correlated. To consider whether benefits were being driven by variable property size, I correlated benefits and areas of properties. Both measures of benefit were more strongly correlated with property area than with one another (Table 3.3a).

Table 3.3 Pearson correlation coefficients (p values in parentheses). (a) Correlation between area (ha) of property, land value, sales value, benefit calculated with NVIS national vegetation types, and benefit calculated with REs (Queensland regional ecosystems). (b) Correlation between sets of selected properties. BCR indicates selections based on rankings for benefit-cost ratios. B indicates selections based on rankings for benefits only. For BCR, numbers refer to the datasets described in the Methods: 1. benefits defined based on national NVIS data and costs based on land value; 2. benefits defined based on national NVIS data and costs based on sales value; 3. benefits defined based on Queensland regional ecosystems and costs based on land value; 4. benefits defined based on Queensland regional ecosystems and costs based on sales value. For B (no cost data), NVIS indicates benefit based on national vegetation mapping and REs indicate benefit based on mapping of regional ecosystems in Queensland.

(a)

	Area (ha)	Benefit (NVIS)	Benefit (REs)	Cost (land value)
Benefit (NVIS)	0.928			
	(0.000)			
Benefit (REs)	0.954	0.893		
	(0.000)	(0.000)		
Cost (land value)	0.031	0.066	0.019	
	(0.000)	(0.000)	(0.000)	
Cost (sales value)	0.001	0.001	0.001	0.006
	(0.792)	(0.694)	(0.743)	(0.002)

(b)

	BCR-1	BCR-2	BCR-3	BCR-4	B-NVIS
BCR-2	0.426				
	(0.000)				
BCR-3	0.826	0.443			
	(0.000)	(0.000)			
BCR-4	0.359	0.808	0.424		
	(0.000)	(0.000)	(0.000)		
B-NVIS	0.024	0.048	0.023	0.054	
	(0.000)	(0.000)	(0.000)	(0.000)	
B-Res	0.023	0.058	0.022	0.073	0.609
	(0.000)	(0.000)	(0.000)	(0.000)	(0.000)

Allocation strategies

The eight prioritizations varied spatially (Figure 3.3, Table 3.3b). BCR selections were more similar to one another than to benefit-only selections, while the two benefit-only selections were strongly correlated (Table 3.3b). The stronger spatial similarities of benefits than costs (Table 3.3a) were reflected in the selections. The two pairs of BCR data sets (1/3 and 2/4) that used different benefits but the same costs were strongly correlated. The two pairs (1/2 and 3/4) that used the same benefits but different costs were less strongly correlated.

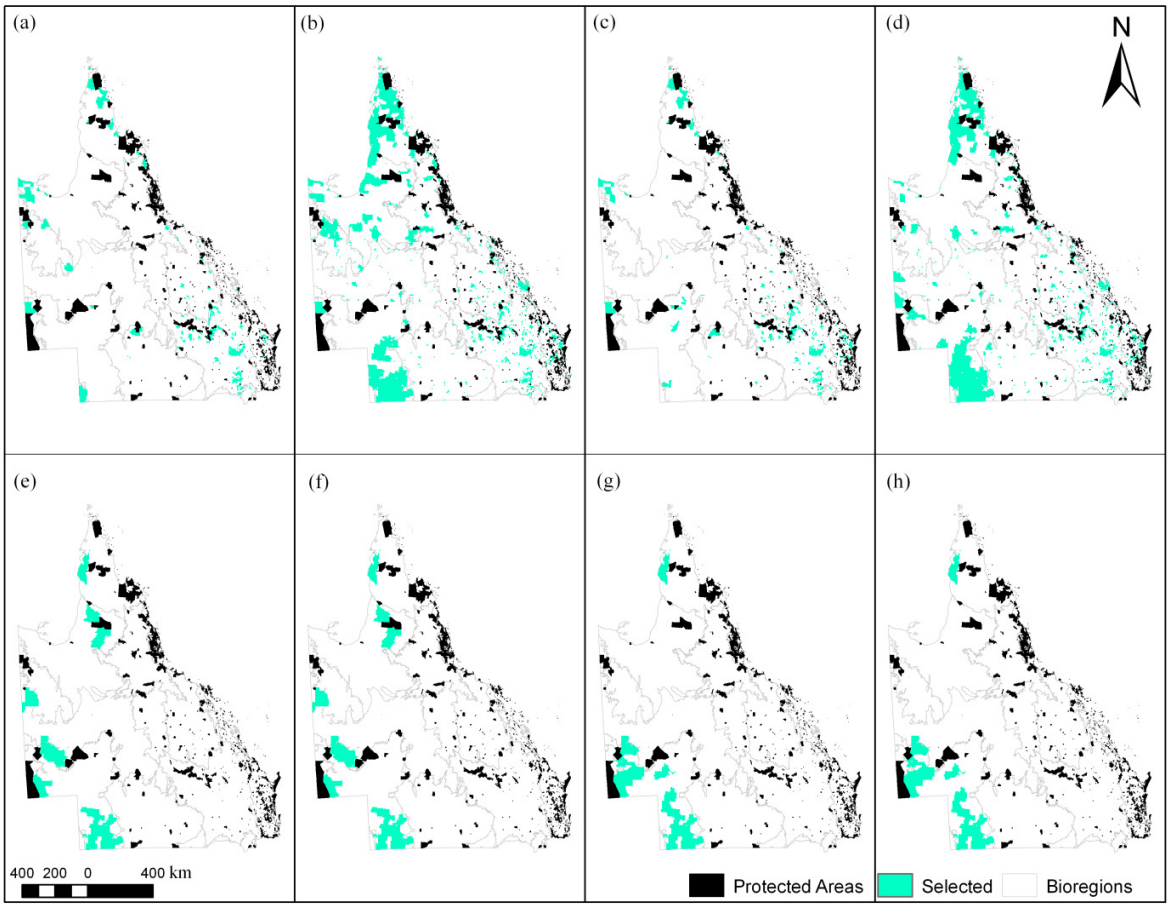


Figure 3.3 Properties selected for addition to the Queensland protected area estate with two selection strategies and four data sets. (a) *BCR selection using data set 1:* benefits defined based on national NVIS data and costs based on land value. (b) *BCR selection using data set 2:* benefits defined based on national NVIS data and costs based on sales value. (c) *BCR selection using data set 3:* benefits defined based on Queensland regional ecosystems and costs based on land value. (d) *BCR selection using data set 4:* benefits defined based on Queensland regional ecosystems and costs based on sales value. (e) *Benefit selection using data set 1:* benefits defined based on national NVIS data and costs based on land value. (f) *Benefit selection using data set 2:* benefits defined based on national NVIS and costs based on sales value. (g) *Benefit selection using data set 3:* benefits defined based on Queensland regional ecosystems and costs based on land value. (h) *Benefit selection using data set 4:* benefits defined based on Queensland regional ecosystems and costs based on sales value.

BCR selections were more efficient than benefit-only selections, selecting more properties and providing a minimum of 10% protection to more regional ecosystems (Table 3.4). In addition, BCRs selected properties in more bioregions than benefit-only selections, while benefit selections were clumped in the south west of the state where properties are very large (Figure 3.3). Including higher-resolution regional ecosystems resulted in marginally more regional ecosystems being protected with a minimum of 10% compared to the NVIS vegetation data for both benefit and BCR prioritizations (Table 3.4). BCRs that used sales values selected many more regional ecosystems to at least 10% than BCRs that used land values (Table 3.4). The same large differences applied to selection of threatened regional ecosystems (those categorized by DERM 2009 as Of Concern or Endangered).

Table 3.4 Characteristics of sets of selected properties using different measures of benefits and costs. Total area selected, number of properties selected, number of regional ecosystems (REs) with at least 10% protected (out of a total of 1,308), number of REs with at least 10% protected that were categorized as Of concern (OC) or Endangered (E) (a total of 583 REs in these categories and total cost in sales value. (a) BCR selections for all four data sets. (b) Benefit-only (B) selections for all four data sets. For benefit only (B), because costs were not considered in these selections, the results for data sets 1 and 2 (NVIS national vegetation types) are identical, as are the results for data sets 3 and 4 (Queensland regional ecosystems).

(a)

	BCR: data set 1	**BCR: data set 2**	**BCR: data set 3**	**BCR: data set 4**
Total area selected (ha)	5,112,966	19,095,339	5,315,507	18,921,675
Number of properties selected	89	619	116	764
Number of REs with at least 10% protected	123	360	126	368
Number of REs with at least 10% protected which are OC or E	38	120	38	125
Total cost based on sales vale	119,801,000	115,683,000*	109,839,000*	115,888,000*

(b)

	B: data set 1	**B: data set 2**	**B: data set 3**	**B: data set 4**
Total area selected	8,346,138	8,346,138	8,437,116	8,437,116
Number of properties selected	11	11	12	12
Number of REs with at least 10% protected	58	58	59	59
Number of REs with at least 10% protected which are OC or E	8	8	7	7
Total cost based on sales value	119,884,000	119,884,000	94,392,000*	94,392,000*

*Total cost was less than $120 million as the next ranked property would have exceeded the allocated budget.

Despite using a benefit measure that weighted areas by the proportion of vegetation remaining since clearing to prioritize vegetation that has been heavily cleared, neither BCR nor benefit-only strategies represented threatened regional ecosystems in proportion to their listing. Compared to 45% of regional ecosystems listed as threatened across the state, for example, benefit prioritization with regional ecosystems selected only 7 threatened ecosystems out of a total of 59 (~12%), and BCR with data set 4 selected 125 out of a total of 368 (~33%).

Info-gap model

The BCR prioritization was more robust than the benefit-only prioritization for all except one level of uncertainty (α) (Figure 3.4). At 13% uncertainty in my variable estimates ($\alpha = 0.13$), ~42% of both the BCR portfolio (320 of 764 areas selected) and the benefit-only portfolio (5 of 12 areas selected) would be misallocated. Alternatively, if 20% misallocation is acceptable ($D_c = 20\%$), then there could be up to 7% uncertainty in the estimates of b and c ($\alpha = 0.07$) for the BCR prioritization, but only 2% ($\alpha = 0.02$) uncertainty in the estimate of b for the benefit-only prioritization.

Discussion

There are increasing calls for transparent, cost-efficient decisions in conservation planning (Naidoo et al. 2006). Benefit-cost ratios offer one approach, but come with limitations and uncertainties that need to be better understood. I discuss below the implications of using alternative surrogates for benefits and costs and ways of understanding the robustness of BCRs to uncertainty.

Effects of correlations and variability of data on selections

I measured the performance of the different selection strategies and data sets by comparing the number of regional ecosystems and threatened regional ecosystems with a minimum of 10% protection. BCRs were more efficient than benefit-only prioritizations according to both performance measures. However, depending on the benefit and cost measures used, my four BCR strategies varied in the number of regional ecosystems with a minimum of 10% protection. Because the more variable data components will drive selections (Babcock et al. 1997), my selections were largely driven by the costs rather than benefits of properties. Therefore,

including higher-resolution benefit data only marginally increased the number of regional ecosystems with at least 10% protection compared to the broad-scale NVIS data. In contrast, including cost data that more accurately reflected acquisition costs (i.e. sales data instead of land values) resulted in a three-fold increase in protection (Table 3.4).

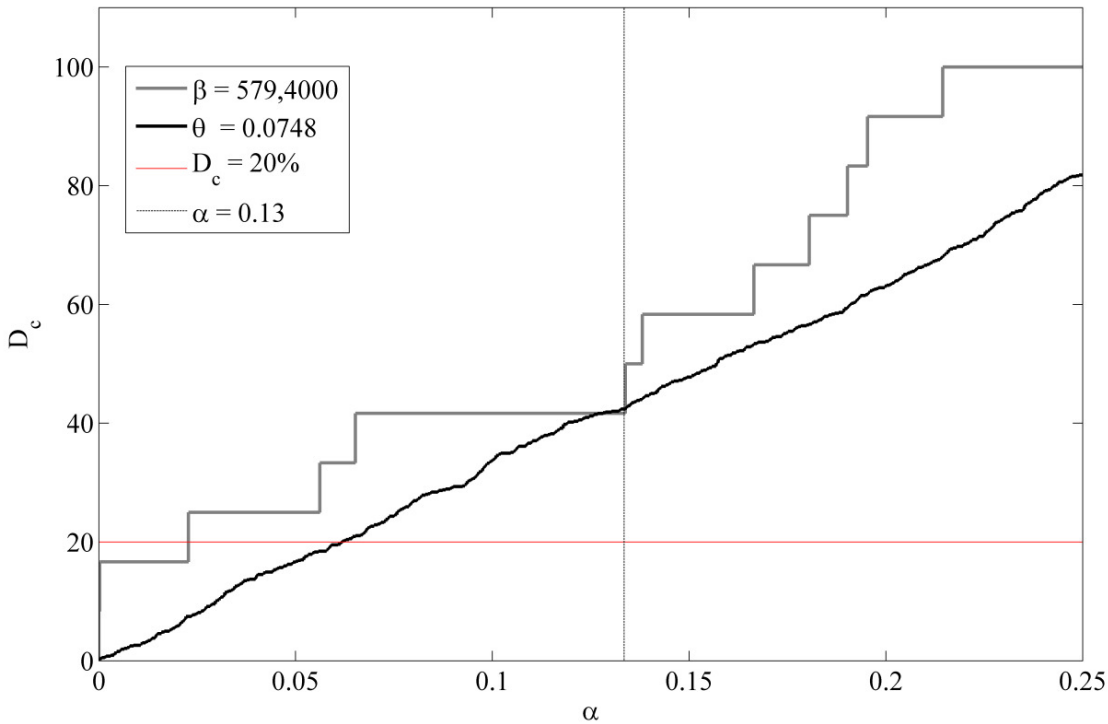


Figure 3.4 Robustness of two allocations strategies: BCR and benefit ranking for acquisition of new protected areas in Queensland. D_c is percentage of the properties selected for acquisition that were excluded or included in error and α is the level of uncertainty in my estimates of b and c. Critical values θ and β were set based on a budget of $120 million. The robustness curve for the BCR allocation is shown in black and corresponds to the budget-driven θ = 0.07485. The robustness curve for the benefit allocation is shown in grey and corresponds to the budget-driven β = 579,400. A vertical grey line at α =0.13 is drawn for reference as this is the value at which the robustness of the two strategies is equal. An example of a critical threshold D_c is shown for 20% of the portfolio being misallocated (horizontal red line). The points at which the critical threshold line intersects the robustness curves indicate the levels of uncertainty (α) that can exist in my parameters while still giving an acceptable number of misallocations. For BCR (black line, budget-driven θ, and D_c = 20%), this level of uncertainty (α) is 0.07. For benefit allocation (grey line, budget-driven β, and D_c = 20%), this level of uncertainty (α) is 0.02.

Surrogates for benefits

I used the benefit definition developed by Fuller et al. (2010) which they applied to subcatchments of relatively uniform size. Fuller et al. reported that this measure performed well, meeting targeted protection for 54 of 58 vegetation types. However, in my case study, this

benefit formula performed less well. The principal reason seems to be the large variation in sizes of properties - the actual units of acquisition and management - across Queensland, ranging 0.01 to 1 million ha. This size range caused the benefit measure to reflect property size more than the presence of heavily cleared vegetation, for which it was designed, as indicated by the strong correlation between property size and benefit measures (Table 3.3a). This result indicates the need for a better measure of benefit, particularly when property sizes are highly variable. A more appropriate benefit measure would not be directly related to the sizes of properties, unlike the measure of Fuller et al. (2010), but might instead reflect the contributions of properties to quantitative objectives for vegetation types and the likelihood of further loss or decline in condition of vegetation.

Surrogates for costs

Sales values of properties are likely to reflect acquisition costs much more accurately than land values, and land value was a poor surrogate for sales value in my study. The two measures of cost were uncorrelated, and sales value was much more variable spatially than land value. Therefore, using land value as a surrogate for sales value resulted in a less efficient selection of properties (~ 5 million ha selected within the budget) and lower coverage of regional ecosystems than when sales value was used directly (selecting ~ 20 million ha and protecting three times as many regional ecosystems).

Problems with surrogates: neglecting the broader socio-economic context

I selected my costs to reflect the typical measures used in the literature (Carwardine et al. 2008; Fuller et al. 2010) as well as the Queensland government's financial promise. My analyses used surrogate measures of acquisition cost but neglected the other possible costs and savings associated with acquisition, establishment, and management of conservation reserves (Table 3.2). This means that my property selections, while efficient for acquisition costs, could increase other, presently hidden costs compared to selections informed by complete accounting. If I assume an average per ha management cost of $8.32 across Queensland (for details see Adams et al. 2011), then the BCR prioritization will increase annual management costs by $157 million (or over the next 20 years, assuming 3% inflation, by $1 billion). In reality, per-unit-area management costs of reserves will vary strongly according to configuration, use and landscape

context (Balmford et al. 2003; Frazee et al. 2003). Anticipating this variation would cause selections to minimize the sum of acquisition and management costs, and probably alter the selections reported here. Additional benefits could be achieved if spatial variation in other aspects of social and economic costs and benefits were considered.

Making robust conservation decisions: how to reduce the number of misallocations

The use of BCR allows for cost-efficient allocations, yet uncertainties in estimates might cause actual outcomes of projects to differ from expected outcomes. This could result from using surrogates of full costs and benefits of projects, which shift allocations of funds between projects. It could also result simply from the inevitable uncertainties in estimates of any benefits and costs. My case study demonstrates that BCRs are more cost-efficient and robust than benefit-only selections. BCR selected an order of magnitude more properties and the much larger absolute number of misallocations still represented a smaller proportion of the selected portfolio than the misallocations from the benefit-only prioritization.

Even with its greater robustness, however, my BCR still misallocated many projects. At an uncertainty level of 10%, at which managers would have 10% error in their benefit estimates such as vegetation mapping and 10% error in their cost estimates such as sales values, my BCR mistakenly indicated investment in 259 properties across Queensland. Ground-truthing would solve only part of this problem. During the process of site inspection and adjustment of selections, from BCR or any other approach (e.g. Cowling et al. 2008), misincluded properties would be identified, but properties that had been misexcluded would not. These are properties that, given better data, would have had higher BCR rankings than some included properties. At 10% uncertainty in my example, there were 154 misexclusions. Uncertainty in BCRs therefore remains a problem for planners to address.

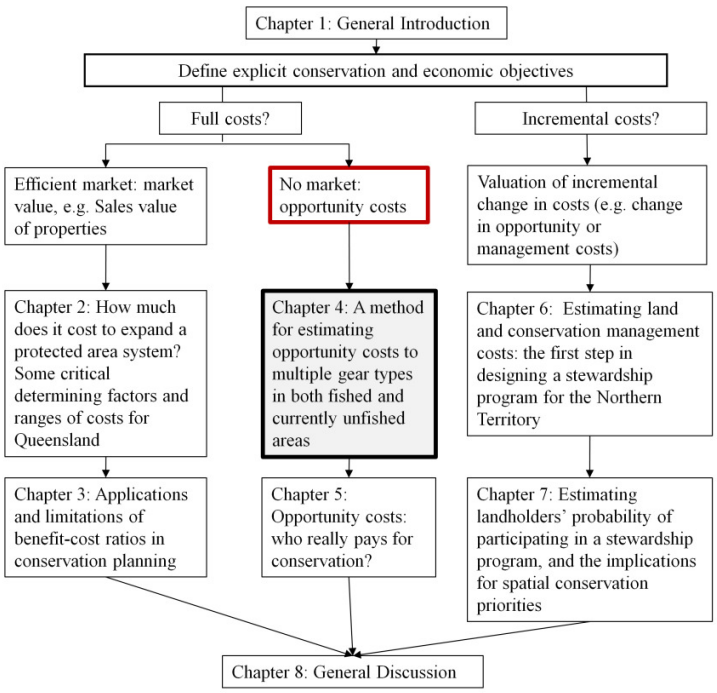
Conclusions

My analysis demonstrates how both benefit and BCR prioritizations can be affected by the definitions of benefits and costs. Neither the benefit nor BCR prioritizations were effective at meeting a minimum level of protection, such as 10%, for all regional ecosystems. That result was not unexpected. The difficulty of meeting representation targets with ranking procedures

has been demonstrated (Pressey & Nicholls 1989), and this has long been an argument for selecting sets of parcels using an optimization strategy directed at achieving representation objectives. My results are in contrast to those of Fuller et al. (2010) who found that their simple ranking method met representation objectives for most vegetation types. Their finding is likely to be atypical and could be related to the very coarse thematic resolution of the vegetation data they used and low variability in size of their selection units. In addition, the formulation of benefit by Fuller et al. (2010) failed to preferentially select vegetation types that had been heavily cleared (regional ecosystems categorized as Endangered or Of Concern), so a better benefit measure is needed for Queensland and, probably, other regions. The BCR prioritization was also sensitive to the cost measure, with a three-fold increase in protection of regional ecosystems, and different spatial priorities, when using sales value as opposed to land value. My application of the info-gap model demonstrates that using BCR prioritization results in uncertainties that are not explicitly considered in the basic implementation of the prioritization. Therefore, managers wishing to use BCR should explicitly consider potential shifts in prioritizations depending on how benefits and costs are defined and how much certainty is involved in their definition.

Chapter 4 A method for estimating opportunity costs to multiple gear types in both fished and currently unfished areas[1]

I present a novel method for calculating the opportunity costs to fishers from their displacement by the establishment of marine protected areas (MPAs). I used a fishing community in Kubulau District, Fiji to demonstrate this method. I modelled opportunity costs as a function of food fish abundance and probability of catch, based on gear type and market value of species. Count models (including Poisson, Negative Binomial and two zero-inflated models) were used to predict spatial abundance of preferred target fish species and were validated against field surveys. A profit model was used to investigate the effect of restricted access to transport on costs to fishers. Spatial distributions of fish within the three most frequently sighted food fish families (Acanthuridae, Lutjanidae, Scaridae) varied, with greatest densities of Lutjanidae and Acanthuridae on barrier forereefs and greatest densities of Scaridae on submerged reefs. Modelled opportunity cost indicated that highest costs to fishers arise from restricting access to the barrier forereefs. I included the opportunity cost model in Marxan, a decision support tool used for MPA design, to examine potential MPA configurations for Kubulau District, Fiji Islands. I identified optimum areas for protection in Kubulau with: (a) the current MPA network locked in place; and (b) a clean-slate approach. My method of modelling opportunity cost gives an unbiased estimate for multiple gear types in a marine environment and can be applied to other regions using existing species data.



[1] Adams, V. M., M. Mills, S. D. Jupiter, and R. L. Pressey. 2011. Improving social acceptability of marine protected area networks: a method for estimating opportunity costs to multiple gear types in both fished and currently unfished areas. Biological Conservation **144**:350–361.

Introduction

The rapid degradation of marine resources resulting from human activity has motivated a global movement to increase the protection of the oceans (IUCN 2009). The establishment of marine protected areas (MPAs) is a widespread and acknowledged tool for conserving biodiversity and providing ecosystem services, with widely reported increases in biomass of fisheries resources, size of target species, and species richness within protected areas (Lester et al. 2009). However, these benefits will only be realized through effective protected area design as well as compliance and enforcement.

Systematic conservation planning can account for trade-offs between benefits and social costs during the selection of areas by explicitly defining biodiversity, fisheries and socio-economic goals (e.g. Ban & Klein 2009; Gaines et al. 2010). Systematic methods are currently preferred for designing MPA networks in developed countries (Sala et al. 2002). Despite progress in the technical aspects of designing effective networks, recent research highlights the need to more effectively include socio-economic data in conservation planning with the aim of increasing the potential to implement plans (Naidoo et al. 2006; Polasky 2008).

Use of socio-economic information is especially important in the context of developing countries where these data are generally limited and social acceptance is a critical factor in determining MPA success (Ban et al. 2009; Johannes 1998). While inclusion of socio-economic data in the design of MPAs has increased in the last decade (Ban & Klein 2009), spatial variation in costs to stakeholder groups needs to be better understood (Adams et al. 2010; Klein et al. 2008a). This is especially important in Pacific island countries where: communities are highly dependent on marine resources for subsistence (Adams et al. 1997); many fishers have limited spatial and occupational mobility (e.g. Cinner et al. 2009); and customary marine tenure places social and governance constraints on MPA network design (Aswani & Hamilton 2004).

The most prevalent type of socio-economic data in marine conservation planning relate to fisheries catch (Ban & Klein 2009). Catch per unit effort (CPUE) data are typically derived from log books in regulated fisheries or, more commonly in developing countries, from socio-economic survey questions about fishing location, gear used, and the type and amount of fish

caught. These data reflect the spatial distribution of fishing effort and can be combined with ecological surveys to examine the effects of fishing on fish community composition (Jennings & Polunin 1996). The collection of CPUE data is becoming standard practice, particularly for commercial fisheries within developed countries (Ban & Klein 2009), while national-scale collection of artisanal and subsistence CPUE data is possible at relatively low-cost in developing countries (IAS 2009). Recent studies have examined how to use CPUE data to plan for multiple fisheries so that MPAs do not displace opportunity costs disproportionately onto single stakeholder groups (Klein et al. 2008a).

One limitation of CPUE data is that, by definition, they only capture the current fishing effort. This implies, often wrongly, that existing MPAs have no opportunity cost. It also ignores areas without current effort that might be suitable for harvest if currently fished areas became unavailable through the establishment of MPAs. A response to these limitations in conservation planning has been to define cost as a function of both effort and extent of areas being considered for conservation to ensure that no potential MPAs are seen as having zero opportunity cost (e.g. Game et al. 2008). However, for areas with no current fishing effort, assuming homogeneous per-unit-area cost does not capture the potential heterogeneity of effort. Areas outside MPAs can lack current fishing effort for several reasons, including the common inability of survey data to capture the seasonality of fishing distributions, and the lack of access to motor boats, particularly in developing countries (Salas & Gaertner 2004). As access to motor boats increases, it is reasonable to expect that fishing effort will change and move further offshore to areas currently not fished.

Because of the limitations of CPUE data as a basis for conservation planning, a method is needed to estimate the opportunity costs of areas that are currently not fished. Decisions about MPA size and configuration can then account for the heterogeneous opportunity costs to fishers in both fished and currently unfished areas. A method has been developed to estimate the opportunity costs to agriculture of forested parts of landscapes (Naidoo & Adamowicz 2006), but an analogous method has not been published for marine environments. In response to this need, I present a novel method for estimating the opportunity costs of establishing MPAs to groups of rural fishers using multiple gear types. I describe the model and then demonstrate its application

with a case study from Kubulau District, Fiji, to show how the resulting map of opportunity costs can be used with conservation planning software to recommend a new, more socially acceptable, configuration of community-managed MPAs. I address four main questions that are relevant not only in Kubulau but also for fisheries management globally:

(1) Where are the preferred target fish species located?
(2) Where is current fishing effort focused and how does it vary by gear type?
(3) What are the differences between current and potential opportunity costs?, and
(4) How can the current MPA network be modified to reduce conflict with users?

Methods

Study area

Kubulau District is an administrative unit of Bua Province, in south west Vanua Levu, Republic of Fiji Islands (Figure 4.1). Traditional fishing grounds (*qoliqoli*) in Fiji have been legally demarcated by the Fiji Native Lands and Fisheries Commission. The 261.6 km^2 of inshore waters within the Kubulau qoliqoli contain a diverse array of habitats, including reef flats, seagrass beds, coastal fringing reefs, soft bottom lagoons, patch reefs, offshore barrier reefs and deep channels. Estimates of biomass of targeted food fish from underwater visual census (UVC) surveys along Kubulau forereefs between 2007 and 2009 range from 0.04 to 15.8 tonnes ha^{-1} (WCS, unpublished data).

In 2005, the communities of Kubulau formally established a network of village-managed (*tabu*) areas and MPAs covering nearly 80 km^2 of the qoliqoli, including 17 tabu areas and 3 MPAs (Namena, Nasue and Namuri; Clarke & Jupiter 2010). Tabu areas may be periodically harvested by owners of traditional fishing rights at the discretion of the village chief, while the MPAs are permanently closed. The initial design of the tabu areas and MPAs was informed by both socio-economic and biological research undertaken by local managers and their conservation partners – the Wildlife Conservation Society, WWF, Wetlands International-Oceania and the Coral Reef Alliance. A ridge-to-reef management plan was completed for Kubulau District in July 2009 and has been endorsed by the council of chiefs (WCS 2009). However, lack of consideration for the traditional fishing rights of certain clans has created conflict over access to some closed areas,

with violent altercations in at least one case between community fish wardens and locals wanting to fish in an MPA (Clarke & Jupiter 2010).

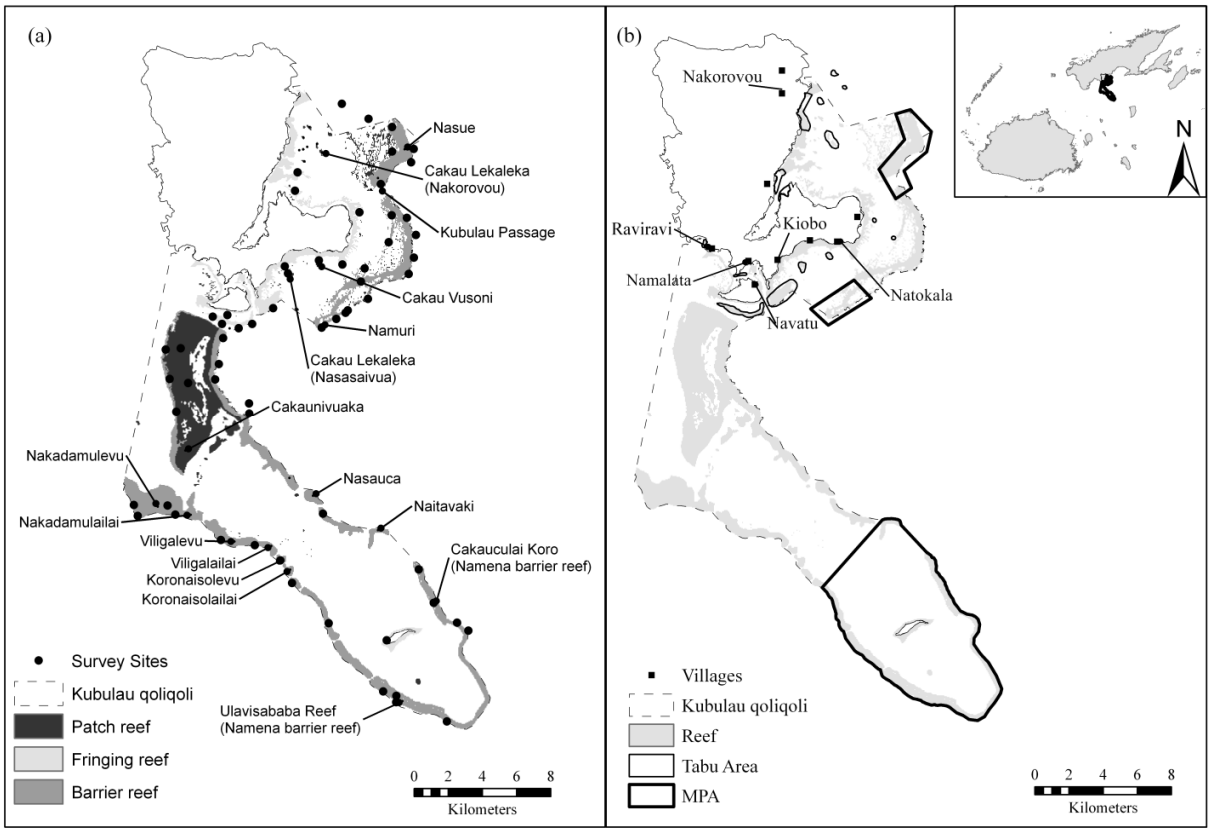


Figure 4.1 The study area, Kubulau District, Vanua Levu, Fiji. Inset shows map of Fiji with Kubulau District and traditional fishing ground (qoliqoli) demarcated for reference. (a) Reef habitats (patch, fringing and barrier) in the Kubulau qoliqoli and names of reefs. Sites of biological surveys are also shown. (b) Villages in Kubulau District with labels indicating villages surveyed for CPUE data (Navatu, Kiobo, Nakorovou and Raviravi) and villages with motorized vessels (Navatu, Namalata, Kiobo, Natokalau, Nakorovou). Current tabu areas and MPAs are delineated.

The total population of Kubulau District is approximately 1,000 people distributed across 10 villages and 1 settlement. Presently only 5 of the coastal villages (Navatu, Namalata, Kiobo, Natokalau, Nakorovou) have motorized vessels for fishing. In addition, one motor boat has been donated to the entire district by a local NGO for enforcement and is occasionally used for fishing. Six fishing gear types were identified, with preferred gear types including gill nets, spearguns and hand line (Appendix 3, Table A3.1).

Opportunity cost and profit models

Figure 4.2 shows a flow diagram of the data inputs, intermediate models and steps for the full models of opportunity cost and profit. The following sections describe the parts of the flow diagram.

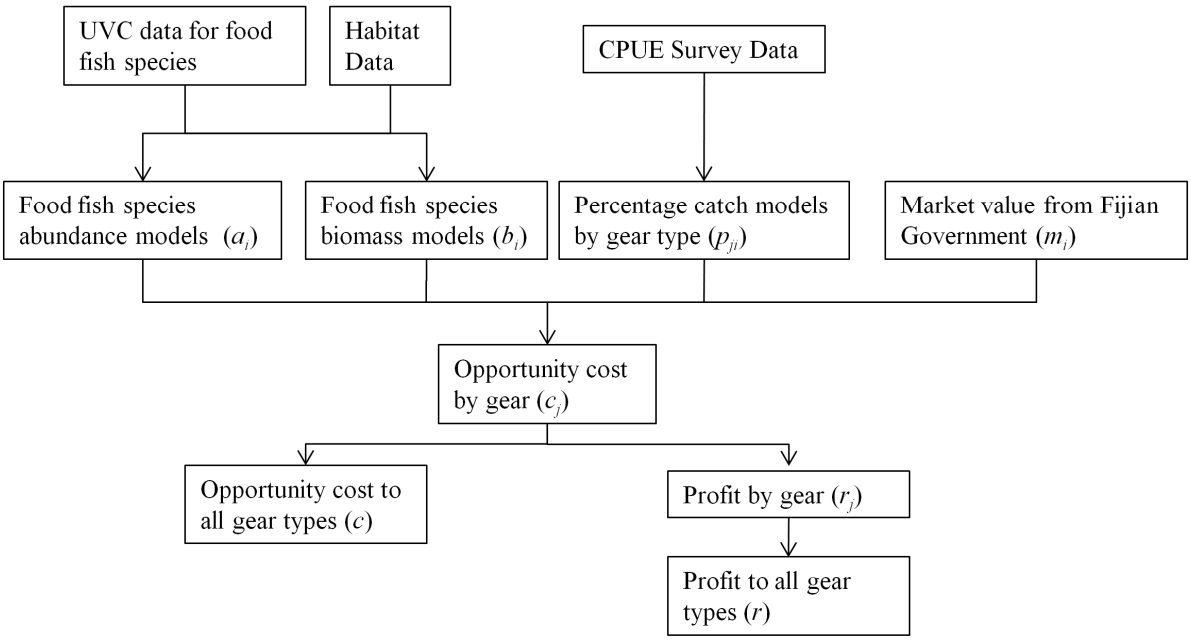


Figure 4.2 Flow diagram of data inputs and model steps for opportunity cost and profit models for all gear types.

UVC data for food fish species

In April-May and September 2009, underwater visual census (UVC) surveys were carried out at 63 locations within the Kubulau qoliqoli. Observers measured fish abundance and size within the following families that are targeted for consumption and sale: Acanthuridae, Balistidae, Carangidae, Haemulidae, Lethirinidae, Lutjanidae, Scaridae, Serranidae (groupers only) and Siganidae. Sites were chosen to maximize spatial representation across reef habitats with a minimum of three replicate transects per site, typically distributed across depth categories. Measurements of fish size (total length) and abundance were scored along replicate (n = 3 to 5) 5 m x 50 m belt transects. Transects were deep (12 -15 m) and shallow (5 m – 8 m) at most forereef sites, and shallow and reef-top (1 - 3 m) at backreef sites. Each sighted fish >2 cm was classified to species level within size categories (2-5, 6-10, 11-15, 16-20, 21-25, 26-30, 31-35, 36-40 cm). The length of fish >40 cm was estimated to the nearest cm to improve estimates of

biomass. Biomass was calculated from size class estimates of length (L_T) and existing published values from Fishbase (Froese & Pauly 2009) used in the standard weight-length expression $M = aL_T^b$, with *a* and *b* values preferentially selected from sites closest to Fiji (e.g. New Caledonia). If no length-weight (L-W) conversion factor was present for the species, the factor for a species of similar morphology in the same genus was used (Jennings & Polunin 1996). If a suitable similar species could not be determined, the average for the genus was used. Because most of the New Caledonia fishes were measured to fork length (FL), a length-length (L-L) conversion factor was obtained from Fishbase where possible to convert from total length (TL) to FL before biomass estimation. Food fish species (n = 148) were selected from the recorded families for spatial modelling of abundance throughout the qoliqoli (see Food fish species abundance models section).

Habitat data and other predictor variables

I focused on coral reef habitats for my modelling because: (1) the majority of targeted fish biomass within the qoliqoli was found to be associated with coral reefs; and (2) relatively comprehensive spatial layers were available for exposed and submerged coral reefs but not for other habitats. Exposed and submerged coral reefs were digitized by the Fiji Department of Lands from aerial photographs taken in 1994 and 1996. Where data were missing, I digitized exposed reefs from Fiji topographic map sheets at 1:50,000 scale but were unable to digitize additional submerged reefs. I classified reefs primarily as barrier, patch or fringing based on previous surveys and reference to Landsat data (Table 4.1). To derive biophysical predictor variables for species abundance models, I further classified the reef types by: exposure to tides; exposure to waves; and depth (Appendix 3, Table A3.2). Depth was classified as shallow (depth ≤ 10 m) or deep (depth > 10 m) based on contour lines from digitized nautical charts. Other predictor variables were protection status and linear distance from shore (Appendix 3, Table A3.2). Distance from shore for each survey location was calculated using ArcInfo 9.2 (ESRI) software.

Table 4.1 Reef habitat type, percentage of total reef cover in study region, area in study region, percentage of survey sites allocated, and conservation targets.

Habitat	Percentage of total reef cover in study region	Area of habitat in study region (km^2)	Percentage of survey sites	Conservation target (%)	Conservation target (km^2)
Barrier	54%	36.9	63.5%	30%	11.1
Fringing	17%	20.1	19.0%	30%	6.0
Patch	29%	11.7	17.5%	30%	3.5

Food fish species abundance models

Food fish species count and biomass were taken from the UVC survey data and pooled by site and standardized for number of transects. Because species abundance data are often characterized by a large number of zeros, zero-inflated models have recently been developed which allow for the concurrent estimation of occurrence and abundance (e.g. Joseph et al. 2009a). For each species I considered four different models: Poisson (P), negative binomial (NB), zero-inflated Poisson (ZIP) and zero-inflated negative binomial (ZINB) (see Appendix 3 for mathematical details). The Poisson and negative binomial models are standard count models used to estimate abundance. The zero-inflated models simultaneously model probability of detection and abundance. Because Poisson models can be sensitive to low numbers I selected species for modelling that occurred in at least 15% of survey sites. This allowed me to use data on 54 species from 22 genera (Appendix 3, Table A3.3).

I fitted the models using the general linear model (glm) and zero-inflated model (zeroninfl) functions in R (R Development Core Team 2005). These packages use maximum likelihood to estimate coefficients for the generalized linear models (Poisson and negative binomial) and zero-inflated models. I used forward and backward removal to select the best subset of predictors for each model (see Appendix 3, Table A3.3 for predictors selected for each species). For each species I selected the best model by comparing proportion of zeros (predicted zeros/observed zeros) and Akaike's information criterion (AIC, Akaike 1974). The best-fitting model has the lowest AIC (see Appendix 3 for discussion of goodness of fit). Abundance was predicted for each species across the qoliqoli on a 50 m grid to reflect abundance data pooled by site and standardized to abundance per 2,500 m^2 (50 m X 50 m). To determine abundance by family and fish price class I predicted abundance by species and summed abundances for these groupings.

Food fish species biomass models

Because many of the food fish species exhibit ontogenetic movements from inshore to offshore, I modelled biomass for each species to capture heterogeneous distribution of biomass across habitats (e.g. Lutjanus fulviflamma juveniles are found in mangrove estuaries while adults commonly school in deep lagoons; Froese & Pauly 2009). Based on UVC data, I estimated biomass for each species with multiple regression, using both forward and backward stepwise removal to select the best fit model (Appendix 3, Table A3.4). All predictors considered for abundance were included in the biomass models.

CPUE survey data

Between May 2008 and June 2009, area specific catch per unit effort (CPUE) information (in catch person^{-1} hr^{-1} m^{-2}) was collected from fishers from four villages (Raviravi, Navatu, Kiobo, Nakorovou) within Kubulau District. Trained community volunteers recorded information once per week from all fish landings in the village during a 24 hour period. Fishers were asked for information on the total number of fish caught, the number of people fishing, the time spent fishing, the gear used for fishing, and the transport used for fishing. All participants (n = 191 total) were asked to indicate where they fished on a map. Of the 191 fishers, 72 drew polygons for fishing areas and the others identified their fishing spots with points. In cases where one of the polygons was associated with more than one gear type or method of transport, only the most efficient gear type and method of transport was chosen to represent the polygon to avoid underestimating opportunity costs.

To create a single layer to represent fishing effort, all fishing spots identified as points were translated into polygons with an area equivalent to that of the largest drawn polygon with the same combination of transport and gear, to ensure that the extent of fishing grounds was not spatially underestimated (Appendix 3, Table A3.1). A final uniform CPUE was calculated for each polygon by dividing the catch by the number of people fishing, time spent and fishing area.

Percentage catch models by gear type

For each gear type I compared the predicted abundance of food fish species to the number of fish caught as reported in CPUE surveys and expressed this as percentage catch (catch/abundance).

Percentage catch was modelled spatially by gear type across the entire qoliqoli using multiple regression analysis, where AIC was used to determine the best subset of predictors (Appendix 3, Table A3.5 and A3.6).

Market value from Fijian Government

Market value, or sale price, of species from 2009 was obtained from the closest Fiji Department of Fisheries district office in Savusavu (Table 4.2).

Table 4.2 Fish price as set by the Fiji Department of Fisheries branch office in Savusavu, Vanua Levu, Fiji.

Class	Price (FJD/kg)	Family
A	$3.00	Lethrinidae, Serranidae, Siganidae
B	$2.50	Carangidae, Haemulidae, Lutjanidae
C	$2.00	Acanthuridae, Scaridae, Balistidae

Opportunity cost by gear

To estimate the opportunity costs to fishing, I considered the gear types in the region and the food fish species for each as identified by catch records in the CPUE data. Based on the food fish species identified for each gear type, I defined the opportunity cost to gear type j to be c_j

$$c_j = \sum_{i=1}^{n} p_{ji} a_i b_i m_i$$

where n was the number of species for gear type j, p_{ji} was the percentage catch for gear type j of species i, a_i was the expected abundance of species i, b_i was the expected biomass of species i, and m_i was the market value of species i. Opportunity cost by gear type was estimated for all reefs on a 50 m grid to match the outputs of models of abundance and biomass.

Opportunity cost to all gear types

I defined the opportunity cost, c, of each 50 m reef grid cell as the sum of opportunity costs to all gear types weighted by the current proportion, w_j, of the total number of fishers represented by each gear type in the fishery:

$$c = \sum_{j=1}^{J} w_j c_j$$

where w_j was calculated as the number of fishers using gear type j divided by the total number of fishers. This approach captured the current distribution of gear types in the fleet, recognising that a fishing site is not exclusively available for fishing by any one gear type.

Profit by gear

In developing countries, such as Fiji, many of the fishers are subsistence and therefore are unlikely to be market driven (e.g. Cakacaka et al. 2010). Also, not all opportunity costs are presently realized because most transport types restrict access to many reefs and fuel costs limit the attractiveness of using boats to fish on distant reefs. Therefore, I wanted to ensure that my model was capturing both current fishing effort as well as potential effort indicated by modelled opportunity cost. I incorporated input costs and differential access by considering expected profit, r_{jm}, from catches at each 50 m reef grid cell by gear type, j, with transport type, m. I excluded time costs, such as forgone revenue from other activities, in my profit model because occupational mobility in this region is limited. I restricted my consideration of input costs to supplies for motorized transport, in this case only fuel. For transport by boats, I considered profit by gear type, j, to be

$$r_{jm} = \max(c_{jm} - f, 0)$$

where c_{jm} is the opportunity cost and f is the fuel cost. Fuel costs were estimated at FJD$0.78 per km using a 25 hp engine (based on expected average price of $1.45 per litre, Fijian Government February 2009). For each gear type, j, with transport type, m, I set profit equal to zero for distances beyond the maximum possible distance travelled with transport type m and, in the case of boats, where profit becomes negative. For each 50 m reef grid cell the expected profit, r, for a gear type j is the weighted sum of transport types, m:

$$r_j = \sum_{m=1}^{M} t_{jm} r_{jm}$$

where t_{jm} is determined by the current distribution of transport types, m, for gear type j and calculated within each group of fishers using a particular gear type, j, as the number of fishers with transport type m divided by the total number of fishers.

Profit to all gear types

I defined the profit, r, of each 50 m reef grid cell as the sum of profits to all gear types weighted by the current proportion, w_j, of the total number of fishers represented by each gear type in the fishery:

$$r = \sum_{j=1}^{J} w_j r_j$$

where w_j was calculated as the number of fishers using gear type j divided by the total number of fishers.

Spatial correlations for costs and profit

To compare the opportunity cost and profit models with the CPUE data, Spearman's rank correlation was calculated by gear type and for total catch across 250 m grid cells in which CPUE data were present (n = 952). For these correlations, I aggregated my modelled data to 250 m grids because this was the size of the smallest reported fishing ground. As my opportunity cost and profit models showed evidence of positive spatial autocorrelation I only considered the correlation coefficients, as these are unaffected by autocorrelation, and do not report the significance values (Balmford et al. 2001; Nhancale & Smith 2011). However, the CPUE data did not show evidence of spatial autocorrelation, therefore I report significance values associated with correlation coefficients for CPUE data by gear type.

Design of cost-effective MPAs

I used Marxan software (Ball et al. 2009) to explore options for design and reconfiguration of a cost-effective MPA network for Kubulau that met the conservation targets for all reef types (Table 4.1). The conservation target of 30% was based on the Fijian Government's declaration at the Barbados Plan of Action in Mauritius in 2005 to protect 30% of its inshore waters. I used a 50 m grid for my planning units to match the resolution of my modelled opportunity costs. For each 50 m grid, I recorded the type of reef habitat and costs based on CPUE and estimated opportunity cost. I selected CPUE as a cost measure to reflect current fishing effort. I selected opportunity cost to capture the expected fishing distribution as access to motorized transport increases and fishing behaviour becomes more market-driven. I considered four scenarios:

Scenario 1- I used CPUE as the cost layer and did not include current tabu areas and MPAs (***clean slate CPUE)***.

Scenario 2 – I used CPUE as the cost layer and required that current tabu areas and MPAs were included in the reconfigured MPA network (***locked in CPUE*).**

Scenario 3- I used opportunity cost as the cost layer and did not include current tabu areas and MPAs (***clean slate Opp***).

Scenario 4 – I used opportunity cost as the cost layer and required that current tabu areas and MPAs were included in the reconfigured MPA network (***locked in Opp*).**

Marxan uses a simulated annealing algorithm to find good solutions to the mathematical problem:

minimize $\sum_{i=1}^{Ns} x_i c_i + b \sum_{i}^{Ns} \sum_{h}^{Ns} x_i * (1 - x_h) cv_{ih}$

subject to the constraint that all the representation targets are met

$$\sum_{i}^{Ns} x_i r_{ij} \geq T_j \quad \forall j$$

and x is either zero or 1

$x_i \in \{0,1\} \ \forall \ i$

where r_{ij} is the occurrence level of feature j in site i, c_i is the cost of site i, N_s is the number of sites, N_f is the number of features, and T_j is the target level for feature j. The control variable x_i has value 1 for sites selected for the reserve network and value 0 for sites not selected.

The first equation minimizes the penalties associated with the cost of the network and its configuration or shape. The parameter cv_{ih} reflects the cost of the connection, in this case simply the shared boundary, of planning units i and h. The parameter b, is the boundary length modifier (BLM), a user-defined variable that controls the importance of minimizing the total boundary length of the selected areas. For each scenario, I selected the BLM with the method described by Stewart & Possingham (2005), intended to achieve a level of connectivity between selected areas that does not unduly increase the overall cost of the solution. For each scenario I then ran Marxan with the simulated annealing schedule and 1,000 repeat runs. I measured the similarity of solutions from the four scenarios by spatial correlation (Spearman's) of selection frequency.

Results

Abundance models indicated that the greatest numbers of targeted fish were within the families Acanthuridae, Lutjanidae and Scaridae, which vary spatially in abundance across the qoliqoli (Figure 4.3). Acanthurids were the most abundant family for all reef types (Figure 4.3a). Acanthurids were most abundant on barrier forereefs and submerged patch reefs and occurred in medium densities on inshore fringing reefs (Figure 4.3a). Lutjanids were most abundant on barrier forereefs (Figure 4.3b). Scarids had high abundance on all types of submerged reefs, with highest abundance on forereefs and the fringing reef around Namenalala (Figure 4.3c). Abundance by market class also varied spatially (Figure 4.3). Species in class C were most abundant and widely distributed across all habitats in the qoliqoli, particularly on seaward facing slopes of patch reefs and forereef slopes of barrier reefs (Figure 4.3f). Class B fish were most abundant on outer barrier forereefs (Figure 4.3e), while class A fish were least abundant in these areas and most abundant on submerged inshore barrier and patch reefs (Figure 4.3d).

For all gear types combined, CPUE, as indicated by fisher survey records, was relatively uniform across all inshore reefs with the highest effort occurring on fringing reefs near villages (Figure 4.4a). Modelled total opportunity cost was highest for offshore barrier forereefs that are currently largely inaccessible to fishers without boats. Modelled total profit was highest for barrier forereefs that can be accessed with a communal raft known as a bilibili (within 3 km of the shore). Exposed fringing reefs had the lowest total values for opportunity cost and profit (Figure 4.4a). Opportunity cost and profit had similar magnitudes, with maximum values of $12 FJD per 2,500 m^2. Across areas with CPUE data, total modelled opportunity cost and profit were positively correlated with total CPUE (Table 4.3).

Table 4.3 Spearman's rank correlations (ρ) across currently fished reefs of catch per unit effort (CPUE) with modelled opportunity cost (opp) and profit (prof) for different gear types. Values correlated are for 250 m X 250 m grid cells (total area of 60 km2, n=952).

	$\rho_{opp,CPUE}$	$\rho_{prof,CPUE}$
Gill net	-0.311	0.481
Hand line	-0.024	0.197
Hand spear	-0.115	0.090
Hawaiian sling	-0.007	0.065
Speargun	0.285	-0.102
Trolling	0.049	-0.045
Total	0.158	0.160

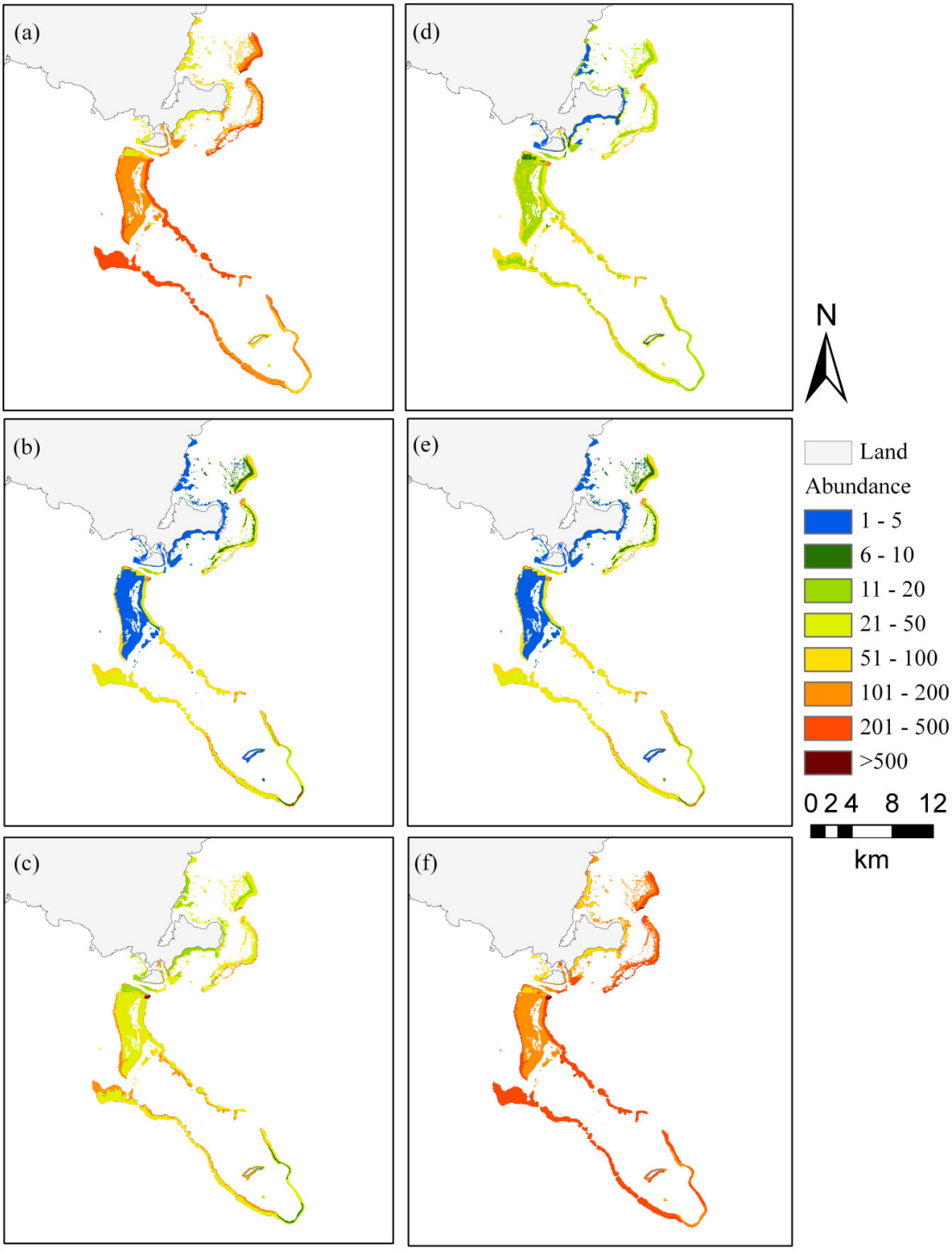


Figure 4.3 Modelled abundance per 2,500 m^2 (50 m X 50 m grid cell) for the three most abundant families and by market class in Kubulau District. (a) Acanthuridae. (b) Lutjanidae. (c) Scaridae. (d) Market class A; families included are Lethrinidae, Serranidae and Siganidae. (e) Market class B; families included are Carangidae, Haemulidae and Lutjanidae. (f) Market class C; families included are Acanthuridae, Balistidae and Scaridae.

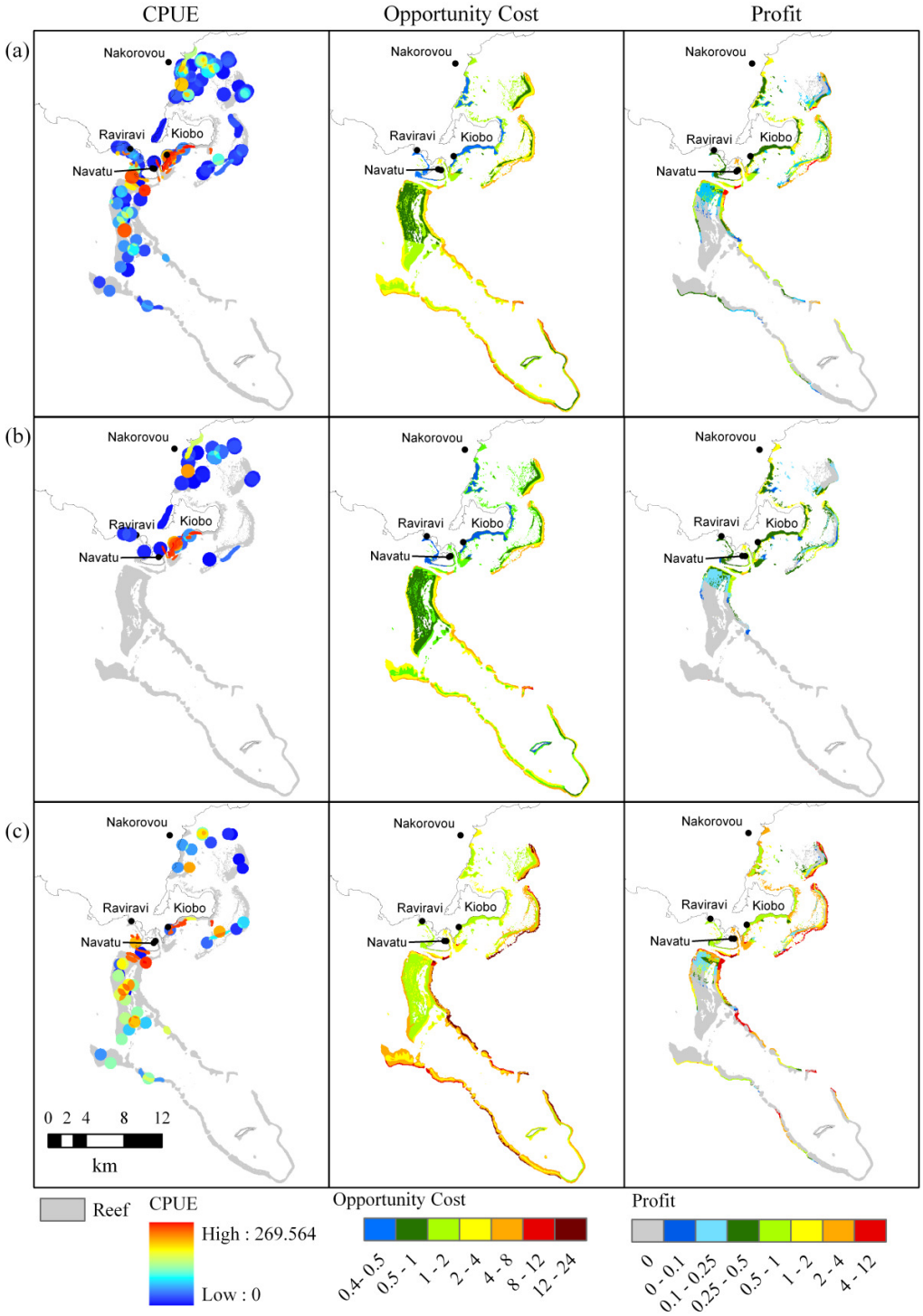


Figure 4.4 Catch per unit effort (CPUE), opportunity cost and profit in Kubulau District. First column shows total CPUE, calculated as catch person^{-1} hour^{-1} m^{-2}. Second column shows modelled opportunity cost in Fiji Dollars (FJD) per 2,500 m^2 (50 m X 50 m grid cell). Third column shows modelled profit in Fiji Dollars (FJD) per 2,500 m^2 (50 m X 50 m grid cell). (a) Total across all gear types. (b) Gill net. (c) Speargun.

CPUE values were spatially dissimilar between gear types, with negative correlations between most gear types (Table 4.4a). CPUE from gill nets and spearguns were the least similar (ρ = -0.256). Total CPUE was positively correlated with all gear types except for Hawaiian sling and trolling. Total CPUE had the largest positive correlation with catch by speargun (ρ = 0.601). Opportunity cost and profit were spatially very similar between gear types, with much larger coefficients than for CPUE (Table 4.4b,c). The relative strengths of correlations between gear types for opportunity cost and profit were, however, similar to those for CPUE (Table 4.4b,c). For both opportunity cost and profit, speargun and gillnet had the largest positive correlations with total values.

Modelled opportunity cost and profit had similar distributions, but varied in magnitude and distribution by gear type (Figure 4.4). Speargun users had the highest opportunity costs and profit of all fishers with a maximum value of \$24 FJD/2,500 m^2 (Figure 4.4c) followed by gill net users (Figure 4.4b). Speargun users had positive profit across more offshore reefs than other gear types, which predominantly had zero profit for offshore reefs (Figure 4.4). Modelled opportunity cost had mixed spatial correlations with CPUE by gear type while profit by gear type was predominantly positively correlated with CPUE (Table 4.3). Modelled opportunity cost was negatively correlated with CPUE for gear types that are used primarily for nearshore fisheries (e.g. gill net, ρ = -0.311). Opportunity cost was positively correlated with CPUE for gear types that use offshore reefs (e.g. speargun, ρ = 0.285). The largest positive correlation between profit and CPUE was for gill nets (ρ = 0.481). The largest negative correlation between profit and CPUE was for speargun (ρ = -0.102; Table 4.3).

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Table 4.4 Spearman's rank correlations (ρ) between gear types of catch per unit effort (CPUE), opportunity cost and profit (total and by gear type) across the 60 km^2 of currently fished reefs. Values correlated are for 250 m X 250 m grid cells (total area of 60 km^2, n=952). (a) CPUE from interviews; shading indicates non-significant relationship; for others, * $p < 0.05$, ** $p < 0.005$, *** $p < 0.001$. (b) modelled opportunity costs. (c) modelled profit.

(a)	**Gill net**	**Hand line**	**Hand spear**	**Hawaiian sling**	**Speargun**	**Trolling**
Hand line	0.007					
Hand spear	0.266***	0.198***				
Hawaiian sling	-0.086**	-0.136***	0.019			
Speargun	-0.256***	-0.186***	-0.046	-0.024		
Trolling	0.024	0.056*	0.048	-0.069*	-0.172***	
Total	0.192***	0.184***	0.216***	-0.019	0.601***	-0.212***

(b)	**Gill net**	**Hand line**	**Hand spear**	**Hawaiian sling**	**Speargun**	**Trolling**
Hand line	0.957					
Hand spear	0.993	0.950				
Hawaiian	0.975	0.934	0.987			
Speargun	0.993	0.954	0.990	0.981		
Trolling	0.983	0.930	0.981	0.985	0.985	
Total	0.997	0.961	0.993	0.980	0.998	0.985

(c)	**Gill net**	**Hand line**	**Hand spear**	**Hawaiian sling**	**Speargun**	**Trolling**
Hand line	0.917					
Hand spear	0.938	0.840				
Hawaiian sling	0.901	0.865	0.975			
Speargun	0.843	0.838	0.723	0.718		
Trolling	0.905	0.868	0.975	0.997	0.772	
Total	0.889	0.867	0.776	0.762	0.994	0.766

Modelled opportunity cost had mixed spatial correlations with CPUE by gear type while profit by gear type was predominantly positively correlated with CPUE (Table 4.3). Modelled opportunity cost was negatively correlated with CPUE for gear types that are used primarily for nearshore fisheries (e.g. gill net, $\rho = -0.311$). Opportunity cost was positively correlated with CPUE for gear types that use offshore reefs (e.g. speargun, $\rho = 0.285$). The largest positive correlation between profit and CPUE was for gill nets ($\rho = 0.481$). The largest negative correlation between profit and CPUE was for speargun ($\rho = -0.102$; Table 4.3).

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The current network of tabu and MPAs in Kubulau qoliqoli cover 40% of all barrier reefs, 36% of fringing reefs, and 2% of patch reefs. Selection frequencies for the clean-slate scenarios with CPUE and opportunity costs were strongly and positively correlated ($\rho_{1,3} = 0.880$, $p < 0.001$). Selection frequencies for the locked-in scenarios with CPUE and opportunity costs were also strongly and positively correlated (Spearman's rank correlation coefficient $\rho_{2,4} = 0.867$, $p < 0.001$). Correlations were weaker but still significant between selection frequencies for the clean-slate and locked-in scenarios for CPUE ($\rho_{1,2} = 0.320$, $p < 0.001$) and opportunity cost ($\rho_{3,4} = 0.082$, $p < 0.001$). All scenarios selected portions of Cakaunivauaka reefs to meet the patch reef target of 30% (Figure 4.5).

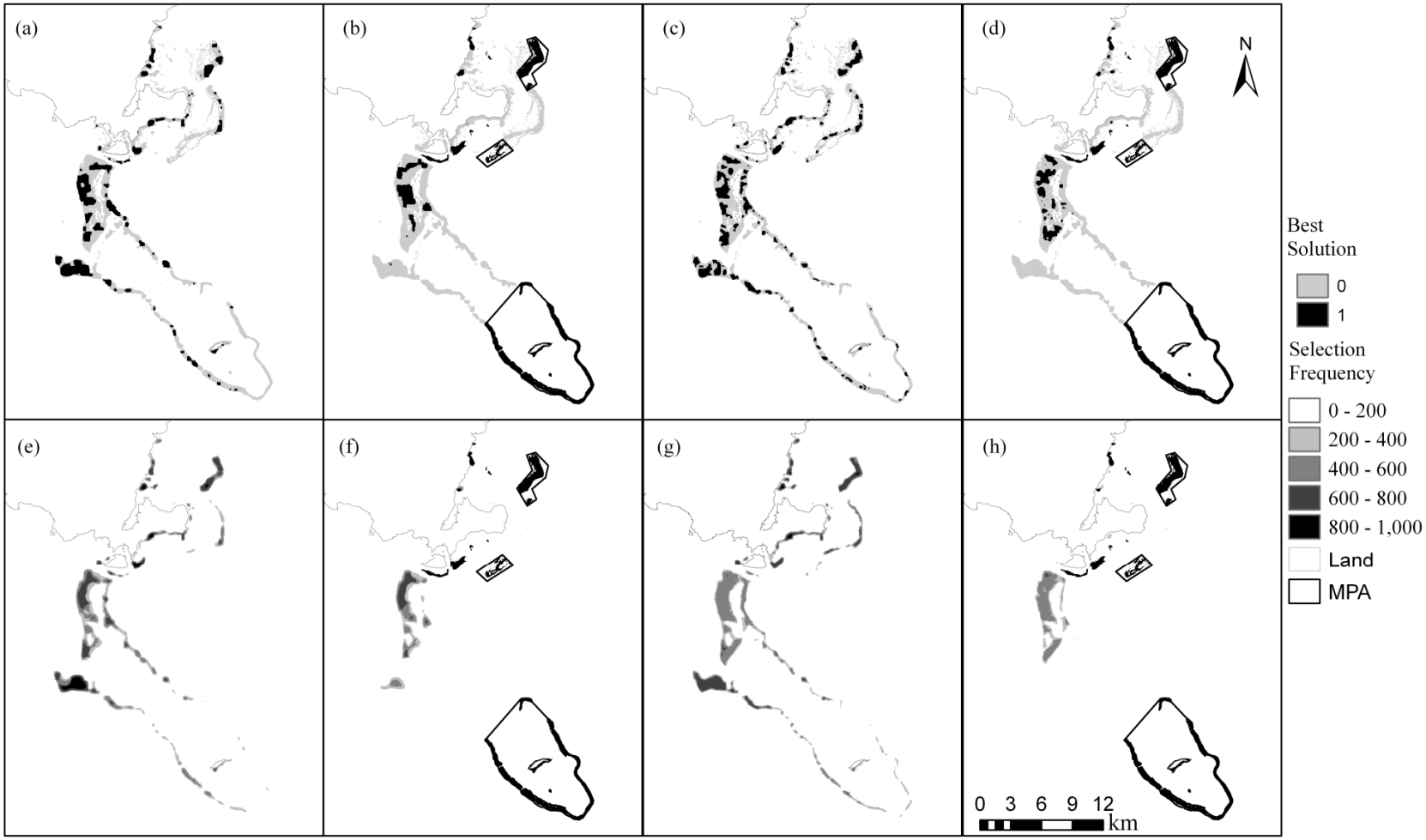


Figure 4.5 Marxan best solutions (top row) and selection frequencies (bottom row) for the four scenarios for Kubulau District. (a) Best solution for scenario 1, clean slate catch per unit effort (CPUE). (b) Best solution for scenario 2, locked in CPUE. (c) Best solution for scenario 3, clean slate opportunity cost (Opp). (d) Best solution for scenario 4, locked in Opp. (e) Selection frequency for scenario 1, clean slate CPUE. (f) Selection frequency for scenario 2, locked in CPUE. (g) Selection frequency for scenario 3, clean slate Opp. (h) Selection frequency for scenario 4, locked in Opp.

For the locked-in scenarios, the barrier reef and fringing reef targets were predominantly met by the current tabu areas and MPAs, so nearly all additional areas selected were patch reefs within Cakaunivuaka Reef (Figures 4.5b,d). Best solutions from the clean-slate analyses also selected areas of Cakaunivuaka reefs while indicating that Namuri MPA plus some of the inshore community tabu areas could be replaced by adding protection to Nakadamalevu reef (Figures 4.5a,c). Despite the strong overall correlation, there were some notable spatial differences in selection frequencies for the clean-slate scenarios using CPUE versus opportunity cost. Selection frequencies were higher in the southern portion of Namena barrier reef for the opportunity cost scenario but higher on the fringing reef near Navatu for the CPUE scenario (Figures 4.5e,g).

There were two expected general results from analysis of total costs of scenarios (Figure 4.6). First, CPUE and opportunity cost were each minimized when directly used in the Marxan analyses. Second, for both CPUE and opportunity cost, clean-slate solutions were less costly than locked-in solutions. Specifically, the opportunity cost scenarios resulted in best solutions that reduced opportunity costs and profit by approximately 12% compared to CPUE scenarios (Figure 4.6). However, using opportunity cost resulted in selecting MPAs with substantially higher CPUE (5-20% of total CPUE, Figure 4.6). CPUE and opportunity cost produced comparable results in terms of total area selected.

The spatial location of effort across Kubulau qoliqoli is primarily driven by access to transport. The majority of fishers from Kubulau do not have access to motorised boats and fish largely for subsistence (Cakacaka et al. 2010). Therefore, current effort across all gear types is highest on reefs closest to villages (inshore fringing and patch reefs). The same trend of heavy effort on nearshore reefs has been observed in other regions where access to motorized transport is limited (Aswani 1998; Bene & Tewfik 2001). Therefore, my profit model best reflected current fishing effort because, unlike the opportunity cost model, it considered accessibility and travel costs.

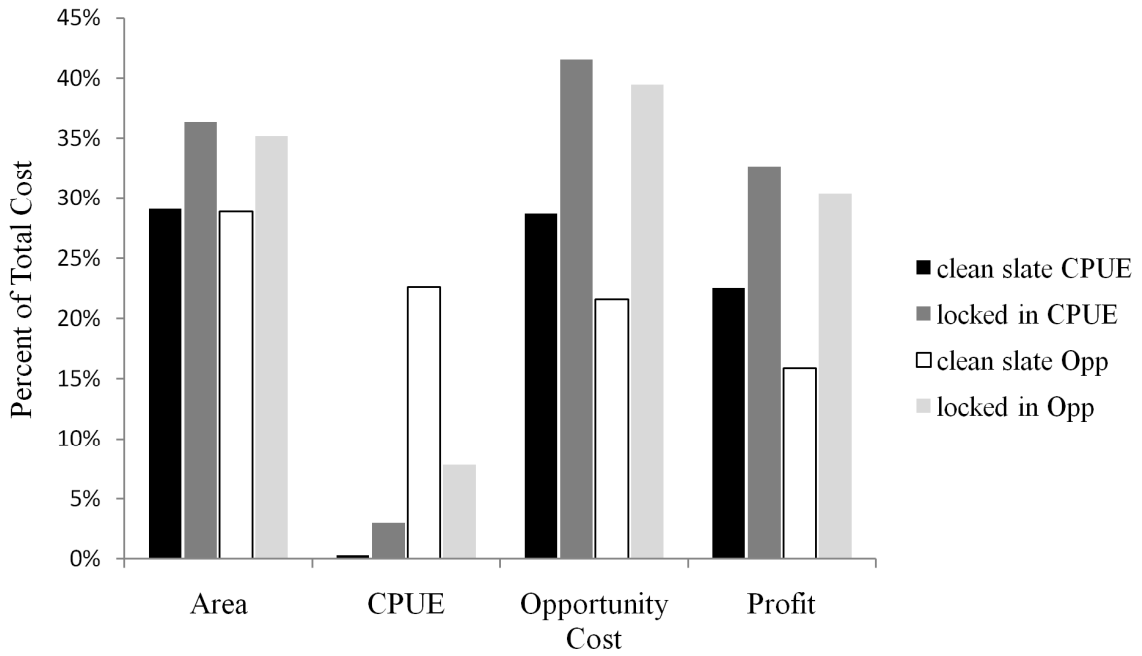


Figure 4.6 Total cost to fishers in Kubulau District calculated for Marxan best solutions for the four scenarios, displayed as percentage of total cost for four cost metrics: area, catch per unit effort (CPUE), opportunity cost (Opp), and profit. I determined % total cost for all metrics by dividing the summed cost of the areas selected for protection by the total cost across the study area. I included area as a cost metric to demonstrate that summed areas selected in Marxan runs were comparable between cost scenarios, even though other metrics differed strongly.

By contrast, the quantity of catch is largely driven by access to markets and more efficient gear, such as spearguns and gill nets, owned by several fishers mostly from Navatu and Kiobo operating on an artisanal scale. Highest profit values for gill nets are on the fringing reefs where they are typically used to target schooling fish, particularly mullets (Rawlinson et al. 1995). Meanwhile, high profits for spearguns are on the offshore barrier reefs, where they are a preferred gear type due to their ability to efficiently target larger, more valuable fish (Dalzell 1996; Kuster et al. 2005; McClanahan & Mangi 2000). Because spearguns and gill nets require considerable initial investment (approximately $300FJD for a speargun and $200FJD for a gill net), it is unlikely that fishers are willing to make the financial outlay unless they can guarantee revenue from catch (Veitayaki 1990).

Fishers from Navatu, Namalata and Kiobo have reliable access to a middle man who lives in Navatu village and regularly sells fish to the markets in Savusavu (Clarke & Jupiter 2010). Therefore, they have the financial incentive to invest in motorboats which enables them to access distant, high-density fishing grounds with highly efficient gear to maximize catch. Increased access to markets could drive purchases of spearguns and motorized transport,

causing higher fishing effort on the offshore reefs causing the profit model to converge on the opportunity cost model. This type of market-driven fishing displacement has been observed in other multispecies fisheries, where the high value of catch has strongly influenced the location of fishing effort, particularly in situations with rising fuel prices and depletion of commercial stocks (Arrelano & Swartzman 2010).

In fact, it is likely that the number of catch locations from artisanal fishers on outer barrier reefs is under-represented in CPUE surveys. Two likely reasons are a fear of releasing information on choice productive fishing grounds and awareness that some of these fishing locations violate community management rules (e.g. are within MPA boundaries). For example, within the past year, fishers from Navatu have been repeatedly caught fishing within the Namena Marine Reserve (Clarke & Jupiter 2010), although these catch locations were never recorded on CPUE forms. In such cases where recorded fishing effort might not accurately match actual or future effort, the opportunity cost model provides an unbiased alternative. It allows conservation planners to value reefs that are currently reported as unfished and, when designing MPAs, to minimize displacement of fishers not included in surveys or who misreported their fishing grounds.

Marxan results and implications for reconfiguration of MPAs

There were notable similarities between the Marxan scenarios that used CPUE and opportunity cost. Both locked-in approaches indicated that high-priority additions to the MPA network were areas within Cakaunivuaka Reef. Both clean-slate approaches were spatially similar and indicated other potential areas that could be added to the network, such as those within Nakadamulevu Reef. However, the CPUE clean-slate approach identified the fringing reef near Navatu for protection because there is currently a tabu area on this reef and therefore no effort (Figure 4.5a). In contrast, the clean-slate approach using opportunity cost left this reef open to fishing because of its high cost (Figure 4.5c). The opportunity cost scenarios had lower total costs when considering opportunity cost and profit, but considerably higher costs when considering CPUE (Figure 4.6). This indicates that using opportunity cost in Marxan reduces the impacts to both current and future fishing effort. However the communities should be consulted regarding the Marxan scenarios to determine whether the opportunity cost models produced amenable MPA selections. Consultation with the communities will allow the stakeholders to determine the relative importance of maintaining current fishing grounds over future fishing grounds.

Given that recent monitoring from the Namuri MPA suggests that it is being substantially affected by poaching (Jupiter et al. 2010), and outputs from both clean-slate scenarios do not select sites within Namuri, it would be worthwhile to suggest a trade-off to the Kubulau community. This would open portions of Namuri to fishing in exchange for protection of areas within Nakadamulevu and the adjacent Nakadamulailai reefs. Closure of these areas could potentially also offset opening a portion of Namena Marine Reserve, which has been heavily contested by one of the Navatu clans, members of which have been repeatedly caught fishing in Namena.

Because the clan perceives inequity in the distribution of costs and benefits of the present tabu areas and MPAs, the Marxan solutions could reduce conflict by producing more socially acceptable configurations (Lal 2005). Some of this conflict might have arisen because the environmental and social goals of MPA establishment and management were ill-defined from the outset. Having a clear understanding of these goals is critical to ensure that MPA design does not adversely affect current fishing industries and community identity (Ban et al. 2009; Klein et al. 2008a).

Model applications and conclusions

The opportunity cost model provides data for use in decision support tools for conservation planning such as Marxan and Marxan with Zones (Ball et al. 2009; Watts et al. 2009). My modelling approach incorporates socio-economic considerations and can be applied in regions with poor data on human uses and those where people are highly dependent on natural resources to ensure that conservation actions minimize impacts on local communities (Ban et al. 2009; Ban & Klein 2009). Standard socio-economic survey methods often have limitations. In marine applications, these include lack of participation by users of all gear types, partial disclosure of fishing areas, and lack of standardized methods for integrating spatial data (for methods on standardizing data collection with GIS see De Freitas & Tagliani 2009). My model differs from standard survey approaches by accounting for variation in the spatial distribution of natural resources rather than focusing on current extractive effort.

For marine regions, my model is appropriate for considering socio-economic goals in fisheries where gear preferences or access to transport are expected to change dramatically (Salas & Gaertner 2004). In these regions, there may be little to no existing biological and

socio-economic data. While my model requires multiple modelling steps, it can be adapted for data-poor regions. The main data restriction for my model is existing fish count surveys which may not exist in some regions. However, most regions in which NGOs are active will have fish abundance or presence/absence data would be suitable for the approach described here. The remaining data for predicting fish abundance could be acquired strictly from remote sensing and basic navigational charts (such as depth and reef classification). The socio-economic inputs such as percentage catch can be estimated based on several survey days in villages. Due to data quality issues and multiple model inputs, error in the model can be tested using standard techniques such as info-gap and sensitivity analysis (Halpern et al. 2006). Sensitivity analysis might be particularly useful if particular inputs in the model have higher associated error. For example, in my model the percentage catch models have high uncertainty associated with them due to small sample size. Therefore, a sensitivity analysis could demonstrate how modelled opportunity cost and profit vary with increments of error in estimates of percentage catch by gear type. This analysis could be incorporated into Marxan scenarios to investigate how sensitive area selections are to variability in the opportunity cost model.

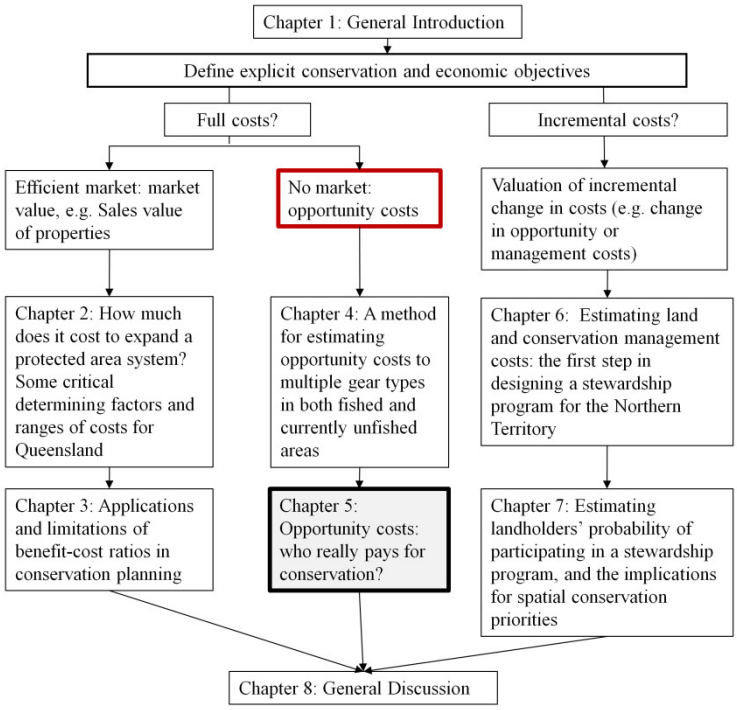
Although Marxan is a static planning tool, recent applications have implemented Marxan in dynamic multi-year simulations (Visconti et al. 2010). Fishing effort and fish abundance are likely to interact dynamically through time and a multi-year dynamic approach to MPA design may be more realistic (e.g. see Christensen et al. 2009). My models can be adapted to incorporate fisheries dynamics for multi-year simulations. For example, my profit and opportunity cost models can be used to examine how costs will change as market access drives changes in access to gear type and transport. Changes in gear type can be modelled with opportunity cost by exploring a range of weightings to reflect a larger proportion of fishers using efficient gear types such as spearguns. The effects of changes in transport on spatial effort can be explored in the profit model by altering weightings to reflect a larger proportion of fishers using motor boats, as has been observed across Fiji (Kuster et al. 2005).

Despite the acknowledged importance of socio-economic data in resource management, previous studies have suggested that artisanal fishers in developing countries are not always market-driven (Daw 2008; Pet-Soede et al. 2001), with fishing behaviour being determined more by factors such as values on time, risk aversion, and cultural identity (Bene & Tewfik 2001; Salas & Gaertner 2004). These factors can be considered explicitly by identifying them

in direct consultation with stakeholders and then treating them as additional input costs in the profit model. The expected costs from such scenarios can then be used in conservation planning software to provide a more thorough exploration of configuration options for MPAs and the range of impacts on local communities. The profit model therefore allows for a comprehensive analysis of tradeoffs between conservation actions and local economic development that is not possible with CPUE data.

Chapter 5 Opportunity costs: who really pays for conservation?[1]

Designing conservation areas entails costs that, if considered explicitly, can be minimized while still achieving conservation targets. Here I focus on opportunity costs which measure forgone benefits from alternative land uses. Conservation planning studies often use partial estimates of costs, but the extent to which these result in actual efficiencies has not been demonstrated. My study partitions land costs into three distinct opportunity costs to smallholder agriculture, soybean agriculture and ranching. I demonstrate that opportunity costs to single stakeholder groups can be inaccurate measures of true opportunity costs and can inadvertently shift conservation costs to affect groups of stakeholders disproportionately. Additionally, I examine how spatial correlations between costs as well as target size affect the performance of opportunity costs to single stakeholder groups as surrogate measures of true opportunity costs. I conclude that planning with opportunity costs to single stakeholder groups can result in cost burdens to other groups that could undermine the long-term success of conservation. Thus, an understanding of the spatial distributions of opportunity costs that are disaggregated to groups of stakeholders is necessary to make informed decisions about priority conservation areas.



[1] Adams, V. M., R. L. Pressey, and R. Naidoo. 2010. Opportunity costs: who really pays for conservation. Biological Conservation **143**:439–448.

Introduction

As discussed in the previous chapters, to achieve conservation goals, conservation planning must address social and economic, as well as biological, criteria (Carpenter et al. 2006). However, most research to date has neglected social and economic factors, despite agreement that they ultimately influence the success of conservation efforts (Polasky 2008). The most common approach to systematic conservation planning has been to minimize the total number or extent of selected areas, implicitly assuming that land costs are homogeneous (Carwardine et al. 2008). In fact, land costs are highly heterogeneous and studies that have explicitly incorporated land acquisition costs have demonstrated large cost savings in achieving conservation objectives (Ando et al. 1998; Ferraro 2003a; Naidoo & Iwamura 2007; Polasky et al. 2001). Other studies have shown that incorporating opportunity costs minimizes losses to commercial fishers or reduces their displacement (Klein et al. 2008a; Richardson et al. 2006; Stewart & Possingham 2005). While these studies have accounted for socio-economic costs by including various forms of available data, an important unresolved question for conservation planning is how the cost surrogates used in these studies account for the full costs of implementing conservation plans.

Economic costs of conservation planning include five main components: acquisition, management, damage, transaction, and opportunity (Naidoo et al. 2006). A full accounting of costs would include all five, but this is likely to be difficult or impossible in most planning exercises. This requires the use of surrogate measures of cost (e.g. opportunity costs to single stakeholder groups, Possingham et al. 2000; Stewart & Possingham 2005). Surrogate measures are substitutes that should ideally estimate actual costs, or at least have strong spatial correlations with actual costs, when the necessary data for complete accounting are unavailable. In this way, cost data are similar to data on biodiversity, the complexity of which precludes complete measurement and requires surrogate measures, all of which come with limitations that are still being investigated and debated (e.g. Pressey 2004; Rodrigues & Brooks 2007). Analogous investigations about the use of cost data in conservation planning have been scarce.

Studies using costs in conservation planning have included one or several surrogates, but not total measures of cost. Typically studies select the largest component of total costs as a surrogate measure. For example, the cost to a government of establishing a protected area is dominated by acquisition and long-term management costs. Because acquisition costs are

readily estimated by land market values they are typically used as the surrogate measure. Additionally, it has been estimated that acquiring land for protection is likely to exceed subsequent management costs by large factors (Balmford et al. 2003).

Aside from the immediate costs to conservation organizations, there are usually large social costs of conservation action (Balmford & Whitten 2003). To ensure that conservation assessments move to implementation of actions on the ground and in the water, it is insufficient to consider only the costs to conservation organizations. Rather, social assessments and explicit considerations of stakeholders directly involved in or affected by conservation actions, whether they are displaced landholders or government organizations, are needed (Knight et al. 2006; Pierce et al. 2005). Opportunity costs are an appropriate cost measure to explicitly account for the social costs of establishing protected areas and other conservation interventions. Opportunity costs of conservation are the costs associated with forgone opportunities to convert land to profitable uses. Investigation of opportunity costs to single stakeholder groups can demonstrate how conservation costs are distributed between different groups.

Some previous studies have examined the opportunity costs of proposed reserves or areas likely to be needed for conservation (Chomitz et al. 2005; Naidoo & Adamowicz 2006). Opportunity costs have also been used in cost-benefit analyses of alternative land uses in conservation planning (Kremen et al. 2000; Lauren et al. 2007; Sinden 2004; Wunder 2007). Yet, opportunity costs have only rarely been used in planning processes to propose new areas that reduce social conflict and the economic costs of conservation (see Klein et al. 2008a; Richardson et al. 2006; Stewart et al. 2003; Williams et al. 2003). Stewart et al. (2003) and Richardson et al. (2006) considered the opportunity costs to single stakeholder groups but did not investigate whether these represented total forgone opportunities. Williams et al. (2003) tried to include opportunity costs to multiple stakeholder groups with expert knowledge by determining an aggregate measure of conflict rather than partitioning costs to individual groups. Klein et al. (2008a) attempted a complete accounting of opportunity costs to stakeholders along California's central coast by estimating combined fishing effort by different users in each candidate conservation area. They did not, however, analyze the costs to individual stakeholder groups.

There have been many conservation planning studies that questioned how subsets of biodiversity (target) data affected planning outcomes (Cabeza & Moilanen 2001; Williams et al. 2006). Analogous studies have not, however, investigated how using different subsets of cost data (i.e. surrogate measures of cost such as opportunity costs to individual stakeholder groups) affects the results of planning. Studies examining different biodiversity surrogates have demonstrated dissimilarities in conservation outcomes depending on what biodiversity surrogates were selected (Williams et al. 2006). An important missing perspective in conservation planning with spatially variable costs is therefore the extent to which the use of partial estimates of opportunity costs – the costs to a single stakeholder group – adequately account for full opportunity costs. My study is designed to fill this gap.

Including partial or incomplete estimates of opportunity costs might help to reduce conflict and mitigate the cost of conservation action, but it is equally possible that it will unwittingly increase costs that have not been considered. The use of surrogate measures can have an unpredictable net outcome and will be highly dependent on the region being studied and the socio-economic factors considered. While I am not arguing that available cost data should be ignored if they are incomplete, I demonstrate the potential for data that are incomplete or misinterpreted to have negative net effects on the opportunity costs of conservation action. This potential needs to be recognized and accounted for. I address three main questions through the analyses in my case study.

My first question is: how do spatial correlations between costs, relative spatial variability of costs, and size of conservation targets affect the importance of selecting accurate cost surrogates? I address the issue that selecting an effective cost surrogate requires an understanding of all the opportunity costs in a region and their relative spatial distributions, as well as the influence of conservation objectives on the performance of single costs as surrogates for true opportunity costs. The understanding of the spatial relationships between different costs should inform the selection of a cost surrogate and might even preclude the use of a cost surrogate.

My second question is: does using a surrogate measure of opportunity costs – in my case, the costs to a single stakeholder group – serve as an adequate proxy in conservation planning for full opportunity costs and result in cost efficiencies relative to these full opportunity costs?

My third question is: how does including a surrogate measure of opportunity costs affect different stakeholders in a planning region, or in other words, who is really paying for conservation? This final question examines whether planning with partial costs can displace the selection of conservation areas in such a way as to disproportionately affect some stakeholders and thereby increase conflict between extractive use and protection of natural resources. This differs from my second question, in which I am only concerned with whether a cost surrogate can truly reflect full costs. Instead, my third question considers how using a cost surrogate can affect the multiple stakeholders present in a region and what the larger socio-economic implications of disproportionate conservation costs might be.

Methods

Planning region

I used the Mbaracayu Forest Biosphere Reserve (the "biosphere reserve") in Paraguay as a case study (Figure 5.1a). The biosphere reserve, in the Upper Jejui River Basin, consists of a core protected area of approximately 64,000 ha and a larger surrounding sustainable use area. The total area of the biosphere reserve is nearly 300,000 ha. The biosphere reserve has been identified as a high priority area for conservation interventions compared to other similar regions in Paraguay (Keel et al. 1993). It is within the highly threatened Upper Parana Atlantic Forest ecoregion and has experienced extensive loss and fragmentation of forest in the last 30 years. Additionally, the biosphere reserve contains upper Parana Atlantic forest and cerrado, vegetation types that are characteristic of two of WWF's Global 200 priority ecoregions (Olson & Dinerstein 2002). The cerrado and cerradon grasslands are known to be rich in species (Silva & Bates 2002). I chose the biosphere reserve as my study region because of the availability of high-resolution data on costs and vegetation types. The cost data are unusual in that they can be disaggregated into opportunity costs of conservation to three different stakeholder groups.

Land uses and conservation costs

Outside of the core protected area approximately 43% of the sustainable use area has been converted to human uses (Figure 5.1a). About 30,000 people inhabit the sustainable use area. Their main livelihood is smallholder agriculture, the expansion of which threatens the forested land surrounding the core protected area. The other two major land uses are cattle ranching and soybean farming. The three land uses represent three stakeholder groups ranging from local growers to multinational investors.

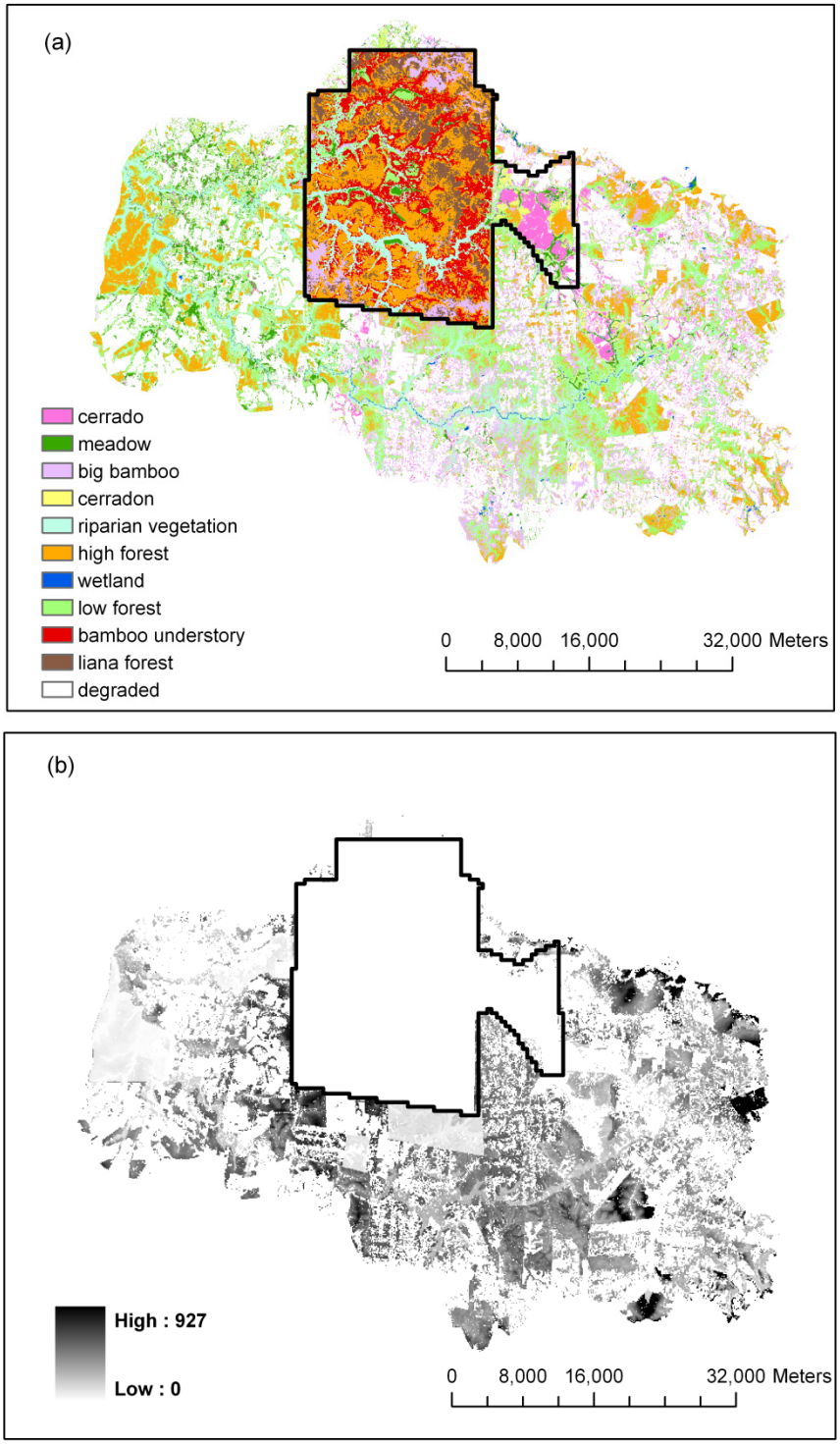


Figure 5.1 The Mbaracayu Forest Nature Reserve. (a) Vegetation types. I allocated a target of 40% of present extent to all vegetation types, except for degraded vegetation, cropland and pasture. The core protected area is outlined in black. (b) Summed opportunity costs (USD per hectare) at a discount rate of 20% for the Mbaracayu Forest Nature Reserve. White areas have estimated opportunity costs of $0 or no estimate. Estimates of $0 refer to existing reserves or unreserved areas with very low probabilities of conversion to agriculture. Areas with no estimates are already degraded and of no immediate interest for conservation.

In a previous study the opportunity costs of conservation were modelled for the biosphere reserve (Naidoo & Adamowicz 2006). The data and methods from Naidoo and Adamowicz (2006) were used to estimate costs for this study. Because the future land use of any parcel is unknown, I estimated the discounted agricultural benefits as the probability of a parcel of land *i*, being in a particular agricultural state *k* in a particular time step *t*, multiplied by the financial benefits of state *k*, R_k, summed over all potential agricultural states *K*. With the assumptions that the probability of conversion of a parcel is constant over time and the returns associated with agriculture are spatially invariant, then, as *t* goes to infinity, the expected value, or opportunity cost, becomes:

$$EV = \frac{\sum_{k=1}^{K} \sum_{i=1}^{I} P_{ik} R_k}{\partial}$$

The use of probabilities of conversion captured the spatial variations in suitability of land for different uses and therefore captured the spatial variations in opportunity costs. The method used to estimate opportunity costs used information on historic land conversion and factors that influence conversion patterns (Naidoo & Adamowicz 2006). Because the opportunity costs were based on historic trends in land uses, I did not project altered probabilities of conversion into the future but rather assumed them to be fixed over time. For conservation actions occurring over a short time frame this is probably a reasonable assumption.

I used the previously calculated opportunity costs of conservation for each hectare of forest (Naidoo & Adamowicz 2006). Areas already converted to one of these uses were not allocated opportunity costs. These areas were not suitable for conservation action because of their conversion and therefore had no opportunity costs arising from establishment of protected areas. Naidoo and Adamowicz (2006) had previously validated estimates of opportunity cost against 20 property values within the biosphere reserve. There was a strong correlation between estimated and actual costs, particularly for discount rates between 15 and 25%. While actual discount rates for land market prices are not readily available in this region, I considered 20% to be a reasonable rate because of the correspondingly strong correlation between estimated and actual costs (Naidoo & Adamowicz 2006). The opportunity cost ranged from 0 – 972 USD per ha (Figure 5.1b).

To approximate surrogate measures of cost used when incomplete data are available (Stewart et al. 2003; Stewart & Possingham 2005) I used the probability of conversion to land use *k* and the return associated with agricultural land use *k* (R_{ik}) to calculate the opportunity costs of conservation to each stakeholder group, i.e. those involved in smallholder agriculture, cattle ranching and soybean farming. Thus, for land use *k* the associated opportunity cost is:

$$EV = \sum_{i=1}^{I} P_{ik} \frac{R_k}{\partial}$$

To understand the spatial patterns of land uses by the three stakeholder groups I calculated the spatial correlation (Pearson's) among all three opportunity costs, and between each cost and "complementary vegetation". I measured complementary vegetation as the number of hectares of vegetation per planning unit (candidate conservation area, below) that contributed to unmet targets in the planning region, divided by total area of the planning unit (Margules & Pressey 2000). By unmet targets, I mean those not already achieved in the core protected area. Additionally, I calculated the relative spatial variability of complementary vegetation and each of the opportunity costs. As the data showed evidence of positive spatial autocorrelation I only considered the correlation coefficients, as these are unaffected by autocorrelation, and do not report the significance values (Balmford et al. 2001; Nhancale & Smith 2011).

Biodiversity targets

The core protected area in the biosphere reserve has slowed deforestation significantly compared to the adjacent forest. In the past decade, the buffer surrounding the core has experienced 39.25% forest loss compared to 2.13% within the core (Huang et al. 2007). Deforestation outside the core poses a large threat to biodiversity throughout the unprotected portion of the biosphere reserve and has motivated management strategies such as sustainable land use projects and the design of corridors to connect remaining fragments. Additional protected areas, especially private reserves, are being implemented to complement the existing core area. In this paper I explore the costs of this latter approach. I recognized thirteen distinct vegetation classes (Figure 5.1a). Except for degraded vegetation, cropland and pasture, which I did not target for conservation, I set conservation targets for vegetation types of 40% of their present extent. This is an indicative figure that is plausible based on pressures from human populations and uses of natural resources within the biosphere reserve.

For my analysis on the effects of targets on spatial similarity of selected, potential reserves I varied targets from 0% to 100% in 10% increments.

Reservation scenarios

I divided the planning region into 11,675 planning units, each forming a 500 x 500 m cell (25 ha). I considered the core protected area to be committed to conservation and locked it into my analyses. The core consisted of 2,693 planning units and represented at least 10% of all vegetation types.

I examined five conservation scenarios, each with a different measure of cost. This allowed me to test how considering partial costs affected the opportunity cost of implementing a system of reserves additional to the core protected area. It also allowed me to test the extent to which partial costs for individual land uses deflected the selection of reserves to areas that had high costs for other land uses. I calculated the cost *c* of each planning unit *i* as the mean opportunity cost per hectare, including forested land with zero cost per hectare but excluding converted land with no opportunity cost.

Scenario 1("Base"): The cost of an individual planning unit, c_i, was its area. Because I used a uniform 25 ha grid for my planning units, this means that all planning units have equal costs, reflecting the common assumption in conservation planning that costs are homogeneous across landscapes.

Scenario 2 ("Smallholder"): The cost of an individual planning unit, c_i, was its opportunity cost to smallholder agriculture.

Scenario 3 ("Ranch"): The cost of an individual planning unit, c_i, was its opportunity cost to cattle ranching.

Scenario 4 ("Soybean"): The cost of an individual planning unit, c_i, was its opportunity cost to soybean farming.

Scenario 5 ("Full"): The cost of an individual planning unit, c_i, was the sum of its opportunity costs for all land uses, which is a proxy for acquisition cost. I use the term 'full' to indicate the case in which a full accounting of land uses is known and therefore the true opportunity costs of conservation can be calculated.

Reserve design

I used Marxan software (Ball et al. 2009) to design cost-effective reserves that met the conservation targets for all vegetation types (see Chapter 4 for full Marxan method details). I identified, for each scenario, the BLM that minimizes the trade-off between boundary length and cost (termed "optimal" here) using the method described by Stewart & Possingham (2005). For each of my five reservation scenarios and using optimal BLMs, I ran Marxan with the simulated annealing schedule and 1,000 repeat runs.

Analysis of reserve design solutions

I compared the results of the five scenarios by recording, from the best solution for each (the solution with the smallest objective function over 1000 runs), the total area, total boundary length, and total of each of the five cost variables.

I identified priority areas under each scenario by testing the selection frequency (McDonnell et al. 2002) of each planning unit (analogous to its irreplaceability, Ferrier et al. 2000) against the probability that it was selected from a random sample. To do this I assumed that every planning unit had an equal probability of being selected and calculated the probability of selection p (Stewart et al. 2003) as:

$$p = (C - R)/(T - R)$$

where C is the average number of planning units selected across multiple runs, R is the number of units locked in or out (in this case the 2,693 units in the core protected area), and T is the total number of planning units (11,675). The frequency distribution of selection of planning units is binomial if selection is random. I classified planning units as irreplaceable if they were selected more frequently than 95% of random selections.

I compared the spatial similarity of solutions for the five scenarios with two methods. First, I compared the single best solutions from pairs of scenarios using the kappa statistic: an index that measures observed spatial agreement compared to that expected by chance. Kappa values range from -1 (complete disagreement) to +1 (complete agreement). Second, I compared pairs of scenarios with the Spearman rank correlation (ρ) of selection frequency of planning units. For both analyses I excluded the existing core protected area and included only additional areas selected in the Marxan runs.

Effects of target percentage on spatial similarity

I expected that my choice of conservation target would influence the spatial similarity of my scenarios. For small targets, solutions were likely to converge spatially because the core protected area, locked into all scenarios, contained between 10% and 100% of each vegetation type, requiring relatively few additional planning units. Spatial convergence was also likely for large targets because most planning units outside the core protected area would be selected. For intermediate targets, I expected that spatial convergence of scenarios would be relatively small. I examined how different targets affected the spatial similarity of scenario 1 (minimizing area) and scenario 5 (minimizing full opportunity cost). I ran Marxan with the simulated annealing schedule and 100 repeat runs for targets increasing from 10% to 100% in increments of 10%. To understand how clumping reserves with the BLM affected spatial similarity, I repeated the analyses using the optimal BLM from previous analyses and with BLM = 0. For each target percentage and each BLM setting, I then compared the spatial similarity of the two scenarios by correlating the selection frequencies of planning units.

Results

Land use patterns of the three stakeholder groups in the region were dissimilar and highly variable spatially. Across all planning units, opportunity costs to smallholder agriculture were negatively correlated with both opportunity costs to soybean farming and ranching (Table 5.1). Soybean farming was most strongly correlated with full opportunity costs (r=0.79) and thus should be the most accurate surrogate measure. Full opportunity cost and complementary vegetation were uncorrelated (r=0.01) and full opportunity costs were slightly more variable spatially than complementary vegetation ($c_{comp\ \mathrm{veg}} = 63.75$, $c_{full\ cost} = 87.64$).

Table 5.1 Pearson's correlation coefficients (r) and relative variability (coefficient of variation, *c*) of complementary vegetation and opportunity costs to single stakeholder groups.

	Smallholder cost	**Ranching cost**	**Soybean cost**	**Full cost**	**Complementary vegetation**
Correlations (r)					
Ranching opportunity cost	-0.27				
Soybean opportunity cost	-0.11	0.08			
Full opportunity cost	0.28	0.44	0.79		
Complementary vegetation	0.06	0.16	-0.11	0.01	
Variability (coefficient of variation, *c*)	122.22	105.20	360.77	87.64	63.75

Table 5.2 Comparison of solutions from each scenario. (a) Parameters of the best solutions for different scenarios. Cost refers to the units used for each scenario. Scenario 1 is in ha whereas scenarios 2-5 are in terms of the opportunity costs minimized (USD). (b) Kappa statistics (κ) for the spatial agreement between the best solutions under each cost scenario ($p<0.001$ for all). (c) Spearman's rank correlation (ρ) of selection frequency of planning units under each cost scenario ($p<0.001$ for all).

(a)

Scenario	Optimal BLM	Cost	Planning units	Area (ha)	Boundary length (km)
1 – base	0.1	113,725	4,549	113,725	66,200
2 – smallholder	0.01	167,071	5,327	133,175	58,100
3 – ranching	0.05	779,495	5,124	128,100	60,700
4 – soybean	0.001	17,542	5,256	131,400	51,750
5 - full	0.2	2,789,451	4,809	120,225	73,400

(b)

	1 - base	**2 – smallholder**	**3- ranching**	**4 -soybean**
2 – smallholder	0.07			
3 - ranching	0.16	-0.16		
4 - soybean	0.16	-0.08	0.46	
5 - full	0.24	0.03	0.26	0.41

(c)

	1 - base	**2 – smallholder**	**3- ranching**	**4 -soybean**
2 – smallholder	0.21			
3 - ranching	0.23	-0.27		
4 - soybean	0.30	-0.06	0.33	
5 - full	0.55	0.13	0.29	0.57

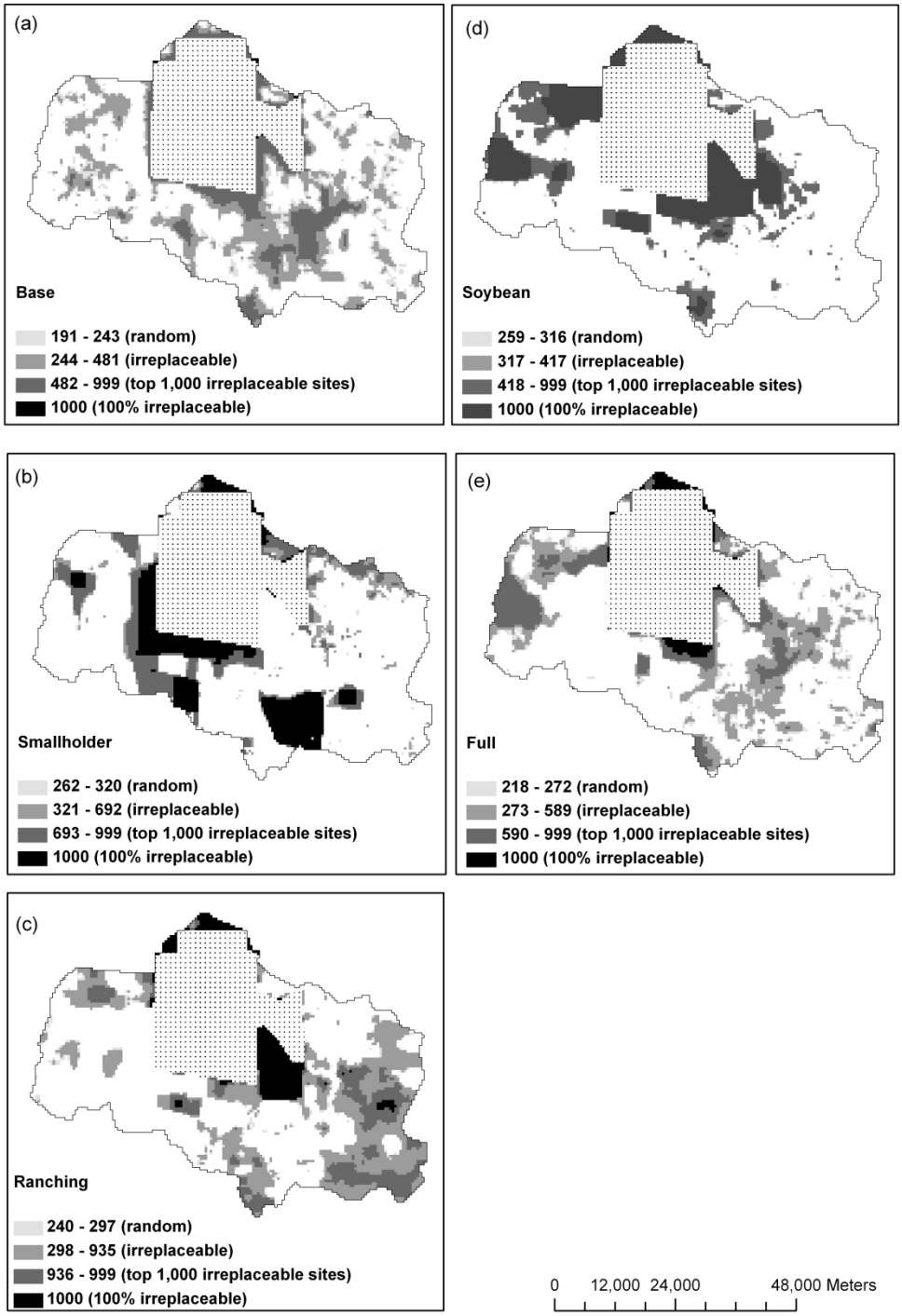


Figure 5.2 Irreplaceability of planning units with different opportunity costs. Planning units with high irreplaceability values are dark grey and black. The core protected area is hatched. Planning units selected less frequently than random are white. The mean number of planning units reported under each scenario is C. The probability that a planning unit will be selected by random is p, with a 95% confidence interval (CI) (Lower confidence limit, Upper confidence limit). (a) Irreplaceability for scenario 1 (base, homogenous costs), C = 4,642, p = 0.217, 95% CI (191, 243). (b) Irreplaceability for scenario 2 (smallholder), C = 5,306, p = 0.291, 95% CI (262, 320). (c) Irreplaceability for scenario 3 (ranch), C = 5,108, p = 0.269, 95% CI (240, 297). (d) Irreplaceability for scenario 4 (soybean), C = 5,276, p = 0.288, 95% CI (259, 316). (e) Irreplaceability for scenario 5 (full), C = 4,891, p = 0.245, 95% CI (218, 272).

The areas and boundary lengths of reserve systems were similar under all scenarios (Table 5.2a). However, the spatial pattern of reserves selected under each scenario varied greatly (Figure 5.2). The kappa statistics (Table 5.2b) indicated that scenario 5 (full opportunity costs) was most similar to scenario 4 (soybean farming) (κ=0.41) and least similar to scenario 2 (smallholder agriculture) (κ=0.03). Correlations (Table 5.2c) showed that the selection frequencies of planning units from scenario 5 (full opportunity costs) were most similar to those in scenario 4 (soybean farming) (ρ=0.57) and scenario 1 (no cost data) (ρ=0.55). The kappa statistics and correlations showed the same general patterns: opportunity cost to soybean farming (scenario 4) was the best surrogate measure of full opportunity costs (scenario 5), followed by area (scenario 1). These analyses also indicated some spatially dissimilar reserve designs. In particular, smallholder agriculture had a negative kappa statistic and Spearman's coefficient with both ranching and soybean farming (Table 5.2b,c).

The five scenarios had very different partial and full opportunity costs (Figure 5.3). Including full opportunity costs in the reserve selection algorithm (scenario 5) resulted in the lowest opportunity cost of conservation. Scenario 1 (base, homogeneous costs) and scenario 4 (soybean) provided the second lowest, and almost identical, opportunity costs. Each scenario involving partial opportunity costs (scenarios 2-4) resulted in the lowest cost for that land use, but increased full opportunity costs compared to scenario 5 and increased costs to at least one other stakeholder group relative to scenario 5. For example, compared to scenario 5, scenario 2 (smallholder agriculture) reduced costs for smallholder agriculture by 80%, but increased opportunity costs to soybean farming by 700%, opportunity costs for ranching by 200%, and full opportunity costs by almost 100%.

The spatial correlations in selection frequency between scenario 1 and scenario 5 depended on both targets and values of BLM (Figure 5.4). When BLM was zero, as expected, systems of selected reserves were most similar between scenarios with costs ignored (scenario 1) and included explicitly (scenario 5) when targets were low and high. Including full opportunity costs made most difference to the spatial configuration of reserves for targets around 50%. When I used optimal BLMs for each scenario, the correlations decreased linearly from 10%, then increased rapidly from 90% to 100%.

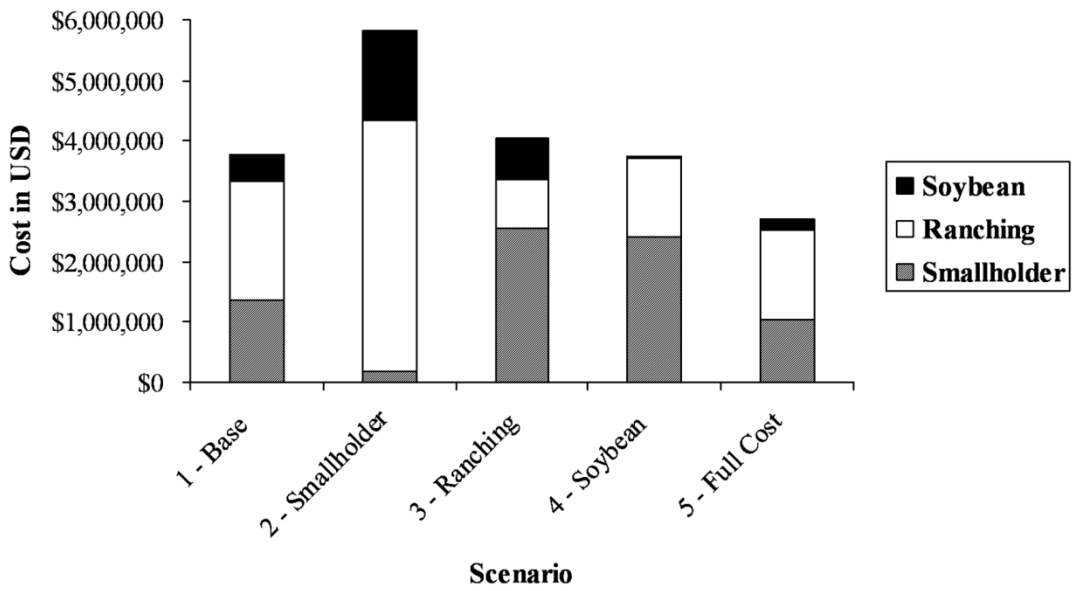


Figure 5.3 Opportunity costs of the best reserve designs for the five planning scenarios. Costs are in USD and are the opportunity costs to each of the stakeholder groups. The full opportunity cost is the sum of the three partial opportunity costs and is represented by the stacked bar.

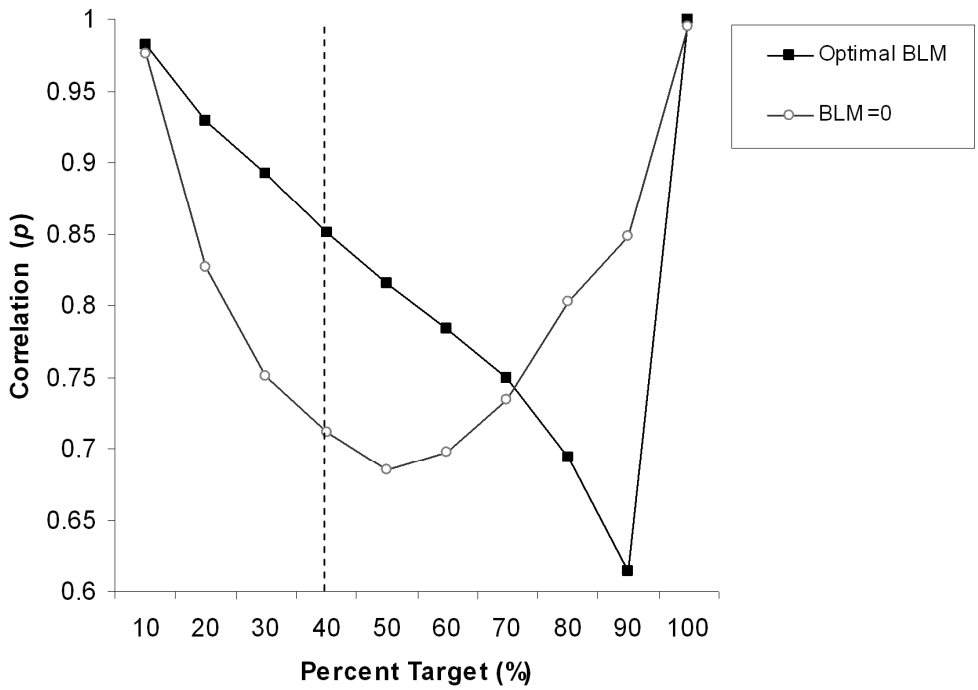


Figure 5.4 Correlations of selection frequencies of planning units between scenario 1 (base, homogeneous costs) and scenario 5 (full opportunity costs) for a range of target percentages with and without BLM. Optimal BLMs were 0.1 for scenario 1 and 0.2 for scenario 5. Dashed line indicates the target used in this study (40%).

Discussion

How do spatial correlations between costs, relative spatial variability of costs, and size of conservation targets affect the importance of selecting accurate cost surrogates?

Previous studies have shown that the efficiency gains of including costs in the planning process are strongly related to the correlation and relative variability of costs and benefits (Babcock et al. 1997; Ferraro 2003a; Naidoo et al. 2006). Babcock et al. (1997) considered under what conditions cost-efficient allocations for a fixed budget resulted in large differences in total benefits as compared to simply maximizing total benefits. They concluded that the two approaches (cost-efficient and maximizing benefits) converge when benefits and costs are negatively correlated and diverge when they are positively correlated. It is therefore important to understand the spatial variability of the conservation features (benefits) as well as the opportunity costs being considered. Additionally, the relative spatial patterns of partial costs, considered here, are likely to influence the importance of choosing the most informative cost surrogate. The size of the conservation target for biodiversity features (benefits) might also affect the spatial overlap between areas selected with explicit costs and those selected assuming costs to be homogenous, but this has not previously been investigated.

In my study region I found no correlation between full opportunity costs and complementary vegetation as a rough measure of benefit, implying that some efficiency gains could be expected in explicitly minimizing full opportunity costs (Babcock et al. 1997). The closest surrogate to full opportunity costs was the opportunity cost to soybean farming which was highly correlated with full opportunity costs but more spatially variable. I can understand why cost to soybean farming was the best available surrogate of full opportunity costs by considering the relative magnitude of benefits from alternative land uses. The net present value of soybean farming ($1,010 per ha) was five times greater than that of smallholder agriculture ($192 per ha) and four times that of ranching ($281 per ha) (for full details see Naidoo & Adamowicz 2006) and therefore largely determined the opportunity costs of conservation. Yet, soybean farming was much more spatially variable than full opportunity costs ($c_{soybean}$ = 360.77 compared to c_{full} = 87.64). Thus, while soybean farming was the best available surrogate for full opportunity costs it still performed relatively poorly as a surrogate. Selections based on area and soybean farming had the strongest correlations with those based on full opportunity costs (Table 5.2c) and also had the smallest losses in efficiency (Figure 5.3). Selections based on ranching and smallholder agriculture had progressively weaker

correlations with those based on full opportunity costs (Table 5.2c) and progressively larger losses in efficiency (Figure 5.3).

I explored how different targets affected the spatial similarity of reserve designs that explicitly considered full opportunity costs and that assumed that costs were homogeneous. Without the boundary length modifier (BLM), the inclusion of explicit costs resulted in a maximum efficiency gain (minimum correlation between solutions with and without costs) for percent targets around 50% and lower efficiency gains for lower and higher targets. When I used optimal BLMs for each scenario there were higher correlations than without BLM for low to moderate targets and lower correlations for very high targets. There are two reasons for these differences with optimal BLMs. First, the core protected area was locked into my MARXAN analyses and, regardless of cost measure, all solutions with BLM tended to contain planning units around this core, tending to align the two scenarios. Second, as targets increased, clusters of areas became more irreplaceable in the south and west of the biosphere reserve but these clusters were different for solutions with and without explicit costs, causing the solutions to diverge. The efficiency gains that I noted from including full opportunity costs were therefore influenced by my target of 40% of each vegetation type and my use of optimal BLMs. My reported cost savings would have been smaller (correlations higher) for lower targets and would have been larger (correlations smaller) for larger targets up to 90%. For my 40% target, my use of optimal BLMs gave smaller cost savings (higher correlations) than without BLM. The same would have been true for targets ranging from 20% to 70%. Cost savings reported by previous studies using explicit costs (Carwardine et al. 2008; Polasky et al. 2001; Stewart et al. 2003) are therefore likely to have been influenced by decisions about targets and BLM. This sensitivity will need to be considered in future studies considering costs.

Does using a surrogate measure of opportunity costs result in cost efficiencies?

If land use patterns by different stakeholder groups are spatially very dissimilar (negatively correlated) then any partial measure might be a poor surrogate for full opportunity cost. For more homogenous systems in which different land use patterns are spatially more similar (positively correlated) any surrogate is more likely to reflect full opportunity costs. This means that understanding the correlations between partial estimates of opportunity costs as well as the distributions of opportunity costs to single stakeholders is important when selecting a surrogate measure. In my case study the correlations between the three opportunity

costs were either negative or very slightly positive, indicating that land use patterns were dissimilar. Therefore, all three partial estimates of opportunity costs failed to minimize full opportunity costs by wide margins. The best surrogate measure of full opportunity costs was opportunity cost to soybean farming, yet even using this surrogate resulted in a 41% ($1.2 million) increase in full opportunity costs over the scenario in which that explicitly included full opportunity costs to guide selections. In previous studies, surrogate measures of full opportunity costs such as opportunity costs to commercial fishermen (Richardson et al. 2006) or rock lobster fisherman (Stewart et al. 2003) have been used with the implicit assumption that full opportunity costs were being minimized. My study demonstrates that surrogate measures do not necessarily minimize opportunity costs.

Significantly, I found that assuming homogeneous costs performed as well or better than partial estimates of opportunity costs in reducing full opportunity costs. The full opportunity costs in scenario 1 (base) and scenario 5 (soybean) were identical. There has been an increasing call to include costs in conservation planning in an effort to make cost-efficient decisions (Naidoo et al. 2006; Polasky 2008), yet my study indicates that using partial estimates of costs might produce less efficient outcomes than assuming homogenous costs. Therefore, if cost-efficiency is the ultimate goal, a better understanding of the components of opportunity cost will be needed (Naidoo et al. 2006). The potential losses in efficiency (a minimum of $1.2 million in my study) suggest that large investments in developing complete cost data can be cost-effective for conservation.

How does including a surrogate measure of opportunity costs affect different stakeholders in the planning region, or in other words, who is really paying for conservation?

Considering the distribution of opportunity costs to single stakeholders is important both for minimizing opportunity costs of conservation and ensuring that the selection of a surrogate measure does not result in displacements of costs to groups whose activities have not been considered. Displacement is likely if opportunity costs to different stakeholder groups are spatially dissimilar. In my planning region, using any of the three cost surrogates disproportionately displaced costs to the other stakeholder groups and increased full opportunity costs. Reducing opportunity costs to smallholder farmers increased costs to ranching and soybean agriculture by 200% and 700%, respectively, compared to scenario 5 that minimized full opportunity costs. When costs were minimized for ranching and soybean,

costs to smallholders increased by 150% compared to scenario 5. Minimizing full opportunity costs spread costs much more evenly among the three stakeholder groups.

There are many economic goals that can be represented by the inclusion of appropriate cost measures in conservation planning tools. The most common goal has been to include cost measures that minimize the financial costs to conservation organizations, such as land acquisition and perpetual management costs (Faith et al. 1996; Naidoo et al. 2006). A more recent approach has been to include cost measures that represent forgone opportunities for resource users (Cameron et al. 2008; Klein et al. 2008a; Klein et al. 2008b). By including these costs, the conservation plan can explicitly minimize local socio-economic impacts and potentially result in a more implementable conservation plan which has larger buy-in with the local community. Additionally, recent advances in conservation planning tools have resulted in the release of Marxan with Zones, which allows for the use of explicit costs and economic goals with respect to each stakeholder group (Watts et al. 2009). Coupling explicit knowledge of costs to each stakeholder group with Marxan with Zones might ensure a more equitable distribution of costs among stakeholders (Klein et al. 2009b).

In many regions, it will also be important to consider the stakeholders represented by each opportunity cost and their role in the success and implementation of conservation action as well as the alignment of conservation costs with wider policy settings. Including the opportunity costs to different stakeholder groups might also reflect different strategies for development. For example, minimizing opportunity costs to soybean farming could represent a macroeconomic goal at a national level to encourage multinational investment. Considering opportunity costs to smallholder agriculture might represent the goals of minimizing local impact and reducing local poverty and displacement of communities.

Linking poverty reduction and conservation planning has become a priority (Adams et al. 2004). This means that conservation in developing nations requires more than a preoccupation with minimizing the total costs of conservation. It requires an understanding of the social and economic effects of individual opportunity costs for people with least wealth and influence. In my study, a full accounting of opportunity costs achieved a cheaper solution than assuming homogenous costs. It also reduced costs proportionately to all stakeholder groups. Even so, this did not consider that different groups might respond very differently to changes in income. Smallholder farmers on low incomes might have smaller tolerances to

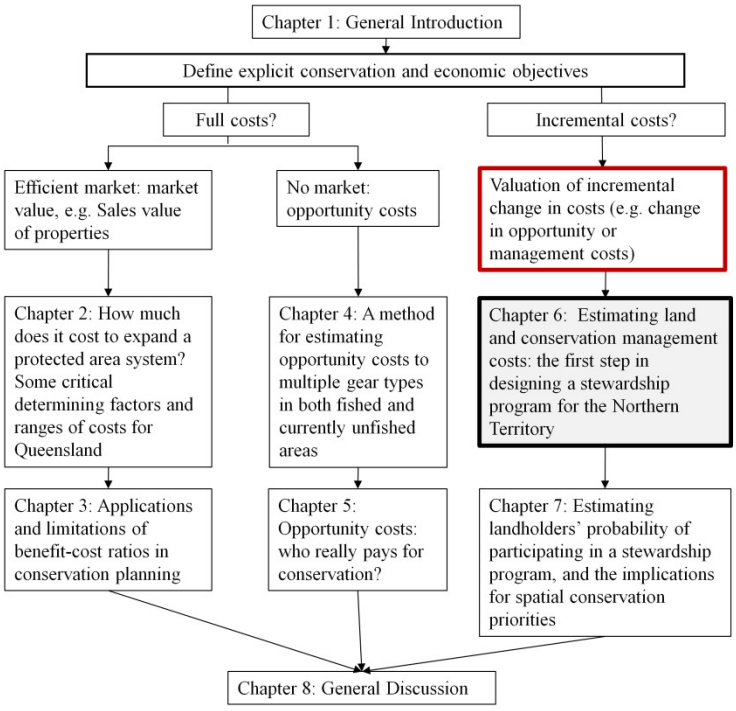
reduced income (e.g. Ferraro 2002) and some stakeholders in a region could have household incomes below the poverty index. It is therefore important to consider options to increase their incomes even if this shifts costs to other groups. The proximity of different proposed reserves to existing settlements could therefore be important to consider when planning the final reserve network. Ghate (2003) showed that the placement of a protected area in India affected nutrition standards of the local population because of reduced incomes.

The costs to stakeholder groups discussed here assume that there is not adequate compensation for their land. In many scenarios, where land markets are fully operational and land rights are well defined, the stakeholders could choose to sell their land and would receive full compensation (Knight et al. 2010). In this case it would simply be the conservation organization or society at large that pays for conservation. However, in many cases the assumptions that full compensation is provided does not hold true. In the case of direct or indirect payment schemes, mechanisms for payment to stakeholders have often been reported as flawed or unsuccessful (Balmford & Whitten 2003; Ferraro 2002; Ferraro & Kiss 2002). Furthermore, the opportunity costs to subsistence farmers might not adequately provide for a household to move and buy another parcel of land elsewhere where the land value could be much higher. Therefore, mobility and access to resources, in addition to opportunity costs and compensation issues, play heavily in determining who pays for conservation and whether detrimental effects are felt by stakeholder groups (Cinner et al. 2009; McClanahan et al. 2008). To avoid these detrimental effects it will be important for planners to understand how much land can be removed from production while still allowing income growth, and to design sustainable alternative sources of income (Balmford & Whitten 2003).

More generally, it is clear that analysis of a single surrogate for cost will not reflect the true socio-economic characteristics of a region. Rather, there are likely to be major benefits from a complete investigation of affected stakeholders, their different land uses and opportunity costs, and ways of balancing conservation and economic goals. Different conservation priorities will be identified by including different costs, emphasizing the need for care in selecting cost surrogates. Previous studies have taken an important step in incorporating cost surrogates into planning (e.g. Carwardine et al. 2008; Richardson et al. 2006; Stewart & Possingham 2005), but my results indicate the importance of more thorough analyses of the costs and socio-economic impacts of conservation plans.

Chapter 6 Estimating land and conservation management costs: the first step in designing a stewardship program for the Northern Territory[1]

Stewardship programs providing financial incentives for conservation on private lands are increasingly common. I estimate the potential costs of a stewardship program in the Daly River catchment, Northern Territory, which would underwrite the cost difference between routine land management and the additional requirements of conservation management on grazing properties. Based on survey responses from landholders, I first assess the current costs of land management in the catchment and use regression to identify key drivers of spatial variation in both routine land management costs and conservation-oriented management costs. I define conservation-oriented management costs as the costs required to meet objectives for both routine property management and conservation. I then estimate the additional costs of conservation management over and above routine land management at an average of $1.99 per ha. I conclude that, if the most cost-effective properties are targeted, an annual budget of $1,000,000 would support stewardship agreements covering 90% of the catchment's area. Much of the cost-effectiveness of stewardship payments would come from the significant economies of scale in managing large pastoral properties and leveraging the costs of routine land management already met by landholders.



[1] Adams, V.M., Pressey, R.L., Stoeckl, N., In review. Estimating land and conservation management costs: the first step in designing a stewardship program for the Northern Territory. Biological Conservation

Introduction

Public reserve systems are no longer considered sufficient for biodiversity conservation and there is a need for 'off-reserve' programs to complement protected areas (Chazdon et al. 2009; Joppa et al. 2008). One reason is the limited potential for protected areas to expand across private tenure and traditionally owned areas, related both to restrictions on budgets for acquisition (James et al. 1999a) and the willingness of landholders to engage in conservation (Guerrero et al. 2010). Other reasons are disproportionately high management costs of the often small and isolated reserves on private land (Bruner et al. 2004; Frazee et al. 2003), and the need to buffer protected areas from the adverse impacts of changes in nearby land uses (Hansen & DeFries 2007).

As a response to these constraints, competitive easement and payment programs are increasingly common policy tools for achieving conservation on private land. Examples include the US Conservation Reserve Program (CRP) (Claassen et al. 2008) and the Victorian BushTender Program (Stoneham et al. 2003). These programs are designed to consider both the costs and benefits of proposed projects when selecting successful bids. Research has demonstrated that programs such as these can increase their cost-effectiveness by considering complementarity (Hajkowicz et al. 2007) and multiple spatial criteria (Wunscher et al. 2008).

Depending on the payment program, objectives for conservation of biodiversity or ecosystem services vary, and this variation is reflected in the valuation of payments (Ferraro & Kiss 2002). For example, payments could be for 'retirement' of land from production, such as the CRP program, and therefore reflect the opportunity costs of lost production to the landholder (hereafter simply referred to as 'opportunity costs'). Alternatively, payments could be for provisioning of ecosystem services and therefore reflect management as well as opportunity costs (Pagiola 2008). Lastly, conservation easements can stipulate management requirements without restricting land use such as grazing intensity or numbers of livestock (Rissman 2010). In this case, payments would reflect management costs rather than opportunity costs, where management costs include staff expenses (e.g. salaries), regular operational costs (e.g. fuel and other supplies), and recurrent capital costs (e.g. purchase and replacement of vehicles and field gear), together referred to as recurrent annual expenditures (Bruner et al. 2004; Naidoo et al. 2006).

In this chapter, I estimate the cost of a stewardship program with characteristics typical of current strategies for conservation management agreements and covenants in Australia. These strategies predominantly target changes in land management rather than land use. Covenants in Australia typically have permanent title implications for a nominated portion of properties, although compatible and sustainable land use can continue. For example, in Queensland, landholders can elect to apply a voluntary perpetual conservation covenant, known as a Nature Refuge (IUCN category VI protection), on areas of high cultural and/or ecological significance on their properties. However, grazing can continue on these properties and Nature Refuges might have little or no impact on stocking rates. This is because pastoral properties in northern Australia are generally very large (tens to hundreds of thousands of ha) and are generally poorly developed in terms of subdivision, fencing and water-points (Oxley et al. 2005). Consequently, paddocks (fenced management areas within properties) are large and stocking rates are low – less than 10 head of cattle per square km in 2001 (Fitzsimons & Wescott 2007). Agreements that require graziers to set aside relatively small areas of vegetation or change fire management regimes thus have little to no impact on production. Therefore, the costs of stewardship programs on extensive pastoral properties are predominately those associated with management (rather than opportunity) costs. Here I focus on the management costs of a stewardship program that seeks to change the management regimes of pastoral properties.

Data on costs for use in conservation planning inevitably require spatial modelling. Modelling of management costs uses known costs and potential cost drivers in some areas to predict the management costs of other areas based on characteristics such as size, landscape context, and requirements for infrastructure. The few studies relevant to local variations in management costs within planning regions (Blom 2004; Frazee et al. 2003; Wilkie et al. 2001) are unlikely to provide cost estimates that can be exported to other regions because of idiosyncratic cost structures and between-region variation in the relative importance of predictors. Most models of management costs have been global in extent (e.g. Balmford et al. 2003; Bruner et al. 2004; James et al. 2001; James et al. 1999a; James et al. 1999b). While estimating broad variation in management costs, global studies lack the resolution to identify within-region variations in costs, and can be inaccurate for specific areas (Ban et al. 2011).

The literature has therefore provided little guidance on *a priori* estimation of the potential costs of conservation programs that target only changes in management regime. This is in

contrast to the well established estimation techniques for programs that compensate for opportunity costs (e.g. Chomitz et al. 2005; Stoms et al. 2004). Thus, there is a clear gap in the literature regarding appropriate estimation methods for payments associated with stewardship programs such as those being implemented in Australia (e.g. the Nature Refuge program), the United States (e.g. the conservation easement program described by Rissman, 2010), and Canada (e.g. covenants and conservation agreements, Fischer et al.).

My main aims in this study were to estimate the costs to government of a stewardship program, and to examine the potential for such a program to achieve spatial conservation objectives. I therefore required estimates of the spatially variable costs of routine land management and the additional costs of achieving conservation objectives. I use data from the Daly River catchment in the Northern Territory, Australia as a case study (Figure 6.1). I apply regression techniques based on property-scale variables to estimate routine land management costs and conservation-oriented management costs at the resolution of individual properties to answer the following questions:

1. What factors drive routine land management costs in the Daly catchment?
2. What are the current expenditures on routine land management by landholders in the Daly catchment?
3. How much could a stewardship program cost in the Daly catchment if it covered the additional activities of landholders, over and above routine land management, to achieve conservation objectives?

I also examine preferences of landholders for the structure of stewardship payments. It is likely that the structure of the program will influence landholders' interest in participating, so preferences are highly relevant to the design of the stewardship program.

Methods

Study region

The study region was the whole of the Daly River catchment in the Northern Territory, buffered by 10 km to ensure that peripheral conservation features and properties were considered (Figure 6.1a). The Daly catchment is approximately 5.2 million ha (7.9 million ha with buffer), extending from the coastline south-west of Darwin to 250 km inland. The Daly River and its main tributaries are themselves important conservation features, the Daly being one of northern Australia's largest rivers with unusually consistent year-round flow. Riparian strips contain some of the most extensive gallery (rainforest) vegetation in the

Northern Territory. The average size of private properties in the Daly is ~10,500 ha. Properties larger than 5,000 ha represent approximately 13% of landholders but about 90% of the catchment's private land (Figure 6.1b). Although there are many conservation priorities within the catchment, the Northern Territory government has indicated that it is unlikely that these priorities will be addressed with further acquisition of properties for national parks because of the large property sizes and correspondingly large acquisition and management costs. Instead, the region is suitable for off-reserve programs involving stewardship payments in conjunction with conservation agreements between the government and landholders.

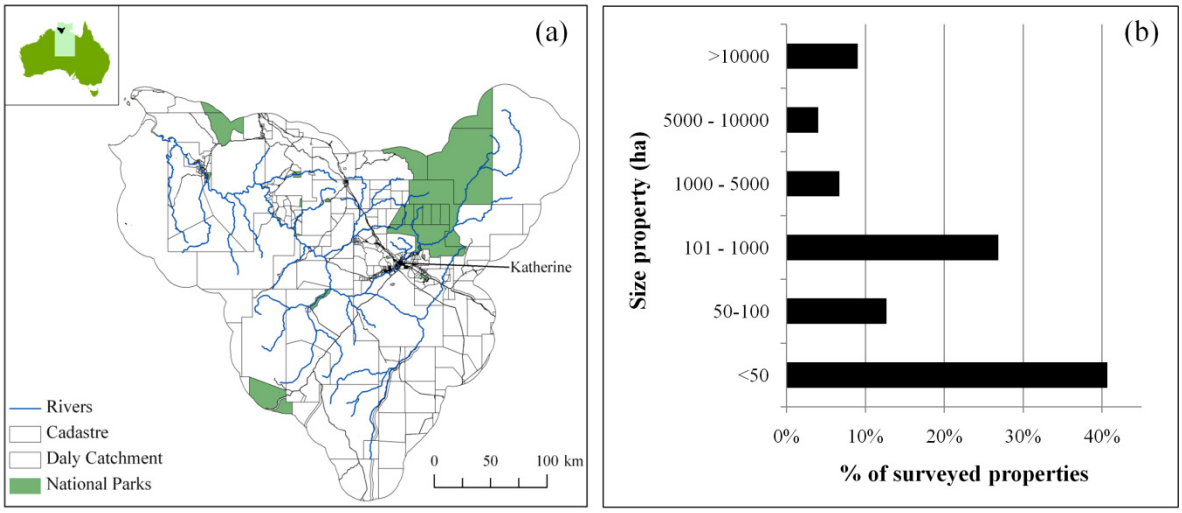


Figure 6.1 The Daly catchment and pastoral and horticultural properties. (a) Rivers, national parks and boundaries of properties. Two large national parks adjoin near the north-east corner of the catchment: Katherine Gorge National Park and the southern portion of Kakadu National Park. The map inset shows the Northern Territory with pale shading and the Daly catchment in black. (b) Size distribution of the 440 properties included in my survey.

While the catchment has experienced low levels of clearing (~5%), changes in fire regimes have been dramatic and increased weed infestations threaten native species. These changes, together with long-term grazing have been implicated in the decline of the region's mammals and granivorous birds (Franklin et al. 2005; Woinarski et al. 2010; Woinarski et al. 2011). Because there has been little clearing, revegetation and changes in land use will not be a focus of stewardship agreements in the area. Rather, agreements will address changes in management regimes to mitigate the predominant threats on private properties to meet conservation objectives.

Routine land management and conservation-oriented management

A full list of threats to land production and native species and the types of routine and conservation-oriented management activities recommended to mitigate threats is in Table 6.1. Importantly, many of the land management practices routinely used for pastoral enterprises in the Daly catchment are similar to those recommended for conservation. For example, weed control protects pastures and stock, but also native species, and graziers remove weeds manually, mechanically or with chemicals or cool burns. Control of feral animals also has benefits for both production and conservation. Graziers use routine fire management practices – typically a combination of maintaining fire breaks and reducing fuel loads through cool burns - to reduce the risk of uncontrolled fires that threaten stock and property. Fire management is important also for biodiversity conservation, although with the objective of increasing the heterogeneity of post-fire successional patches across large landscapes. In this case, modification of routine burning regimes would improve the biodiversity outcomes of fire management (Woinarski et al. 2011). Fencing projects are typically used by graziers to deter unauthorized access to the property, exclude feral animals, or adjust grazing pressure on native pasture species. Fencing also contributes to conservation objectives, though with some additional costs, for example to protect grazing-sensitive rainforest patches and riparian zones. In short, there are many synergies between routine land management and conservation management in the Daly catchment and across the northern rangelands generally, and much potential to adjust routine practices to achieve conservation outcomes without large financial outlays (Table 6.1).

Survey design

I structured my survey of landholders to relate management activities to factors that threaten both production and conservation values, namely: fire, weeds, feral animals, erosion, and unauthorized access. Specifically, I gathered data on the percentage of property threatened, the frequency of land management activities (e.g. controlled burning, removal of weeds) associated with mitigating these threats, and the number of labour days required per month to undertake land management activities. I also asked landholders to provide a full list, for the previous year, of: (a) routine land management costs (broken down into those associated with labour, supplies, long-term infrastructure, and other items); and (b) conservation-oriented management costs (with a similar break-down). Questions were structured to reflect previous studies on conservation costs (Balmford et al. 2003; Balmford et al. 2004; Gravestock et al. 2008; Moore et al. 2004) (for full survey see Appendix 4).

Table 6.1 Threats to land production and conservation values. For each threat are listed the impacts to production and natural values, associated routine land management and conservation management actions, and expected changes from routine to conservation management.

Threat	Impacts to production and natural values	Routine land management	Conservation management	Change
Weeds	Weeds threaten both production and natural values by altering the composition of native vegetation, replacing palatable native plants, and threatening crops.	Chemical applications, cool burns, manual and mechanical removal.	Chemical applications, cool burns, manual and mechanical removal.	Change from routine to conservation management would require potentially reducing pesticides and investing in more labour for manual removal of weeds. In addition, conservation management might require more intensive weed management in native pastures.
Fire	Fires reduce the biomass of native pastures, injure stock, damage fencing and other infrastructure, and, with inappropriate regimes, reduce populations of many native plants and animals.	Maintaining fire breaks and using cool burns (early in the dry season) to reduce fuel loads of unpalatable plant material.	Maintaining fire breaks and using cool burns (early in the dry season) on a rotational patch basis to maintain the heterogeneity of post-fire successional stages.	Change from routine to conservation management would require a change in the regime of cool burns from annual to irregular spatially and temporally.
Erosion	Erosion can damage pastures, native vegetation, and cause siltation of wetlands, stock water-points, and water courses.	Maintaining vegetation cover by avoiding overgrazing; revegetation, by removal of stock or other means, of areas susceptible to erosion.	Maintaining vegetation cover by avoiding overgrazing; revegetation, by removal of stock or other means, of areas susceptible to erosion; exclusion of grazing from sensitive areas such as wetlands and watercourses.	Change from routine to conservation management might reduce grazing in pastures where vegetation cover is critically low or fence sensitive areas such as wetlands and watercourses in conjunction with creating alternative water-points so that stock can still access water. Cell grazing, in which smaller paddocks are grazed on a rotational basis, can maintain production while reducing erosion.
Feral Animals	Feral animals cause erosion along banks of wetlands and streams, reduce production by impacting pastures, and reduce populations of some native plant and animal species.	Culling feral animals; fencing pastures to exclude feral animals.	Culling feral animals; fencing sensitive native vegetation to exclude feral animals.	Change from routine to conservation management would require potentially increasing time spent on culling activities and additional fencing of vegetation adversely affected by feral animals.
Unauthorized access	Unauthorized access for recreational use of off-road vehicles or hunting feral pigs can damage fence lines, kill or injure livestock, cause erosion, and damage native vegetation.	Fencing is the main strategy for minimizing unauthorized access but is often not effective.	Fencing is the main strategy for minimizing unauthorized access but is often not effective.	No change.

To help inform the design of the stewardship program, I also asked landholders to indicate their preferences for types of funding. These payment types were identified by a focus group of managers employed by the Northern Territory Department of Natural Resources, Environment, the Arts and Sport (NRETAS) and reflect current or potential funding structures for off-reserve conservation programs (Table 6.2). I asked landholders to rate the types of funding from 0 to 5, with 0 being "would not consider" and 5 being "would be very interested".

Table 6.2 Potential payment types for stewardship schemes identified by a focus group of managers employed by the Northern Territory Department of Natural Resources, Environment, the Arts and Sport (NRETAS).

Supplies	Stewardship agreements with landholders would include explicit lists of conservation management activities for each property, such as fencing, weed management, and fire management. Supplies needed for agreed activities, for example fencing supplies and grading equipment for fire management, would be provided as payment.
Financial payment	Financial payments are typical of stewardship payment programs (e.g. Hajkowicz et al. 2007) and would constitute an agreed amount to cover the activities specified in a stewardship agreement for each property.
Clearing offset	There are strict clearing laws for the Daly catchment allowing a maximum of 30% of any vegetation type to be cleared across the whole catchment and a maximum of 70% of any property to be cleared (NRETAS 2010). Many landholders have stated a desire to clear more than the maximum property allowance. A clearing offset would allow greater than the 70% clearing maximum on a property, given that the property contains conservation features of interest that would remain unaffected by clearing and that the landholder enters into a stewardship agreement.
Conservation certification	A conservation certification would be a type of public recognition for a property participating in a stewardship agreement. This option could give a marketing advantage to property owners with conservation tourism on their properties.
Rangers	Stewardship agreements with landholders would include an explicit list of conservation management activities for each property, such as fencing, weed management, and fire management. Rangers employed by the Northern Territory Government (e.g. Indigenous rangers or conservation managers) would provide labor and access to equipment such as pesticide sprayers and graders to support agreed conservation actions.

Sampling and response rates

For my survey, I considered only land parcels of 10 ha or larger and outside the town of Katherine. Properties in the town or smaller than 10 ha are unlikely to be good candidates for conservation agreements because they are predominantly residential and unmanaged. Excluding the town of Katherine, 763 parcels covered at least 10ha. But not all parcels were held by private landholders. While stewardship agreements might be appropriate for Aboriginal and some public land not already committed to conservation, the method of engagement and types of payments would differ from those for landholders engaged in cropping or grazing. I therefore excluded land held by Aboriginal communities (50 parcels), and government agencies (192), leaving 521 "relevant" parcels. The total area of the remaining land to be included in my survey was 4.2 million ha (54% of the catchment) and the mean parcel size was 10,500 ha. Surveys were mailed to landholders of all 521 parcels between May and August 2009. In-person interviews were conducted in September 2009 with some respondents who had not replied by mail.

There were 20 in-person interviews, with some landholders electing to respond in this way, others identified by referrals from these respondents, and the remainder randomly sampled from remaining properties. These interviews involved the same wording of questions in the same order as the mail survey. Responses to in-person surveys were not statistically different from those to mailed surveys, and in-person respondents did not ask for clarification of questions, indicating that interpretation was consistent with the intended meanings of questions. Data were aggregated to owners where they held multiple parcels because owners indicated that they managed properties as a whole rather than as separate parcels (440 owners for 521 properties). The response rate to the survey was ~23% (92 of 440 owners), reflecting average response rates in social science surveys. The distribution of my total survey respondents (92), by size of property, was representative of the property size distribution for all properties (Table 6.3).

Of the 92 surveys completed by mail or in-person, 21 respondents indicated that they were not actively managing their land. Of these 21 properties, 65% were smaller than 50 ha and, according to respondents, were solely residential, making land management unnecessary or inefficient. Excluding these left a sample size of 71 for analysis. Of the 71 remaining respondents, 15 reported that portions of their land were used for crops, requiring dissimilar routine land management to that for pastoral use. While the routine land management associated

with the grazed portions of these properties would be compatible with conservation-oriented management, I was unable to separate the respective portions of management costs. I therefore excluded properties with cropping from further analysis, leaving 56 properties in my sample.

Table 6.3 Distribution of property sizes for the 440 properties across the Daly catchment identified for the survey (Full catchment) and across the 92 survey respondents (Returned surveys).

Property size (ha)	Full catchment	Returned surveys
<50	41%	30%
50-100	12%	9%
100-500	23%	29%
500-1000	5%	7%
1000 - 5000	6%	9%
5000 - 10000	2%	4%
>10000	11%	12%

Modelling routine and conservation-oriented management costs

I checked responses for internal consistency by comparing the stated number of labour days per month against the frequency of stated land management activities. I also checked for consistency between the stated land management threats and stated land management activities to mitigate them. I validated stated costs against stated activities and current market prices for standard supplies such as fencing materials and pesticides. In all cases, relations were as expected. For example, there were more labour hours and higher labour costs for properties with more threats, and higher costs of fencing and pesticides on larger properties. Responses indicated that feral animals, particularly pigs, are typically culled for recreation rather than necessity. Therefore, landholders do not cognitively track the costs associated with feral control, and this activity is not viewed as imposing a net cost on property management.

Information collected in the surveys was supplemented with digital data, such as vegetation cover on each property, supplied by NRETAS. The surveys and digital data together provided a wide range of variables associated with routine land management activities (Table 6.1) and numerous property descriptors. I tested the full set of variables for correlations and selected a subset for spatial modelling of management costs that best represented land management activities and property characteristics (Table 6.4).

Table 6.4 Variables, with data sources, selected for spatial modelling of management costs based on characteristics of properties, characteristics of landholders, and types of threats (fire, weeds and unauthorized access).

Property and landholder characteristic	Variables selected	Data source
Size of property	ln(property area, ha)	Digital
Land use	ln(graze area, ha)	Digital
Natural characteristics: amount of rainforest and wetlands	ln(rainforest ha), ln(wetlands ha)	Digital
Familiarity of property owner with land	Years on property	Survey
Engaged in conservation efforts	Conservation flag (binary variable, 1=respondent currently manages for conservation objectives)	Survey

Threats	Corresponding actions	Variables Selected	Data Source
Fire	Fire break maintenance, cool burns, weed management to reduce fuel load	Fire management (with weed removal) Given in number of weeks per year management activity was undertaken, 0-52	Survey
Weeds	Weed management with chemicals, manual removal, or cool burns	Weed management (with chemical) Weed management (with fire) Weed management (manual removal) All given in number of weeks per year management activity was undertaken, 0-52	Survey
Unauthorized access	Fencing	Unauthorized access Measured as a percentage of property affected	Survey

With data on the costs of routine land management and conservation-oriented management (routine land management plus additional conservation management) on the surveyed properties, I used two methods to model these costs spatially. One purpose of modelling was explanatory - to understand the factors influencing management costs on the surveyed properties. For this purpose, I therefore developed models for routine and conservation-oriented management costs using all available predictors from the survey and digital data. Another purpose was predictive - to extrapolate conservation management costs across the catchment as a basis for estimating stewardship costs. For this purpose, I therefore developed models for routine and conservation-oriented management costs using only digital data available for all properties.

In both cases I used a modified Cobb-Douglas cost function. The Cobb-Douglas cost function corresponding to the two-input Cobb-Douglas production function can be expressed as a function of *w,* the price of labour, *r*, the price of capital, and *q*, the quantity of product.

$$c(q, w, r) = Aw^{\varphi} r^{\gamma} q^{\delta}$$

where *A, φ, γ, δ* are a function of the Cobb-Douglas production function output elasticities α and β.

A more general form of the function is

$$c = I^{\alpha}$$

where *c* is the cost of output, in this case management, and *I* is a predictor of input cost.

Table 6.5 Cobb Douglas cost function coefficients for the natural log of routine land management cost inputs regressed again the natural log of property size.

	Constant	Coefficient, ln(property size, ha)	*p*
Labour (person days per ha)	2.2309	-0.7795	<0.001
Long-term input ($) per ha	1.3700	-0.137	<0.001
Intermediate input ($) per ha	5.2289	-0.5072	<0.001
Cost ($) per ha	6.7663	-0.591	<0.001

I natural-log transformed the function and tested the components of *c* on a per unit basis, specifically long-term capital, materials, supplies and labour as a function of land area to examine whether land area was a good predictor of per unit costs of capital and labour (Table 6.5). I found that all components of the cost function had significant economies of scale (Table 6.5). Therefore, I used the general form of the Cobb-Douglas function and hypothesized that routine land management cost was a function of the land area, land management practices, and property characteristics. The Cobb-Douglas functional form used was therefore

$$\ln(cost\ per\ ha) = a + b * ln(property\ size) + c * management\ practice + d * property\ characteristic$$

Prior to running regressions I tested the reported routine land management costs per ha as well as conservation-oriented management costs per ha for spatial autocorrelation using the Moran I statistic. The Moran I for both costs indicated that the spatial pattern was random (I=0.02 for both costs) and therefore it was appropriate to use ordinary least squares regression which does

not account for spatial autocorrelation. Lastly, I tested whether my data met the Gauss-Markov assumptions for multiple regression and found that the assumptions held true.

I performed a series of regressions for routine land management costs and conservation-oriented management costs with each of the variables listed in Table 6.4 and their combinations (without interactions) in step-forward and step-backward fashions ('step' function, R Development Core Team 2005). The final models for routine and conservation-oriented management costs had the best AICs (Akaike 1974). The predictive models using only digital data gave me alternative models for the 56 uncropped, surveyed properties but also allowed me to extrapolate management costs across all private properties in the catchment.

Estimating stewardship costs

With the final models based on variables available digitally for all properties, I estimated the costs of routine land management and conservation-oriented management (routine plus conservation management) for each property, assuming that all properties were, like those from the previous section, engaged exclusively in grazing. I estimated stewardship costs as the difference between conservation-oriented management costs and the costs of routine land management, termed here the 'additional' costs of conservation.

Using only digital data available for all properties, I found only two significant predictors. Therefore, I wanted to examine whether the additional variables available from survey data, but not available across all properties, could have a large impact on estimated stewardship costs across the catchment. Using the models based on all available variables for the reduced data set, I constructed cost curves for stewardship costs by predicting routine and conservation-oriented management costs in relation to area of property. Each curve represented a different combination of average values of other predictors from survey responses, allowing me to examine the differences in per ha cost for different combinations.

Informing design of the stewardship program

To identify the most and least 'preferred' payment types, I applied a full pair-wise comparison of the five payment types (Table 6.2) assessed with the likert scale. I used the Wilcoxon rank-sum statistic to test the null hypothesis that payment types had equal median values.

Results

Survey responses covered one million ha of the 4.2 million ha surveyed. The total reported cost of routine land management was $1.8 million, with an additional cost of conservation management of $0.35 million. From the 14 respondents who indicated conservation costs, the average additional cost for conservation management was ~56% of routine land management costs, with a range of 5%-700% depending on the size of the property. I could not determine if these 14 properties were representative of conservation features across the catchment, but the activities were typical of those covered by stewardship agreements.

The best fit model for routine land management costs, using all available variables, included three characteristics of properties and landholders (*ln property area, ha*, *Years on property*, and *Conservation flag*) and four management predictors (*Fire management with weed removal, Weed management with chemical, Weed management with fire* and *Unauthorised access*) (Table 6.6a). The adjusted R^2 value was 0.93. The best fit model for conservation-oriented management costs, using all available variables, included all of the predictors above except for *Weed management with fire* (Table 6.6a). The adjusted R^2 value was 0.909. This model had coefficients with the same signs and similar magnitudes as the model for routine management costs, except for *Conservation flag*, which described the increased costs in leveraging routine management actions for conservation objectives.

The best fit model for routine land management costs, using variables available for all properties in the catchment, included two predictors, *ln(property area, ha)* and *Conservation flag*, and had an adjusted R^2 value of 0.781 (Table 6.6b). The best fit model for conservation-oriented management costs, using variables available for all properties in the catchment, included the same two predictors and had an adjusted R^2 value of 0.778 (Table 6.6b). The coefficient for *ln(property area, ha)* was similar between the two models. However, the coefficient for *Conservation flag* was nearly double for the conservation-oriented management cost model, similar to the difference between models using all available predictors.

Table 6.6 Regression results for routine and conservation-oriented management costs. (a) Final explanatory models for surveyed properties based on all variables (survey and digital); (b) Final predictive models based on variables available for all properties (digital only). All coefficients were significant with $p < 0.001$.

(a)

	Coefficient	
	ln(routine management cost per ha, $ ha-1)	**ln(conservation-oriented management cost per ha, $ ha-1)**
Intercept	7.5052	7.3123
ln(property area, ha)	-0.6955	-0.6971
Fire management (with weed removal)	0.0776	0.0766
Weed management (with chemical)	0.1251	0.1288
Weed management (with fire)	-0.3410	N/A
Unauthorized access	-0.7804	-0.8134
Years on property	-0.0322	-0.0277
Conservation flag	0.6064	1.0240
R^2 adjusted	*0.930*	*0.909*

(b)

	Coefficient	
	ln(routine management cost per ha, $ ha-1)	**ln(conservation-oriented management cost per ha, $ ha-1)**
Intercept	6.8549	6.8675
ln(property area, ha)	-0.6536	-0.6559
Conservation flag	0.6306	1.0972
R^2 adjusted	*0.781*	*0.778*

Stewardship cost curves (Figure 6.2) based on the models for routine and conservation-oriented management costs using all predictors demonstrated that, for small properties (<10ha), stewardship cost per ha differed by up to $100, depending on land management actions (divergence beyond the presented y-axis limit of Figure 6.2). However, due to strong economies of scale, the cost curves quickly converged and, for larger properties (>1000 ha), cost curves differed by less than $1 per ha. For properties of average size (~9,000 ha), cost curves converged on $2.25 per ha and costs continued to decrease to as low as $0.28 per ha for the largest properties.

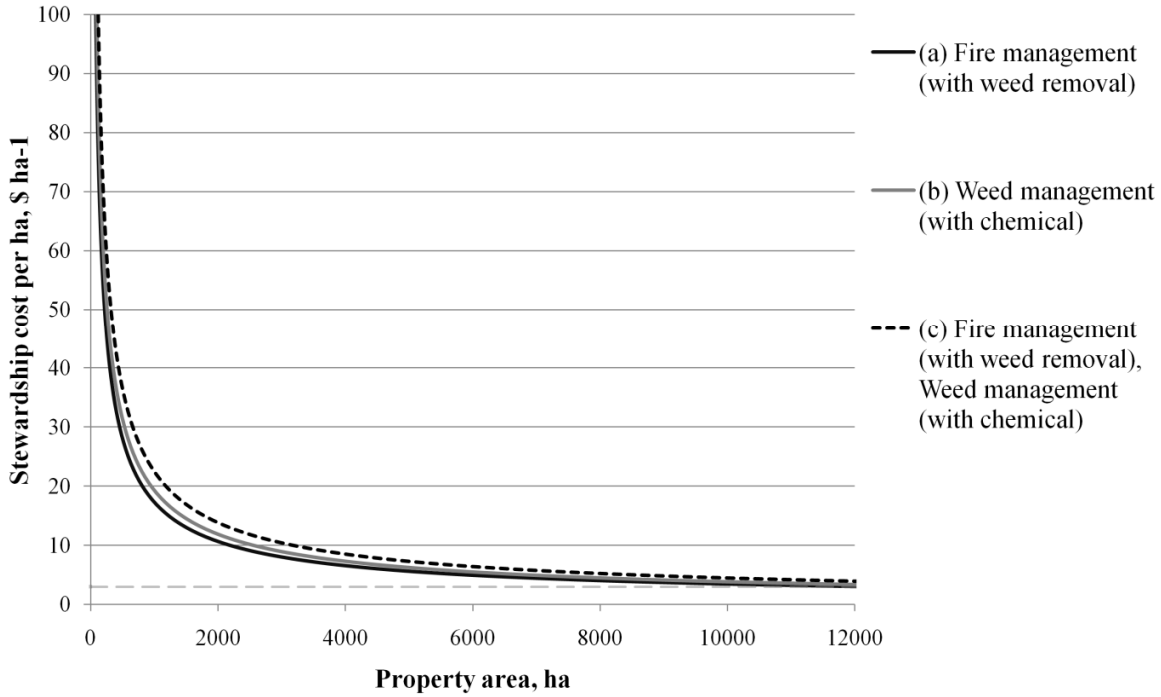


Figure 6.2 Stewardship cost curves based on models using all variables available from the survey responses and supplementary digital data. I generated cost curves by predicting routine and conservation-oriented management costs in relation to area of property using different combinations of average values of other variables from survey responses. Values for stewardship costs were then plotted as the differences between the cost curves for routine and conservation-oriented management. Curve (a) assumes an average value of 2 (twice per year) for *Fire management (with weed removal)* and zero for all other variables. Curve (b) assumes an average value of 2 (twice per year) for *Weed management (with chemical)* and zero for all other variables. Curve (c) assumes an average value of 2 (twice per year) for *Fire management (with weed removal)* and an average value of 2 (twice per year) for *Weed management (with chemical)* and zero for all other variables. The horizontal grey dashed line indicates a cost per ha of $2.25 for reference.

Using the final models based on predictors available for all properties, the estimated costs of routine land management for all of the Daly catchment was $4.2 million. The estimated catchment-wide cost of conservation-oriented management was $12.5 million. I estimated the additional costs of conservation to be the difference between conservation-oriented and routine land management costs. Therefore, for the whole catchment, additional costs of conservation totalled $8.3 million or $1.99 per ha (Table 6.7).

Table 6.7 Estimated expenditures for all relevant properties (440 pastoral properties identified for the survey, ~4.2 million ha) based on models for routine and conservation-oriented management costs using only variables available for all properties.

	Estimated cost of routine land management for all properties	**Estimated additional cost of conservation-specific activities for all properties**	**Estimated cost of routine land management per ha**	**Estimated conservation-specific cost per ha**	**Estimated total costs per ha**
Model using variables based on digital data	$4.2 million	$8.3 million	$1.01	$1.99	$3.01

Responses to the survey question regarding payment programs indicated that respondents preferred programs that would provide supplies or financial payments to clearing offsets, conservation certification, or rangers (Figure 6.3). The Wilcoxon rank-sum test statistic indicated no significant difference between supplies and financial payments, or between clearing offsets, conservation certification, and rangers. The same test indicated a highly significant difference (p-value less than 0.001) for comparisons of supplies and financial payments with the other three payment types. These first two payment methods were therefore clearly preferred by landholders over the other three.

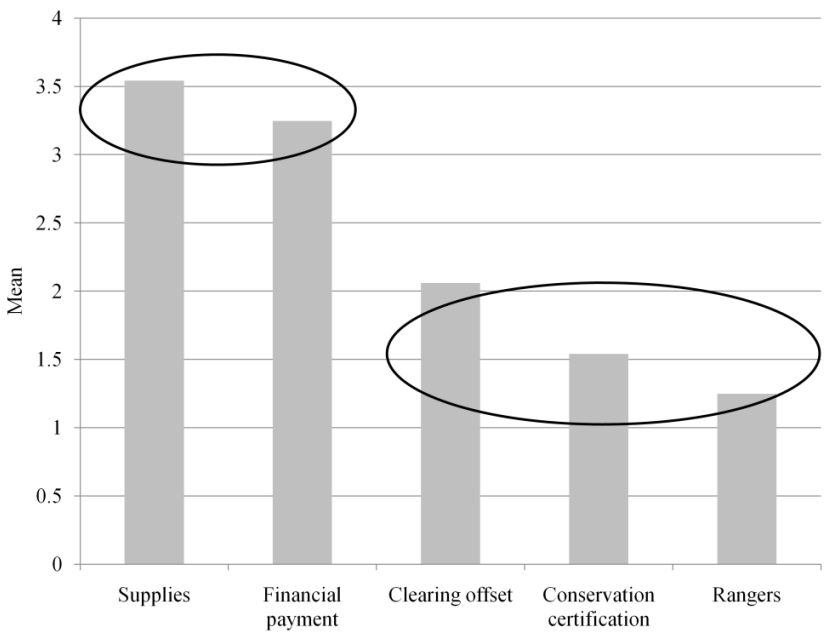


Figure 6.3 Means of stated preferences for payment types described in Table 6.4. The types of payment are ordered by mean from largest to smallest. Ovals link groups with similar preferences (Wilcoxon rank-sum test did not reject the null hypothesis that the median values were the same). The two groups were statistically different (the Wilcoxon rank-sum pair wise tests rejected the null hypothesis that the median values were the same).

Discussion

Previous studies have found economies of scale for management costs of protected areas, meaning that cost per unit area decreases with increasing area (Balmford 2003; Bruner et al. 2004; Frazee et al. 2003). I found the same trends for routine and conservation-oriented management costs in the Daly catchment, with models using both survey and digital data and those using digital data only. However, I also identified other important factors influencing both routine and conservation-oriented management costs. *Fire management (with weed removal)* had a positive coefficient. This activity is typically directed at weeds that increase fuel loads, such as gamba grass, and often requires extensive manual and mechanical work that is more costly than using pesticides and cool burns. In contrast, the negative coefficient for *Weed management with fire* indicates that using controlled burns to manage weeds can be cost-effective. I interpret the negative coefficient for *Years on property* as learning by landholders. Effectiveness of management such as cool burns depends largely on landholders' knowledge of the types of weeds present, weather conditions, and vegetation responses to weather cycles, so experience reduces the cost of management.

I expected that threats to land use would increase costs, but *Unauthorized access* had a negative coefficient. Unauthorized access can break fences, increase soil erosion, and damage paddocks, but cannot be prevented on large, remote properties without enforcement. Landholders tend to respond by reducing management of the affected portion of the property. It seems, therefore, that the reduced costs related to *Unauthorized access* reflect reduced, rather than cheaper, management.

The *Conservation flag* that I used to indicate landholders currently engaged in conservation actions had positive coefficients in all models. In the case of models of routine property management, higher coefficients indicate that conservation-oriented landholders could have a different, and more expensive, standard of routine land management activities than landholders who placed lower priority on conservation management. For models of conservation-oriented management costs, coefficients had similar signs and magnitudes to those for routine land management costs, except for a near-doubling of the *Conservation flag* coefficient. This indicates that the effects of leveraging routine land management actions to meet conservation objectives were captured in my model only by the *Conservation flag*. Given more data (more

respondents and more variables), I might have been able to determine the drivers of costs of specific conservation actions.

Because of economies of scale, stewardship costs per ha decreased dramatically as property area increased. For larger properties, the difference in cost curves fell to less than $1 per ha regardless of management activities, reinforcing property size as the strongest driver in stewardship costs (Figure 6.3). This indicates that, although I did not identify any other significant predictor variables from digital data available for all properties, my stewardship cost estimates are likely to have captured the main trends in stewardship costs in the catchment.

Properties in the Daly catchment tend to be very large, with the average property size being ~9,000 ha. For properties of average area, stewardship costs curves neared ~$2.25 per ha, regardless of management actions, and, for properties larger than 9,000 ha, stewardship costs fell to as low as $0.20 per ha (Figure 6.3). This compares favourably with the reported average conservation management costs of national parks in the Northern Territory of $2.25 per ha (NRETAS 2009a).

Interviews with park managers working in and around the Daly catchment indicated that current park management budgets were inadequate to achieve conservation management goals. This is consistent with reports of underfunding of management in global studies (Gravestock et al. 2008; James et al. 1999a). This implies that the $2.25 per ha cost for conservation management in national parks is an underestimate of the required costs to effectively manage parks. For the same per ha investment, strategic investments in underwriting the costs of conservation on private land could achieve higher levels of conservation outcomes across the catchment.

Previous subsidy programs to support fire management and wetlands management in the Daly catchment have refunded the costs of approved equipment and supplies. Responses to the survey question regarding payment types indicated strongly that landholders preferred programs structured similarly to these previous ones. Additionally, many respondents provided unprompted comments regarding their expectations of a stewardship program. Comments consistently stated that the participants were interested in government staffing support to help create conservation management plans, provide education about the ecology and conservation

requirements of properties, and help monitor the outcomes of management plans as management actions were implemented. Therefore, I believe that a well designed stewardship program would allow for supply subsidies and allocate existing NRETAS staff to support program participants in designing, implementing and monitoring conservation management plans for their properties. The transaction costs of running the program would likely be covered by a budget for two full-time staff.

Designing and implementing a conservation stewardship program for the Daly catchment will require careful consideration of the region's natural features, threats to properties, and social variables around engaging groups of landholders. Stewardship programs should ensure 'additionality' so that payments achieve substantially different levels of management than would be procured without a contract (Ferraro 2008). A program to support the additional costs of conservation-specific management in the Daly catchment would require $8.3 million for 100% participation. However, this level of participation is unlikely and landholders are also unlikely to manage 100% of their properties for conservation. Assuming that only some landholders engage in a voluntary program and that, on average, they actively manage 50% of their properties for conservation, a budget of $1 million per year would allow funding of all 'cost-effective' properties - those with less than $2.25 per ha stewardship costs (more cost-efficient to manage than national parks in the region). A successful program would be cost-effective in achieving conservation outcomes by leveraging local knowledge and underwriting the cost difference between routine land management and conservation-oriented management, estimated at $1.99 per ha annually across the Daly catchment. However, if only the most cost-effective properties are targeted, estimated stewardship payments could fall to $0.28 per ha.

Conclusions

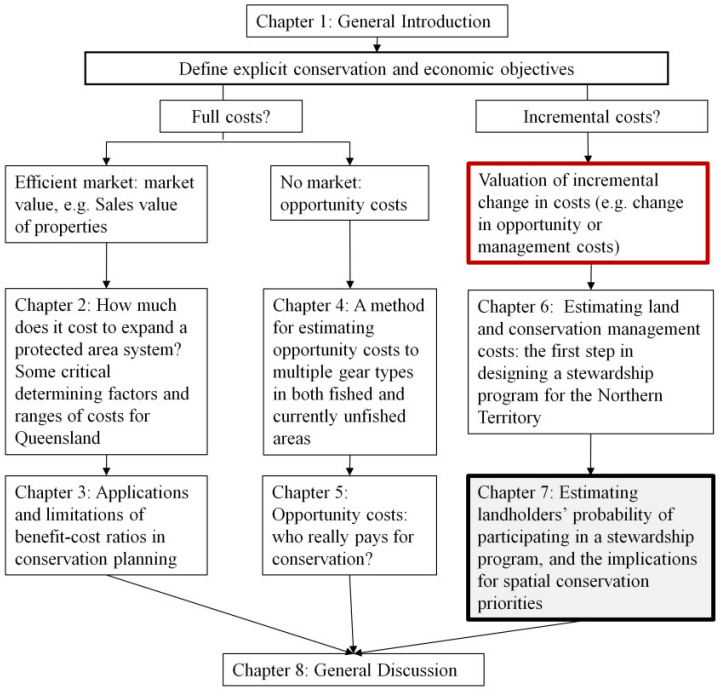
Due to the large economies of scale in both routine and conservation-oriented management costs, the largest properties in the catchment had estimated stewardship costs of less than $2.25 per ha and covered 90% of the catchment. Therefore, strategic engagement with managers of large properties would allow cost-effective conservation funding for a very large portion of the land area. This is a compelling financial argument for trialling a stewardship program, especially given that interview responses indicated positive attitudes of landholders towards a well designed program.

I believe that a program designed using the principles, financial estimates, and payment structures described here would be successful in delivering large biodiversity benefits from marginal changes in existing land management regimes such as managing fire and protecting sensitive habitats from weed invasion. The design and trialling of a stewardship program is the critical next step for the Northern Territory. The primary objective of my study was to estimate how much a stewardship program would cost in the Daly catchment if it covered the marginal costs of shifting routine management to conservation-oriented management. Although my models provide insights into the drivers of spatially variable management costs, they do not indicate how much individual activities, such as fencing or erosion control, would cost on individual properties. If a stewardship program were trialled in the Northern Territory, individual managers would develop property-specific agreements with the government, detailing the threats and associated conservation management actions specific to the property. The costs associated with those actions would be estimated at that time.

My theoretical approach of considering management costs as a measure of stewardship costs might be applicable in the design of other conservation payment programs. Conservation easements that mandate management requirements for properties but do not limit land uses such as grazing are structurally similar to the stewardship payment program considered here. Such easements currently occur in the United States (Rissman 2010) and Canada are likely to be a viable conservation option in many parts of the world. My study is an important step in understanding the costs associated with such programs and their potential financial efficiencies compared to purchase and long-term management of additional national parks.

Chapter 7 Estimating landholders probability of participating in a stewardship program and implications for spatial conservation priorities[1]

The need to integrate social and economic factors into conservation planning has become a focus of academic discussions and has important practical implications for the implementation of conservation areas, both private and public. I conducted a survey in the Daly Catchment, Northern Territory, to inform the design and implementation of a stewardship payment program. I used a choice model to estimate the likely level of participation in two legal arrangements - conservation covenants and management agreements - based on payment level and proportion of properties required to be managed. I then spatially predicted landholders' probability of participating at the resolution of individual properties and incorporated these predictions into conservation planning software to examine the potential for the stewardship program to meet conservation objectives. I find that there was a tension between planning for a cost-effective program and planning for a program that targets properties with the highest probability of participation.



[1] Adams, V. M., R. L. Pressey, and N. Stoeckl. In Review. Estimating landholders' probability of participating in a stewardship program and implications for spatial conservation priorities. Biological Conservation.

Introduction

As discussed in Chapter 6, private land conservation is becoming more prominent and important as expansion of strict protected areas is increasingly constrained by reduced availability of land, insufficient budgets for acquisition, and escalating management costs of small, isolated reserves (Bruner et al. 2004; James et al. 1999a; Joppa et al. 2008). Longstanding conservation programs on private land include the US Conservation Reserve Program (Claassen et al. 2008), the UK Environmentally Sensitive Areas (ESA) program (Dobbs & Pretty 2008) and, in Australia, the Victorian Bush Tender Program (Stoneham et al. 2003).

Farmers, Indigenous owners and other private landholders manage approximately 77% of Australia's land area. This statistic alone indicates that conservation on private land is integral to Australia's biodiversity conservation strategy (Commonwealth of Australia 2008). All Australian states and territories have legislation for conservation covenanting on private properties, although some state programs are longer established and cover larger areas than others (Fitzsimons & Wescott 2004). Several states have competitive tendering for conservation contracts including the Victorian Bush Tender Program (Stoneham et al. 2003), the New South Wales Environmental Services Scheme, and the Queensland Nature Assist program.

Systematic conservation planning has been applied in one trial tender program - the Western Australian Conservation Auction - in which assessment of the benefits offered by properties accounted for complementarity between bids (Hajkowicz et al. 2007). This process demonstrated the potential to integrate well developed auction processes with spatial planning to achieve configurations of private conservation areas that maximize the achievement of conservation objectives within budgets. The ecological outcomes of a program might also depend on aspects of spatial configuration of the properties selected (Lombard et al. 2010; Nicholson et al. 2006; Rouget et al. 2006), but these can also be considered in protection and restoration of private lands (Bryan & Crossman 2008; Seddon et al. 2010).

Even with emerging capabilities in spatial analysis, the potential benefits of covenanting schemes can be difficult to achieve if landholders are unwilling to participate. Therefore, understanding landholders' willingness to participate has two important implications for private land conservation. First, this understanding will shape policy for the design of incentives.

Second, it is vital in identifying areas that are both valuable for achieving objectives and feasible for conservation action (for a protected area example see Guerrero et al. 2010). Knight et al. (2010) examined willingness to sell land for national parks and identified related aspects of social and human capital. However, other factors specific to program design, such as proposed land management, constraints on land title, and delivery of incentives, will also influence willingness to participate.

Of the Australian states and territories, the Northern Territory's policies and funding for conservation on private lands are the least developed, with financial support for conservation covenants and management agreements under consideration. Therefore, I undertook a pilot study in the Daly Catchment to assess the potential for such programs to meet conservation objectives. The program under consideration is for stewardship payments to leverage routine land management to meet conservation objectives on private lands. The program would include covenants, which are perpetual titles on private land, as well as management agreements, which are long-term legal agreements between the government and landholders. In Chapter 6 I examined aspects of designing the program such as costs and payment structures and, reported here, landholders' willingness to participate.

To assess landholder willingness, I used a choice experiment to estimate the probability of participation in the program relative to payment amount and required change in proportion of property managed for conservation. Choice modelling can estimate the effects of combinations of factors on participants' choices and is therefore more useful for designing policies than estimation techniques, such as willingness to pay, that consider only financial effects (e.g. Horne et al. 2005; Jacobsen & Thorsen 2010). My study had three aims.

The first aim was to use the results of my choice experiment to inform the design of a stewardship payment program in the Northern Territory. The choice model allowed me to estimate the expected level of participation, which can indicate the viability of the program and provide guidelines to the government about adequate budgets to meet desired participation levels. The choice model also allowed me to examine landholder preferences for the two mechanisms presented (covenants and management agreements) and how these preferences

varied with respect to payment amount and required change in proportion of property to be managed for conservation.

My second aim was to spatially predict landholders' willingness to participate at the resolution of individual properties and incorporate these predictions into conservation planning software to examine the potential for the stewardship program to meet conservation objectives. Predicting willingness to participate for individual properties allowed me to consider the potential spatial distribution of participating properties and therefore the likely conservation outcomes. This is an important aspect of understanding whether a stewardship program would have the desired impacts of achieving adequate protection for spatially variable conservation features.

The third aim of my study was to analyze how the interactions between willingness to participate and conservation costs can influence solutions identified in spatial conservation planning. My study is the first to incorporate both spatially variable willingness to participate and spatially variable costs. Therefore, this is the first study to elucidate how these two components of the planning problem interact and potentially constrain the capacity to meet conservation objectives.

Methods

Choice modeling experiment and survey methods

The study area was the whole of the Daly River catchment in the Northern Territory, buffered by 10 km (as discussed in Chapter 6). The Northern Territory is the only Australian jurisdiction without well established covenant and conservation management agreements. Therefore, I used the structure of the Queensland Nature Refuge program, which supports establishment of covenants on freehold and leasehold land, as the basis for designing my survey questions. The state of Queensland has more land under covenant (referred to as Nature Refuges) than any other Australian jurisdiction (Adams & Moon In Review) and has recently implemented legislation, called the Delbessie Agreement, to encourage participation in the program by lessees (DERM 2007). Under the Delbessie Agreement, lessees with properties identified as having conservation value can enter into a Nature Refuge agreement and be rewarded with a 10-year lease extension. Alternatively, they can elect to have their properties acquired. I designed my survey based on the assumption that the Northern Territory would model their covenant program on Queensland's and that similar legislation would be considered to support the environmentally sustainable,

productive use of rural leasehold land combined with covenants and management agreements. The survey included questions about the characteristics of landholders and properties, current expenditures on land management and conservation management, and other information specific to the choice experiments.

For the choice experiment, respondents were asked to consider the hypothetical scenario of a stewardship program with three alternatives for landholders: conservation covenant, conservation management agreement, or sell property. Choice experiments typically include a status-quo or default option. In my design, I did not include an 'opt-out' option because I wanted mirror legislation similar to the Delbessie Agreement. In that legislation, 'sell property' could be considered the opt-out or status quo because this is the only option for landholders unwilling to place portions of their properties under covenant. Not all on-farm conservation programs have similar 'conserve or sell' clauses, so the results of this experiment are not transferrable to those situations. Indeed, the probabilities estimated here will exceed those likely to be obtained in situations where neither sale nor participation is necessary. My results are therefore optimistic estimates of environmental outcomes from a stewardship program. I would expect larger shortfalls in meeting conservation objectives with the default option of not participating.

I hypothesized that willingness to participate in a program would depend on the type of agreement (covenant or management agreement), the proportion of property already set aside for conservation, the additional proportion of property to be set aside for the program, and the financial payment relative to additional costs of conservation management. In a pilot study, I tested different variables to reflect these factors that were cognitively accessible to respondents. Based on the pilot study, I represented financial payment as a percentage of stewardship costs because additional costs of conservation management will vary with current management activities and characteristics of properties, including size. I assumed that financial payments would range from 0% to 150% of additional costs, and used incremental amounts across that range to allow interpolation between points in my model (Figure 7.1a). I represented the required change in proportion of property set aside for conservation with five representative combinations identified from the pilot study (Figure 7.1a). I constructed the choice sets using a full factorial design, resulting in 80 different combinations (4 covenant payments x 4 management agreement payments x 5 changes in proportion of property set aside). Respondents

were given a set of definitions for alternative stewardship arrangements or sale of property using an information box and then asked to choose the preferred option in each choice set (example in Figure 7.1b). The choice sets were blocked into 8 versions of the choice experiment, each containing ten choice sets, which I estimated was the maximum number of sets for respondents to resolve before becoming fatigued. Each participant was randomly assigned a block of ten choice sets and I ensured that responses were representative across the 80 choice sets.

For my survey, I considered only land parcels of 10 ha or larger and outside the town of Katherine. Properties in the town or smaller than 10 ha are probably not good candidates for conservation agreements because they are predominantly residential and unmanaged. I sent surveys to all landholders eligible for private land stewardship agreements, defined here as all 440 pastoralist landholders (see Chapter 6 for more details). I used the Dillman tailored design method (Dillman 2007). Of the 440 landholders contacted, 25 requested to be removed from the survey and 50 addresses were no longer active, leaving a total of 365 possible respondents. The response rate to the survey was in line with similar surveys in the region (Zander et al. 2010; Zander & Straton 2010): about 25% (92 of 365 owners), with a total of 710 choice sets completed.

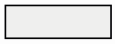
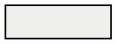
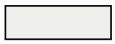
Choice experiment analysis

I analyzed the choice sets using a conditional mixed-effects logit model in STATA version 9. Based on the choice experiment, the probability of an individual *i* choosing an alternative *m* is given by

$$P(y_i = m|x_i, z_i) = \frac{exp^{(z_{im}\gamma + x_i\beta_m)}}{\sum_j exp^{(z_{ij}\gamma + x_i\beta_j)}}$$

where alternative specific variables for individual *i* for alternative *m* are given by z_{im} and coefficients are denoted by γ, case-specific variables for individual *i* are given by x_i, and coefficients are denoted by β. In my choice experiment, conservation payments were alternative-specific while conservation configuration was case-specific and landholder-specific variables were included as case-specific variables.

A

Attribute	Alternative: Conservation Covenant	Alternative: Conservation Management Agreement	Alternative: Sell
Payment level (as a % of stewardship costs)	0% 50% 100% 150%	0% 50% 100% 150%	Market value
Configuration (1)	*From:*  where there are no patches set aside for conservation	*To:*  where there is one small patch set aside for conservation	
(2)	*From:*  where there are no patches set aside for conservation	*To:*  where there are several patches set aside for conservation	
(3)	*From:*  where there are no patches set aside for conservation	*To:*  where there is one large continuous patch set aside for conservation	
(4)	*From:*  where there is one small patch set aside for conservation	*To:*  where there are several patches set aside for conserva tion	
(5)	*From:*  where there are several patches set aside for conservation	*To:*  where there is one large continuous patch set aside for conservation	

B

Imagine that the government would like you to set aside two more 'patches' of land on your property for conservation purposes. This would change the configuration of your land

From:  **where there is one small patch set aside for conservation**	***To:***  **where there are several patches set aside for conservation**	
Would you choose to	**Payment (as a % of Total Costs)**	**Choice** ✓
Accept a **Conservation covenant**, that would require you to pay for the survey costs, and then spend 1-2 days per month 'managing' the extra conservation areas	*and* receive compensation for **50%** of all costs	☐
Or Accept a **Conservation management agreement**, that would require you to purchase some extra supplies (e.g. fencing) and labor (to put the fences in) and would require you to spend an extra 1-2 days per month 'managing' the areas	*and* receive compensation for **100%** of all costs	☐
Or **Sell your entire property at market value**	Market value	☐

Figure 7.1 The choice experiment details. (a) Attribute levels for the choice experiment. The choice provided respondents with three alternatives to select from: conservation covenant, conservation management agreement, or sell property. The choice experiment considered two attributes that might influence respondents' choices: payment level as a percentage of stewardship costs (defined here as the additional costs of managing land for conservation, over and above routine property management) and change in extent and configuration of conservation management, defined relative to current configuration (from) and future configuration (to). I considered four payment levels and five changes in configuration. (b) Example choice set presented to respondents in survey.

I included two landholder-specific variables in my model which were identified as important predictors of land and conservation management costs in Chapter 6: size of property (*ln(property size, ha)*); and a binary flag indicating whether the landholder was currently engaged in conservation management (*conservation flag*). Ideally, I would have also tested whether sale values of properties influenced landholders' choices to sell, but sales data were not reliably available for the region.

Application of choice model

I used my final choice model for two purposes. First, I explored how probability of participation was affected by different payment levels, to understand how to maximize participation. Using the survey sample averages, I estimated the catchment-wide average probability of participation in covenants and management agreements based on three payment scenarios: 50% for both conservation covenant and conservation management agreement, 100% for both conservation covenant and conservation management agreement, and 150% for conservation covenant and 100% for conservation management agreement. For these scenarios, I assumed configuration 2 in which no patches are currently set aside for conservation and that landholders would be required to set aside several small patches for conservation in the future. The second use of the choice model was to estimate the probability of participation of each property so that my planning scenarios could preferentially select properties with higher probabilities of participation. For each property, I estimated probability of participation assuming payments of 150% for covenants and 100% for management agreements and configuration 2 (no patches currently set aside for conservation and several patches to be set aside for conservation in the future).

Planning assumptions and scenarios

Based on my survey and discussions with the Northern Territory government, I identified three realistic parameters of a stewardship program. First, the government would pay a premium to engage landholders in conservation covenants in preference to management agreements because of the perceived benefits of permanent title for conservation. Second, most landholders are currently not managing any areas for conservation, over and above routine property management. Third, landholders participating in the stewardship program would be required to manage several small patches on their property for conservation. I used the estimated probability

of participation for each property (above), which reflected these assumptions, and calculated costs assuming payments of 150% for covenants and 100% for management agreements.

In addition to engaging private landholders, the government is interested in funding new Indigenous Protected Areas. These are agreements between traditional owners and the Australian Government, considered to be similar to national parks. Funding both Indigenous Protected Areas (IPAs) and a stewardship program would equitably provide opportunities for Daly residents to access financial support for conservation management. I assumed an annual budget of $1.5 million to fund Indigenous Protected Areas and stewardship agreements in the catchment. This figure was based on the non-spatial financial estimate of $1 million required for stewardship agreements across about 4.2 million ha of pastoral properties (details in Chapter 6) and a pro-rated estimate of $0.5 million for IPAs over about 2 million ha of Indigenous land.

The Northern Territory has clearing laws for the Daly that dictate what types of vegetation and the percentage of properties and the whole catchment that can be cleared (NRETAS 2010). Although some areas are thereby protected from clearing, their conservation features might not be secure without management to reduce threats such as invasive weeds and altered fire regimes. Therefore, I considered spatial planning scenarios in which these areas were effectively secured by legislation and other scenarios in which they were unprotected and available for management by Indigenous Protected Areas or stewardship agreements.

Spatial planning using Marxan with Zones

I used Marxan with Zones (Watts et al. 2009) to examine possible spatial configurations of Indigenous Protected Areas and stewardship agreements, and the number of conservation objectives met across the catchment.

I divided all properties in the catchment, including pastoral and indigenous, into planning units of 25 ha square grids. For each private property, I calculated the cost of planning units as pro rata proportions of the expected cost of conservation management. For properties under consideration for Indigenous Protected Areas, I used an expected conservation management cost per ha of $2.25 (NRETAS 2009b). I calculated the expected cost per ha of stewardship payments as:

$$E(C) = prob_{cma}c_{cma} + prob_{cc}c_{cc}$$

where $prob_{cma}$ is the probability of landholder i selecting a conservation management agreement given participation in the program ($prob_{cma}/(1\text{-}prob_{sell})$), $prob_{cc}$ is the probability of landholder i selecting a conservation covenant given participation in the program ($prob_{cc}/(1\text{-}prob_{sell})$), c_{cma} is the cost of stewardship payment to landholder i based on property scale cost estimates from Chapter 6, and c_{cc} is 150% of c_{cma}. Marxan with Zones minimizes the total cost of the zoning plan C:

$$C = \sum_{i=1}^{M} \sum_{j=1}^{N} c_{ij} x_{ij},$$

where x_{ij}=1 if the ith planning unit is included in the jth zone, subject to the constraint that a planning unit can only be placed in one zone. I ran Marxan with Zones to achieve objectives within a constrained budget of $1.5 million.

In total I considered six Marxan with Zones scenarios to reflect variable contributions of vegetation protected by legislation, spatially variable costs, and preferential selection of high probability properties (Table 7.1). To examine the capacity of a stewardship program to meet spatial conservation objectives, I selected properties that were most likely to participate in a stewardship program. I did this by including estimated probability of participation as a conservation feature for each pastoral property and set a catchment-wide objective of 15% of the total probability (scenarios 3 and 6) (which is computationally similar to the approach used by Guerrero et al. 2010). The 15% objective was selected to reflect the findings in chapter 6 for the percentage of properties that were cost-efficient (defined as those with per ha costs of conservation management less than those of national parks). This approach allowed me to select the properties that were most likely to participate while considering spatial conservation objectives within the budget I set for the stewardship program. Importantly, this approach also allowed probability of participation to be separated from management cost in the software analyses.

For all six scenarios, I had quantitative objectives for 71 conservation features: mapped vegetation types (62), rainforest and riparian zones (4) and sites of conservation significance (5 in the Daly, NRETAS 2009b). My objectives were 30% of the current extent of each vegetation (as pre-clearing data was not available), and, based on discussions with NRETAS, 100% for rainforest, riparian zones, and sites of conservation significance. I assumed that the different

zones contributed differentially to conservation objectives (Table 7.1), reflecting commitment of management to conservation.

Table 7.1 Marxan with Zones scenarios. I defined scenarios in terms of zones considered, proportional contribution of zones to conservation objectives, costs of management in Indigenous Protected Areas (IPAs), and costs of the stewardship (private pastoral) zone.

Scenario	**Zones included (proportional contribution of zones to objectives in parentheses)**	**Cost IPA**	**Cost stewardship**
Scenario 1 - Uniform costs	1 – Available (0) [a] 2 – National Park (1) 3 – IPA (1) 4 – Stewardship (0.7)	Area	Area
Scenario 2 - Variable costs	As above	$2.25 per ha	Estimated expected stewardship costs per ha
Scenario 3 - Variable costs + probabilities	As above	$2.25 per ha	Estimated expected stewardship costs per ha
Scenario 4 - With Never Clear [b], Uniform Cost	1 – Available (0) 2 – National Park (1) 3 – IPA (1) 4 – Stewardship (0.7) 5 – Never Clear (0.7) [b]	Area	Area
Scenario 5 - With Never Clear [b], variable costs	As above	$2.25 per ha	Estimated expected stewardship costs per ha
Scenario 6 - With Never Clear [b] , variable costs + probabilities	As above	$2.25 per ha	Estimated expected stewardship costs per ha

[a] Currently not managed for conservation but available for management either with IPA or stewardship
[b] Areas covered by legislation that prevents clearing, assuming that this legislation is fully effective for conservation

I assumed a single time step in which areas were engaged for stewardship or IPAs and that engagement and conservation management would continue. To control the aggregation of selected areas (Watts et al. 2009), I identified the zone boundary cost for each scenario with the method of Stewart & Possingham (2005). I ran Marxan with Zones with 100 runs for each scenario and recorded best solutions and selection frequency for each scenario. I compared the spatial similarity of solutions for each scenario by correlating (Spearman rank ρ) selection frequencies of planning units.

Analyzing the relationship between transaction costs and acquisition costs and its implications for spatial selections

Including landholder willingness to sell for conservation or to otherwise participate in conservation actions is motivated by the need to improve implementation of conservation plans (Knight & Cowling 2007; Knight et al. 2010). Willingness to participate can therefore be seen as a surrogate for the transaction costs of engaging with landholders. I assume here that transaction costs are inversely related to willingness of property owners to participate in a stewardship program, and I express willingness as probability of participation. However, the spatial distribution of landholders willing to participate might not coincide with all conservation features. Willingness might also be unrelated spatially to conservation costs such that working with willing landholders might not allow planners to meet all their conservation objectives within a constrained budget.

To test the potential interactions between transaction costs and acquisition costs, which may be the costs of land purchase or in this case the acquisition cost of purchasing stewardship agreements, I constructed two hypothetical datasets to reflect observed trends: 1. assuming properties or planning units of relatively uniform size and uniform per ha acquisition, with normally distributed transaction costs (inversely related to probability of participation); and 2. assuming highly variable property sizes and strong economies of scale for per ha costs of acquisition, with transaction costs (inversely related to probability of participation) positively correlated with property size. The first data type reflects that used by Guerrero et al. (2010) and Knight et al. (2011) which considered purchase of land for protection. The second data type reflects the data found in this study of the Daly River catchment which considers purchase of stewardship agreements.

I considered two hypothetical data sets, each consisting of 150 properties. I estimated transaction costs as a function of probability of participation using the exponential distribution with $\lambda=1$ (given by probability=$1*e^{-1*transaction}$) resulting in transaction cost per ha being equal to –ln(probability). I then compared transaction costs to hypothetical costs of acquisition and total costs (acquisition + transaction). I ordered properties from lowest to highest total cost per ha, indicating a notional cost-efficient order of selection for conservation action. I assumed, for simplicity, that conservation features were uniformly distributed across properties and, therefore, assumed that a 30% representation objective for each feature would be achieved when 30% of the total property area was protected.

Results

The final conditional mixed-effects logit model for the choice experiment is in Table 7.2. The two landholder-specific variables - *ln(property size, ha)* and *conservation flag* - were significant ($p<0.001$). The coefficient for property size was negative, indicating that owners of larger properties were less likely to participate. The coefficient for conservation flag was positive, indicating that owners already engaged in conservation management were more likely to participate. The coefficients for configuration levels were negative, and increasingly so with the extent of change in required proportion of property to be managed for conservation. Accordingly, configuration 3, requiring landholders to change from no patches to one large continuous patch set aside for conservation, had the largest negative coefficient. This trend was similar for both covenants and management agreements. However, the coefficients for covenant configurations were more strongly negative, indicating that landholders were less likely to select a covenant than a management agreement. The positive coefficient for payment level indicates that probability of participation increased with payment level.

Table 7.2 Conditional mixed-effects logit model. CC indicates conservation covenant; CMA indicates conservation management agreement. Configuration was coded as a set of dummy variables (corresponding to alternative changes in configuration in Figure 7.1) with configuration 1 chosen as the status quo. *p<0.05, **p<0.005, *** p<0.001

Variable	Coefficient		SE
CC intercept	0.3704		0.3861
CMA intercept	0.3788		0.3492
Payment	0.0133	***	0.0012
Configuration 2, CC	-1.1400	***	0.3420
Configuration 3, CC	-1.3841	***	0.3562
Configuration 4, CC	-1.0396	**	0.3442
Configuration 5, CC	-1.1116	***	0.3336
Configuration 2, CMA	-0.6737	**	0.3123
Configuration 3, CMA	-1.0710	***	0.3216
Configuration 4, CMA	-0.4958	*	0.3091
Configuration 5, CMA	-0.8896	**	0.3162
Conservation flag, CC	2.2508	***	0.2935
Conservation flag, CMA	1.3770	***	0.2625
ln(property size), CC	-0.6335	***	0.1230
ln(property size), CMA	-0.4577	***	0.1004
N (Choice sets)	710		
Log L	-654.32		
rho2	0.16		

For my three payment scenarios, the predicted probabilities of participation in stewardship arrangements increased from 42% to 64% as payment levels increased (Table 7.3). Respondents always preferred conservation management agreements to covenants. However, the payment premium for covenants substantially increased the probability of participating through a covenant (29% for 150% payment, 18% for 100% payment, Table 7.3). The design of my choice experiment, lacking an alternative for 'opting-out' of negotiations without selling, would probably have produced absolute probabilities of participating higher than if an 'opt-out' choice had been included. However, if the design of the stewardship program reflects the constraints of the Queensland program coupled with the Delbessie agreement I expect that my probability of participation is reliable. In addition, I expect that the relative probabilities are reliably indicative of the preferences of landholders.

Table 7.3 Estimated probabilities of participation for three payment scenarios. CC indicates conservation covenant; CMA indicates conservation management agreement.

	Payment scenarios		
	50% CC, 50% CMA	**100% CC, 100% CMA**	**150% CC, 100% CMA**
Conservation Covenant (CC)	0.13	0.18	0.29
Conservation Management Agreement (CMA)	0.29	0.40	0.35
Sell Property	0.58	0.42	0.36

For the annual budget of $1.5 million to support management of Indigenous Protected Areas and stewardship agreements, not all conservation objectives could be met (Table 7.4) and the shortfalls were consistent for all scenarios (5-9 objectives not met). In all cases, shortfalls were for 100% objectives for rainforest vegetation and sites of conservation significance. By including the legislatively protected vegetation as fully protected, 1.1 million ha was added to initial conservation management. Forty-six of the 71 objectives were met in these legislated areas and national park zones.

Table 7.4 Summary results from Marxan with Zones for the six scenarios, including number of objectives met, area selected for IPAs, stewardship and other legislated protected areas (National Parks and Never Clear [a]), percentage of pastoral properties engaged in stewardship, total cost, and percent of total probability of participation.

	Objectives met (out of 71)	Area (ha) - IPA	Area (ha) - stewardship	Percent properties engaged in stewardship	Area (ha) - National Park and Never Clear [a]	Cost ($ million)	Percent total probability
Scenario 1	62	375,458	824,548	46.36%	635,591	2.3	17.83%
Scenario 2	64	263,340	1,137,863	17.73%	635,591	1.5	14.44%
Scenario 3	64	216,887	1,172,881	20.68%	635,591	1.5	16.05%
Scenario 4	66	386,888	813,112	45.00%	2,731,281	2.5	17.54%
Scenario 5	62	261,037	796,172	23.18%	2,731,281	1.5	13.04%
Scenario 6	62	158,287	976,623	42.27%	2,731,281	1.5	16.18%

[a] Areas covered by legislation that prevents clearing, assuming that this legislation is effective for conservation

Scenarios with uniform costs (1 and 4, Table 7.1) met the most objectives with similar total areas selected for Indigenous Protected Areas and stewardship agreements. However, the total costs of these scenarios were 50% greater than scenarios informed by variable costs (Table 7.4). Including variable costs reduced the number of properties engaged in stewardship agreements by selecting larger properties, a consequence of strong economies of scale for stewardship costs. Including variable costs also lowered the probability of participation because landholders on larger properties were less likely to participate. Adding the objective for probability of participation to consideration of variable costs increased overall probability of participation, but not to the level achieved by scenarios 1 and 4 which had the highest overall probability of participation because they ignored costs. This reflects a tension between two key considerations in the Daly: cost-effectiveness requires that larger properties are selected, but overall probability of participation is thereby lowered.

Selection frequencies for stewardship agreements were positively correlated for all scenarios (Table 7.5), although significant values are unsurprising given the number of planning units (n=327,417). Scenarios without the "Never Clear" legislated areas locked in (1-3) were more strongly correlated to one another than to the other three (ρ = 0.298-0.457). Similarly, scenarios with "Never Clear" areas locked in were more strongly correlated to one another than to the other three (ρ = 0.359-0.469). The maximum ρ value of 0.469 and other much lower coefficients indicated substantial spatial variation between all pairs of scenarios, with all factors defining scenarios leading to spatial differences.

Table 7.5 Spearman's rank correlation, ρ, between selection frequencies for stewardship agreements in pairs of scenarios ($p<0.001$ for all values).

	Scenario 1	**Scenario 2**	**Scenario 3**	**Scenario 4**	**Scenario 5**
Scenario 2	0.298				
Scenario 3	0.304	0.457			
Scenario 4	0.149	0.197	0.197		
Scenario 5	0.138	0.259	0.235	0.359	
Scenario 6	0.149	0.306	0.293	0.430	0.469

Intuitively, probability of participation (willingness) will be inversely related to transaction costs, in that properties targeted by agencies for participation will involve less engagement, perhaps both before and after stewardship agreements have been finalized, if landholders are more willing to participate. In the Daly, I therefore expect that properties with lower probabilities of participation will have higher transaction costs. In the case of uniform per-unit-area costs of acquisition (used by Guerrero et al. 2010 and Knight et al. 2011), the variable transaction costs should drive total cost, and the cost-efficient sequence of properties selected for conservation action would begin where probability of participation is high (Figure 7.2a). In contrast, in the Daly basin, where acquisition costs (given by costs of conservation management associated with stewardship agreements) are variable with strong economies of scale and willingness is inversely related to property size, total costs will be driven by acquisition costs. In this case, the cost-efficient sequence of selection of properties for conservation action begins with the largest properties where willingness is lowest and transaction costs are highest (Figure 7.2b).

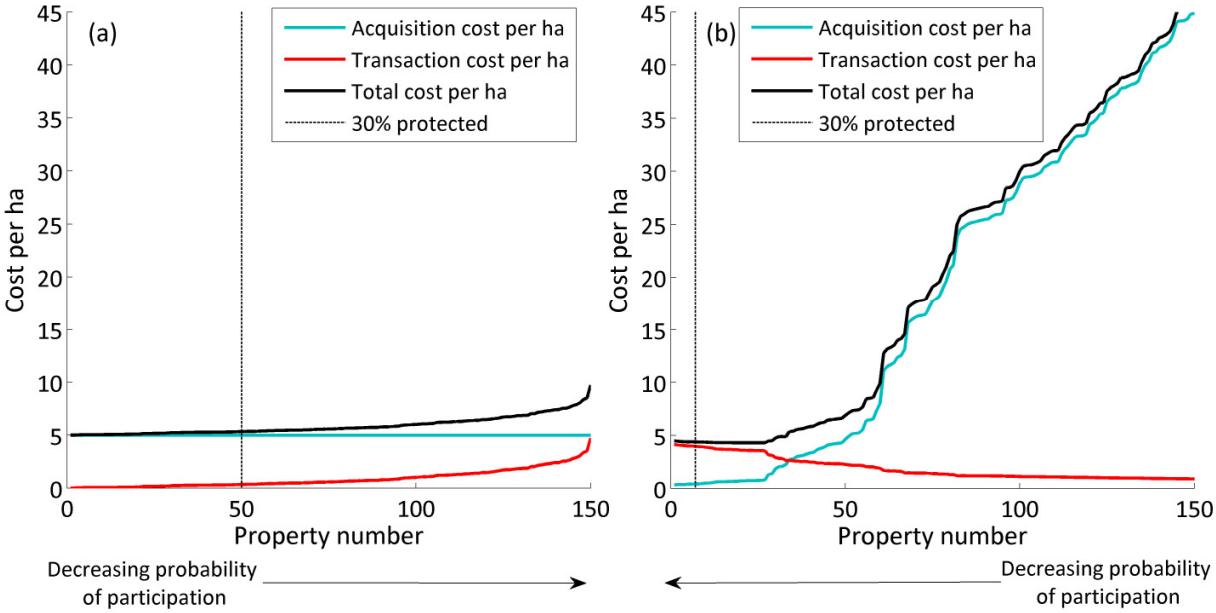


Figure 7.2 Per-unit-area costs of acquisition costs, transaction costs, and total costs for two hypothetical data sets consisting of 150 properties. Acquisition costs could refer to land acquisition, purchase of stewardship agreements, or purchase of other conservation agreements such as payment for ecosystem services. Transaction costs refer to the costs of engagement, assumed to be inversely related to willingness of property owners to participate in the conservation program (probability of participation). In both graphs, properties are ordered from lowest to highest total cost per ha (acquisition costs + transaction costs), indicating a notional cost-efficient order of selection for conservation action. The vertical line in each graph indicates the point at which 30% of total property area would be protected. (a) The type of data used by Guerrero et al. (2010) and Knight et al. (2011) in which land was targeted for purchase, assuming properties of relatively uniform size (~500 ha), uniform costs per ha of conservation management for all properties, and normally distributed probabilities of participation. (b) The type of data found in my study of the Daly River catchment where land is targeted for purchase of a stewardship agreement. Property sizes vary by 5 orders of magnitude, there are strong economies of scale for conservation management, and probability of participation is inversely related to property size.

Discussion

Choice modelling has rarely been applied to management of protected areas or design of conservation incentives (but see Horne et al. 2005; Jacobsen & Thorsen 2010). My choice analysis provides several insights for designing and implementing a stewardship program in the Northern Territory. I estimated that a large percentage of landholders – between 42% to 64% depending on payment levels - would be willing to participate. I found that landholders were financially motivated in their preferences between conservation management agreements and conservation covenants. All else being equal, landholders preferred management agreements, reflecting their reported concerns over the title implications of covenants and potential negative effects on sale values. However, this preference can apparently be weakened with a payment premium for covenants. Covenants have benefits for the government. The first is the security of

permanent titling (Fitzsimons & Wescott 2004). Second, titling allows covenants to be classified as IUCN recognized protected areas (Class VI in the case of Nature Refuges, however private protected areas may qualify for all classes, Fitzsimons 2006) which then contribute to national conservation goals such as the 2020 17% target under the Convention on Biological Diversity (UNEP 2010).

The stewardship payment model developed by Adams et al (In Review) found strong economies of scale with the largest properties being the most cost-efficient. However, in my choice model, the negative coefficient associated with *ln(property size, ha)* indicated that the most cost-efficient properties were also the least likely to participate. My spatial zonings supported the findings of Chapter 6 that the most cost-efficient implementation of the stewardship program would be to engage the largest properties, but my zoning also demonstrated that the budget level of $1.5 million per annum was insufficient for all conservation objectives to be met. Furthermore, if the stewardship program were implemented as a closed-bid auction, probably even fewer conservation objectives would be met because landholders on the most cost-efficient properties would be less likely to submit bids. Rather, the more willing participants would be more likely to have smaller properties that are more costly to manage per ha, and a larger budget would therefore be needed to meet conservation objectives while engaging these landholders. An alternative strategy might be for the government to fund an outreach campaign to increase the probability of larger properties participating.

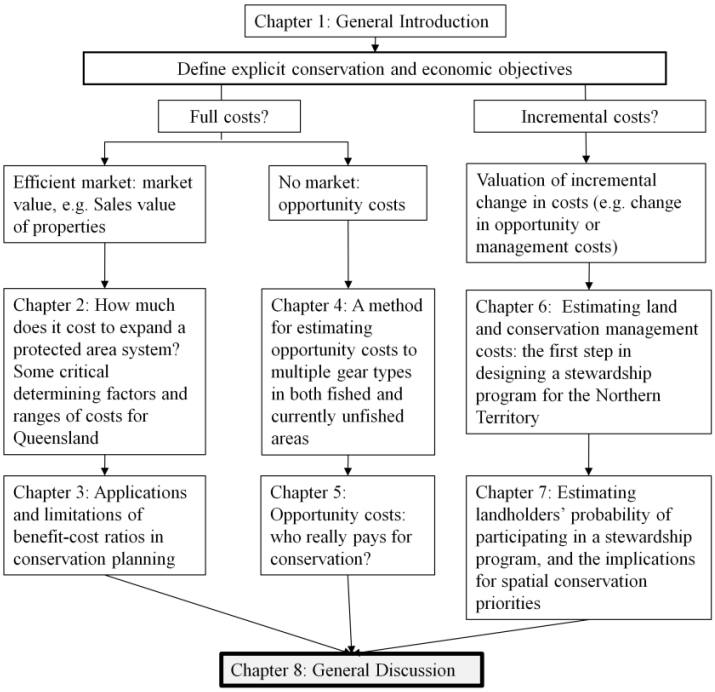
My choice experiment sought to mimic a key characteristic of the Delbessie Agreement by explicitly not offering landholders a choice to 'opt-out'. I believe, however, that my experimental design would not have exaggerated one of my key conclusions: that property size was inversely related to probability of participation, and therefore created a tension between selecting properties that are cost efficient and selecting properties with landholders who are willing to participate. My design would only exaggerate the negative association between property size and probability of participating if probability of selling and property size were positively related, that is, if owners of larger properties were more likely to sell than those of smaller properties. In that case, having the option to sell rather than to engage in stewardship would be more appealing to owners of larger properties. However, I found that property size and number of years of ownership, admittedly an imprecise proxy for propensity to sell, were

uncorrelated. I conclude that it is unlikely that an alternative experimental design would have changed my observed negative association between property size and probability of participation.

Variation in landholders' willingness to participate has been considered in two other spatial prioritizations (Guerrero et al. 2010; Knight et al. 2011). These studies included costs (unrealistically) as an average sales price. In contrast, by selecting areas with data on both variable costs and variable willingness, my study demonstrated that, with a constrained budget, spatially variable costs can be more important than willingness in determining conservation priorities. When I considered probability of participation (willingness) to be an inverse proxy for transaction costs, then total costs in the study of Guerrero et al. (2010) are determined by transaction costs, and the most cost-efficient properties are those with the most willing landholders (Figure 7.2a). In my study area, with strong economies of scale and willingness to participate inversely related to property size, I identified a trade-off between cost-efficiency and willingness to participate (Figure 7.2b). This is likely to be the case more generally, where economies of scale apply to costs such as those of acquisition and management (e.g. Armsworth et al. 2011; Ban et al. 2011; McCrea-Strub et al. 2010). This result provides an important insight into the potential interactions between the spatial distribution of conservation features, costs of conservation, and willingness of landholders to engage in conservation. These interactions will be important to consider for future studies concerned with opportunities for and constraints on implementation. My analysis highlights important design and policy issues associated with implementing a stewardship program in the Northern Territory and other parts of the world. If planners understand the spatial drivers of both costs and probability of participation, then trade-offs can be addressed pro-actively with engagement strategies or arguments for adequate budgets.

Chapter 8 General Discussion

In this chapter I summarize the key findings of my thesis and discuss how my thesis contributes to the scientific basis for incorporating costs of conservation, given explicit socio-economic goals, into the systematic conservation planning process.



Socio-economic costs and conservation planning

As discussed in Chapter 1, there have been calls in the literature to include costs in conservation planning in the belief that the plans that result will be more affordable and efficient, and will also better reflect the socio-economic context of planning regions to make plans more implementable (Naidoo et al. 2006; Polasky 2008).

Incorporating costs into conservation planning requires stating explicit economic objectives (Chapters 2-7). It also requires an understanding of the appropriate conservation actions to meet these objectives so that the right costs can be estimated. While many studies have included costs in the planning process, relatively few have followed the above steps of stating explicit economic objectives and selecting the appropriate cost metrics to reflect these objectives and associated management actions. In addition to often not linking cost metrics to explicit objectives, most studies have not been based on rigorous economic methods.

The goal of this thesis was to contribute to the scientific basis for incorporating costs of conservation, given explicit socio-economic goals, into the systematic conservation planning process. To achieve this I did several things: (1) broadly categorized conservation planning exercises into three major types and identified the appropriate cost metrics; (2) developed methods appropriate for estimating the required cost metrics; and (3) investigated the impacts of uncertainty and socio-economic dimensions, such as preferences of landholders, on outputs of systematic conservation planning.

Thesis outcomes

Objective 1: Develop cost metrics for inclusion in conservation planning based on standard economic methodologies.

To develop cost metrics using economic methods, I first conducted a literature review to identify common contexts of conservation planning and the likely socio-economic goals and associated actions of these. Based on this review I separated conservation actions into two broad categories: 1. Areas protected by taking them out of production, such as establishing national parks or marine protected areas, and therefore the costs associated with this action are 'full' costs; 2. Areas not taken out of full production, with conservation management involving an incremental change in management or use, such as seasonal fishery closures or conservation

management agreements which specify changes in land use or land management, and also specify the incremental costs required. I then identified three conservation planning contexts that reflect these two broad categories and estimated the appropriate costs using standard economic methods.

The first conservation planning context that I identified is the most common application of conservation planning in which areas are selected for full protection in a fully developed land market, with cost of conservation being the acquisition cost of land at market. In Chapter 2 I used sales data from Queensland and applied a geographically weighted regression to estimate sales value for all properties in Queensland. Because sales data are patchy, many studies rely on simple extrapolation techniques or opt for land valuation data that are often available for all properties for tax purposes (Carwardine et al. 2008; Klein et al. 2009a). However, neither of these options provides a rigorous estimate of the actual costs of acquisition of a property. Therefore, the technique that I selected presents a more thorough statistical approach to estimating market value of all properties in a region. Using the estimated sales values of properties, I also calculated the expected acquisition cost as a function of the probability of a property following different legal routes of protection and the associated costs of each legal route. Lastly I included estimated transaction and management costs to select areas based on the total financial cost to the government for adding a property to the protected area estate. This is the first time in the conservation literature that the variable costs associated with different legal pathways for protection have been considered explicitly. This chapter also demonstrated the sensitivity of my cost estimates to the assumptions made about landholder participation in different legal pathways, highlighting the importance of narrowing the expected range of participation in different pathways.

The second conservation planning context that I examined is the case of selecting areas for full protection in developing regions where markets are not fully developed and therefore opportunity costs are a better measure of the full costs of conservation. In Chapter 4 I considered the planning region of Kubulau district, Vanua Levu, Fiji Islands where the community is considering extending or re-configuring the marine protected area network. The appropriate cost measure in this context is the opportunity costs to fishers from loss of fishing grounds. I therefore developed a method for estimating the opportunity costs to different types of fishers

based on gear type and extended the model to calculate profit as a function of gear and transport type. The method that I developed is mathematically analogous to the method developed by Naidoo & Adamowicz (2006) and provides an estimation technique that accounts for the costs of both fished and unfished areas such that the costs are not biased to current fishing patterns. The primary outcome of this chapter was a mathematical approach to estimating opportunity costs in the marine environment. In addition, I found that opportunity costs varied spatially between fishing gear types, highlighting the importance of explicitly considering costs to different stakeholder groups.

The third conservation planning context that I examined is the case of selecting areas, not for full protection, but for participation in incentive programs that might only partially reduce production or require a change in management regime. In this case the cost of conservation is given by the incremental change in either opportunity cost, if production is reduced, or management cost, if participation in the program specifies required conservation management actions. In Chapter 6 I estimated the costs associated with a stewardship program which would not restrict land use but would instead require specific conservation management action from landholders. Therefore, I estimated the costs as the incremental change in land management costs to meet conservation objectives.

Objective 2: Examine the impacts of socio-economic dimensions, such as accounting for tenure structures, stakeholder values and preferences, and differential costs to stakeholder groups, on conservation costs and decisions.

My literature review identified a key gap in the conservation literature: socio-economic dimensions are often neglected in the analysis of conservation costs. For example, there are often many stakeholder groups that are not only involved in the conservation planning process but impacted by the implementation of the conservation actions. Typically, however, cost metrics included in the conservation plan reflect only one group or are aggregated across several groups. In addition, the true costs of conservation can be different from the expected costs because of stakeholder values and preferences. Thus, accounting for conservation costs not only requires methodologies for identifying and measuring the appropriate costs but also understanding the socio-economic dimensions that influence the real costs of conservation as a plan is implemented.

I achieved this second objective by considering different socio-economic dimensions relevant to the three conservation planning contexts. In the first conservation planning context, I examined how stakeholder values can result in variable participation levels in conservation programs and therefore influence the financial costs of extending a protected area system. In Chapter 2, I included the different legal pathways for adding areas to the Queensland protected area estate, considering acquisition and covenants with private landholders which have different associated costs (full market value for acquisition by the government and no acquisition cost to the government for covenants). I found that the total estimated financial costs of extending the protected area are highly sensitive to the expected proportion of landholders willing to place a covenant on their land versus selling their property.

In the second planning context I found that opportunity costs can be spatially dissimilar for different stakeholder groups that use the land and sea in different ways. Therefore, in Chapters 4 and 5, I spatially mapped and compared the opportunity costs to each stakeholder group in each region. In Chapter 5, I calculated how including costs to only one stakeholder group impacts total opportunity costs as well as costs to other stakeholder groups. I demonstrated that socio-economic objectives should be stated explicitly and the opportunity costs that best represent these objectives should be included. For example, if the socio-economic objective is to minimize the impact on local communities who use the land for subsistence farming, then the appropriate cost would be opportunity costs to smallholder farmers. Including other cost metrics would not result in the desired socio-economic outcomes. To date several studies have included costs to multiple stakeholders (Klein et al. 2008a; Klein et al. 2009b); however, none have examined how inclusion of costs to different stakeholders may best represent explicit socio-economic objectives which have variable impacts on the stakeholder groups.

Stakeholder values and preferences can also influence the costs of conservation. In the context of designing a conservation incentive program for private landholders, preferences and values can cause the true costs of a program to deviate from the expected costs. For example, the expected costs of the stewardship program described in Chapter 6 would be the cost difference between routine land management and the additional requirements of conservation management. However, if landholders' perceive this incremental change to be a burden on their lifestyle or

ability to conduct required routine land management, they may expect a premium to be paid for their time and thus the true costs would be greater than the expected costs. In Chapter 7 I used a choice experiment to estimate the probability of landholders participating in a program based on variable payment levels. While the estimated costs of participating in a conservation covenant and conservation management agreement are identical, the choice experiment indicated that landholders perceived an additional cost associated with a covenant and only selected the covenant when a price premium would be paid. Therefore, if the government wishes to design a conservation covenant program, the actual costs will likely have to be inflated to reflect the required premium to encourage landholders to accept this approach.

Objective 3: Examine potential sources of uncertainty in cost estimates and assess the impacts of these uncertainties on financial budget estimates.

In addition to the difficulties of identifying the appropriate measures of conservation cost and estimating them, there are uncertainties associated with these estimates which to date have rarely been acknowledged. I addressed this gap in the literature by first identifying key sources of uncertainty and then applying two methods for assessing the impacts of these uncertainties: sensitivity analysis and info-gap theory. In Chapter 2 I used a sensitivity analysis to assess how uncertainties in three key factors impact financial estimates of the total costs of extending the Queensland protected area estate. These factors were: legal acquisition routes dependent on tenure; subdivision of properties; and size of biodiversity objectives. The sensitivity analysis highlighted that cost estimates are highly sensitive to assumptions made about the percentage of landholders willing to establish a nature refuge. This is very relevant in light of the untested Delbessie Agreement for conservation on leasehold land. In Chapter 3 I developed and applied an info-gap model to assess the robustness of sets of properties selected for acquisition to uncertainties in estimates of costs and benefits. The info-gap model demonstrated that cost-efficient budget allocations using benefit-cost ratios are more robust than simple benefit rankings. However, BCR prioritization results in uncertainties that are not explicitly considered in the basic implementation of the prioritization. Managers wishing to use BCR should consider potential shifts in prioritizations depending on what surrogate measures are used and how much certainty is involved in their estimation.

Evaluating the approach

This thesis used the approach of first stating explicit economic objectives and then identifying the appropriate cost metrics in three conservation planning contexts. This demonstrates the importance of this first crucial and often neglected step. In my thesis I chose to implement the conservation approach in which areas are selected for protection to meet conservation objectives at a minimum cost, but endeavoured to use full accountings of costs and benefits given the data available, to ensure that solutions would be equivalent to those obtained using an economic approach such as maximizing utility. If we have a complete accounting of the costs and benefits of a conservation policy, such as establishing a protected area, then the conservation approach of maximizing conservation benefits at a minimum cost should be equivalent to the welfare economic approach of maximizing the social welfare function. An important next step will be to consider similar questions using a social welfare function approach. Welfare economics allows the analysis of the utility derived by society as a function of the utility received by the individuals in the society and thus allows explicit consideration of individual value judgements or preferences with the use of social welfare functions. For example, recent work by Larson et al. (2010) developed a method for identifying facotrs taht contribute to individual wellbeing, including social, ecological and economic factors, and then measuring the relative contribution of these factors to overall wellbeing. This method can be extended to systematic conservation planning to construct a social welfare function to reflect the individual utility derived from the protection of natural resources as a function of social, economic and ecological costs and benefits (Brereton et al. 2008). The relative contributions of social welfare factors to overall wellbeing provides important insight into how explicit conservation objectives may or may not appropriately reflect the needs of the stakeholders and policy recommendations can be made to align objectives with key welfare factors. Importantly, the social welfare approach may be more appropriate for including multiple values such as variable costs to multiple stakeholders because it allows for consideration of individual welfare functions as well as overall welfare functions. It might also be more appropriate for non-market benefits associated with conservation which cannot be readily translated into dollar values. The systematic conservation approach is more restricted in including costs in single units, such as dollars.

Uncertainties associated with cost estimates

In Chapters 2 and 3 I examined the implications of uncertainties in cost estimates for financial budgets and decision making. The sensitivity analysis examined the important question of how uncertainty enters into cost estimates. However, I did not address what form of statistical distributions uncertainty values take. While the info-gap model presented in Chapter 3 provides a tool for decision makers to assess the robustness of their budget allocations when the statistical distribution of the uncertainty associated with cost estimates is not known, an important next step will be to estimate the distributions of uncertainty values. In the case of expanding protected area systems, the distribution might be estimated by comparing the actual costs of acquisition as land parcels are purchased to the estimated land values used to estimate financial budgets. Tracking the actual costs of expansion against the predicted costs would allow estimation of real uncertainties and the distribution of uncertainties could then be examined to see if they are normal and spatially independent. For example, uncertainty could be higher for land parcels near urban centers where market values fluctuate more or where conservation decisions influence sales values. The expansion of the Queensland protected area system, which is now underway, poses an ideal opportunity to track the actual costs and explore these issues in more depth.

In addition, I did not explicitly address the uncertainty in the cost estimates developed in Chapters 4 and 6. The opportunity costs modelled in Chapter 4 were compared to catch per unit effort (CPUE) data and I found that the modelled data were positively correlated with the CPUE data. This finding indicates that the modelled costs reflect the pattern of actual costs. However, to ground-truth these data and estimate the error associated with the model, it would be necessary to collect data on actual market value of catches across the region. A similar ground-truthing would be necessary to assess the error associated with the cost estimates from Chapter 6.

Limitations of the systematic conservation planning approach to addressing complex socio-economic objectives

The software available for systematic conservation planning is restricted in the availability of functions to include socio-economic costs. Those software packages that are able to include variable costs are based on algorithms that can maximize conservation benefits across multiple conservation features while minimizing across a single cost (Ball et al. 2009). With such limitations to the software, it is difficult to explicitly consider costs across stakeholder groups.

However, running multiple scenarios to include costs to each stakeholder group, as I did in Chapter 5, allows a better understanding of costs across groups. The recent extension to the conservation planning tool Marxan, called Marxan with Zones, allows consideration of multiple costs to different stakeholder groups (Watts et al. 2009). While I used Marxan with Zones to account for multiple types of conservation actions in the Daly (Chapter 7), the algorithm is still limited in its ability to account for multiple socio-economic dimensions. Presently, to apply this software and account for multiple stakeholders, a unique zone with specific costs and benefits must be defined for each group. Therefore, considerations of several groups quickly makes the processing time burdensome.

In addition, even the most recently developed tools, such as Marxan with Zones, do not allow for easy inclusion of stakeholder preferences or values. To include stakeholder willingness to participate in a stewardship program, I had to include probability of participation as a conservation feature and set a target for proportion of properties relative to the total probability of participation across the study area. While this did shift spatial selections to more likely properties, including such considerations is not a well detailed feature of spatial planning software. In addition, existence values or cultural values associated with natural features are not easily mapped or quantified such that they can be included in spatial planning software. Important stakeholder values, such as the satisfaction received from using an area for recreation or cultural values such as artistic inspiration or religion, cannot be readily mapped or quantified in dollar values to be included as costs in algorithms. Instead, these values might be more readily accounted for through a social welfare framework in which individuals' preferences can be quantified and trade-offs between different values considered through relative rankings. Therefore, in my research I only considered those social preferences and costs that could be estimated and mapped spatially. It will be important in future work to consider frameworks that allow a more sophisticated treatment of stakeholder preferences and qualitative values.

Moving towards conservation outside of reserves

Conservation planning has historically focused on protected areas. However, there has been a recent movement in the literature to explicitly consider the much broader suite of conservation tools including private land conservation programs such as easements, covenants and management agreements (e.g. Ferraro 2004; Rissman 2010) and incentive programs such as

payments for ecosystem services (PES) and REDD (e.g. Naidoo et al. 2009; Pagiola 2008). While establishment of protected areas will remain an important part of conservation, off-reserve actions are likely to become a greater focus of conservation planning. While my final two chapters looked at designing and implementing a stewardship program on private lands, my other four chapters focused solely on protected areas. The assumption that protected areas will be established in isolation is unrealistic. A more realistic view of conservation planning would be to consider the different types of management actions occurring outside of protected areas to maintain the ecosystem health of the landscape in and around protected areas. A limitation of my initial chapters is that, for computational simplicity, I assumed that all priority areas were selected for protection as opposed to other types of management such as management agreements with private landholders in Queensland or temporary marine protected closures in Fiji. Recent studies have begun to quantify the contributions of different management strategies to conservation objectives and incorporate these variable contributions explicitly in zoned conservation plans with the use of Marxan with Zones (Klein et al. 2009b; Mills et al. In press).

Management implications

Throughout my PhD I worked closely with management agencies, both government and non-government, to ensure that my work was applicable and easily translated to ongoing conservation management. The work presented in Chapter 2 was conducted in consultation with the Queensland Department of Environment and Resource Management (DERM) and initial scoping work was funded by World Wildlife Fund (WWF) and featured in a report by WWF on funding the promised protected area expansion. My research will be used by DERM to help direct future land acquisitions. The research presented in Chapter 4 was conducted with the Wildlife Conservation Society (WCS) Fiji office. I returned all of the developed data and Marxan analyses to the WCS team so that the research can be used in a report to communities, which can then use the Marxan outputs to consider alternative configurations of marine protected areas. Lastly, the research presented in Chapters 6 and 7 was conducted in conjunction with the Northern Territory Department of Natural Resources, Environment, The Arts and Sports (NRETAS). The Northern Territory government has indicated interest in scoping out a potential private land conservation program and is in the process of developing a conservation plan for the Daly catchment. My research will be presented in a report to NRETAS for its use and will also feed directly into the ongoing conservation planning exercise.

Future work

I will be addressing several of the limitations of my thesis in future research work. In particular I will be examining how to approach the systematic spatial conservation planning problem with a social welfare framework and looking for alternatives to the existing algorithms such that qualitative values and preferences of multiple stakeholders can be accounted for.

Addressing limitations of the systematic conservation planning approach to addressing complex socio-economic objectives

As previously discussed, the current conservation planning algorithms are most readily used to account for quantitative stakeholder values, preferences and costs. Therefore, to date most socio-economic considerations in conservation planning, including those in my thesis, have been restricted to values that can be quantified in dollars. I am now scoping out future research that will evaluate stakeholder preferences and values associated with natural resources such as health, well-being and cultural values. These values will be qualitative and I will examine methods of comparing how they scale and whether they are complementary, such that they respond in the same way to different changes in a natural resource, or whether they are in opposition such that, if one increases, another may decrease. I will then examine how social welfare functions can be applied in systematic conservation planning as well as to other natural resource management approaches such as allocating management budgets across a landscape to mitigate impacts of weeds and feral animals. This approach will allow for a more comprehensive accounting of stakeholder values, both qualitative and quantitative, in resource management decisions.

Moving towards planning for off-reserve management

As discussed in Chapters 6 and 7, private land conservation is becoming a prominent conservation strategy to complement current protected areas as expansion of strict protected areas becomes more limited due to restricted land availability and insufficient budgets. However, the design of these programs, in particular spatial considerations, has been relatively neglected in the conservation literature. Newburn et al. (2005) noted that the conservation planning literature has not addressed important considerations pertinent to planning for private land conservation, such as considering land parcel boundaries and actual costs to landholders such as the incremental change in production or management costs as a result of participating in

a conservation program. While I began to address some of these considerations for the Daly catchment (Chapters 6 and 7), there are still many others yet to be explored. Examples include social incentives for encouraging neighbouring properties to participate collectively and engaging landholders with larger properties so that they are more likely to participate. It is also important to explicitly state the objectives of a program, including the expected ecological and social outcomes, so that an appropriate monitoring program can be implemented to measure changes as the program proceeds. I am in the process of applying the framework presented by Ferraro and Pattanayak (2006) to empirically evaluate the social impacts of a stewardship program. I will be using a survey in the Daly to measure potential social and ecological biases that could impact landholders' decisions to participate and the effectiveness of their actions. I will then use a matched-pair approach to measure the social impacts of conservation actions and design future monitoring protocols that are applicable to measuring impacts of conservation programs across Australia.

Concluding remarks

My thesis highlights the importance of stating explicit socio-economic objectives. Explicit objectives are at the heart of systematic conservation planning (Margules & Pressey 2000; Wilson et al. 2009); however, objectives associated with social and economic values are often neglected. Data available for conservation planning are often limited, in particular data pertaining to economic costs and stakeholder values. Nonetheless, ignoring this aspect of the planning process or using coarse-scale data (e.g. country level estimations based on global predictive modeling Kark et al. 2009; Klein et al. 2010) will likely not reflect the true spatial distribution of costs and could result in negative impacts to stakeholders. In fact, as I demonstrated in chapter 5, area can be a better surrogate of total opportunity costs than measures to single stakeholders.

The drive to include socio-economic costs in conservation planning has been to ensure that plans reflect the true socio-political context of a region and are hopefully more readily implemented such that effective conservation actions follow the assessment phase (Knight et al. 2008; Naidoo et al. 2006; Polasky 2008). However, in the rush to fill this gap in the conservation planning literature, the basic need for stating socio-economic objectives that reflect the broader social, cultural and economic goals of communities has been neglected. Future research should be

directed at understanding social goals and effectively translating these into explicit objectives matched with appropriate cost metrics. Methods for eliciting values and goals and estimation of corresponding cost metrics are widely applied in the economics literature. However, conservation planners and economists must bridge the divide between the two fields so that economic methods can be developed and applied to conservation problems in an integrated and rigorous fashion.

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Appendix 1 Chapter 2 Supplementary methods and tables

Hedonic model for property sales value

The market value of a property can be estimated as the sale value in an open land market. I estimated market values from recent sales of properties in Queensland from 2000-2008 (NRW 2008a, b, c). Dates of land valuation and sales data varied, so I adjusted all values to 2008 dollars using published annual inflation rates (RBA 2009). The land value, sale value, tenure and size of each property were recorded from the sales valuation data. In addition, I sourced property characteristics from state-wide data including the area on each property covered by soil of high productivity, the area on each property that was cleared, distance to nearest town, and occurrence in coastal regions (Table A1.1). Other potential explanatory variables, such as current land use, elevation and slope were not consistently mapped for the entire state. I specified a hedonic price model for the market value with the dependent variable being the log of the sale value per hectare. The hedonic analysis was modelled using two methods: 1. ordinary least squares (OLS) estimation in R (R Development Core Team 2005) and 2. Geographically weighted regression in ArcGIS 9.3.

Full models can result in over-fitting, particularly when the model is used to predict values (Hawkins 2004). Therefore, for the hedonic model using OLS, a reduced model was selected using full forward and backward stepwise removal using the Akaike information criterion (AIC; Akaike 1974) to select the best model ('step' function, R Development Core Team 2005) (Table A1.2).

The sale value per hectare was significantly higher in South East Queensland than all other regions of the state. The sale value per hectare for other coastal areas was also significantly higher than non-coastal properties. Cape York had low sale values per hectare. This is in line with expectations of the land market. Brisbane and surrounding coastal areas in the South East portion of the state have significantly higher sale values while Cape York Peninsula is predominantly state and aboriginal land with only a small portion of freehold land being sold around town sites such as Weipa. For coastal regions, sale value decreased with increasing distance to towns. For non-coastal regions the coefficient of *log(distance to nearest town, km)* was positive indicating that sale value increased with increasing distance to towns. This likely reflects the fact that inland leasehold properties with high sale values are often located far from townships. The three variables used to capture existing and potential development were

log(cleared area, ha), *log(kandasol soil, ha)* and *log(vertosol soil, ha)*. These were all positively related to sale value.

For the geographically weighted regression I divided the properties in Queensland into coastal and non-coastal properties due to the large variation in characteristics such as size (for example average size for coastal properties is 1/40 the size of properties across the remainder of the state). For coastal properties the only predictor without strong local correlations was *log(cleared area, ha)* and adjusted R^2 was 0.688. For non-coastal properties the final variables included were *log(land value per ha)*, *log(cleared area, ha)*, *log(soil, ha)* and *log(distance to nearest town, km)* (adjusted R^2=0.904). The values predicted by the geographically weighted regression were highly correlated with the OLS predicted values, suggesting that my regional dummy variables captured some of the local spatial drivers.

Table A1.1 Predictors considered and sources of data

Predictor	Description	Source
log(property area, ha)	Log of area of property in ha	(NRW 2008a)
log(land value per ha)	Log of land value per ha	(NRW 2008c)
log(cleared area, ha)	Log of cleared area of property in ha as a surrogate measure of existing development	(DERM 2009)
log(kandasol soil, ha)	Log of area of property in ha with kandasol soils as a surrogate measure of suitability for agricultural development	(DERM 2010b)
log(vertosol soil, ha)	Log of area of property in ha with vertosol soils as a surrogate measure of suitability for agricultural development	(DERM 2010b)
log(soil, ha)	Log of area of property in ha with vertosol or kandasol soils as a surrogate measure of suitability for agricultural development	(DERM 2010b)
Tenure: Leasehold (LH), Freehold (FH)	Dummy variable: Leasehold (LH)	(NRW 2008a)
log(distance to nearest town, km)	Log of distance to nearest town in km as a surrogate measure of proximity to urban services	(DERM 2010a)
Regions: South East Queensland, Coastal Queensland, Cape York Peninsula, Remainder of State	Dummy variables: South East Queensland (all coastal local government areas from the southern border to the Sunshine Coast), Cape York (Cape York Peninsula) and Coastal Queensland (all other Coastal local government areas)	(DERM 2010c)
LHxlog(property area, ha)	Interaction term for tenure and property size	
SExlog(distance to nearest town, km)	Interaction term for South East Queensland and distance to town	
Coastalxlog(distance to nearest town, km)	Interaction term for Coastal Queensland and distance to town	
CapeYorkxlog(distance to nearest town, km)	Interaction term for Cape York and distance to town	

Table A1.2 Hedonic coefficient estimates for sale value in 2008 using the reduced model with OLS

Variable	Coefficient	Std. Error	Pr(>t)
log(land value per ha)	0.5035	0.00457	0.0001
log(property area, ha)	-0.4876	0.00524	0.0001
log(cleared area, ha)	0.60291	0.00170	0.0001
log(kandasol soil, ha)	0.0193	0.00472	0.0001
log(vertosol soil, ha)	0.0893	0.00482	0.0001
log(distance to nearest town, km)	0.0358	0.01012	0.0001
South East Queensland	0.2197	0.01455	0.0001
Cape York	-0.1812	0.02794	0.0001
Coastal	0.1579	0.01543	0.0001
LHxlog(property area, ha)	0.0662	0.00549	0.0001
SExlog(distance to nearest town, km)	-0.1792	0.01242	0.0001
Coastalxlog(distance to nearest town, km)	-0.1600	0.01308	0.0001
Constant	2.818	0.02388	0.0001
N=41901 parcels			
R^2 (adjusted) =0.901			

Appendix 2 Chapter 3 Supplementary methods and figures

Info-gap model for parameter uncertainty in conservation decision making

Info-gap theory has three model requirements: a process model to measure performance, an uncertainty model, and a performance requirement (Ben-Haim 2006). The uncertainty model is a mathematical equation describing the error involved in my best estimates of the data. The performance requirement is the critical threshold at which my decisions must perform. I consider the scenario in which a conservation organization is purchasing properties for conservation. Thus, the organization must allocate a set budget over a set *S* of *i* properties for *i=1…N*, where each property, s_i, has an associated benefit b_i and associated cost c_i.

I select the set of properties for purchase using a cost-efficient allocation by including a property in my set $S(\tilde{\gamma},\theta)=\{s:\gamma_i \geq \theta\}$ where $\gamma_i = \frac{b_i}{c_i}$ and I choose θ such that any property with a BCR greater than or equal to θ is selected and any property with a BCR less than θ is excluded from my conservation portfolio (the set *S*). In this application, the selection of θ is budget-driven and derived by ranking properties based on BCRs from highest to lowest and selecting properties until the budget is exhausted.

I first describe the error of my best estimates of b_i and c_i, $\tilde{b}_i$ and $\tilde{c}_i$ using the fractional-error info-gap model (Ben-Haim 2006; Regan et al. 2005):

$$U(\alpha,\tilde{b},\tilde{c})=\left\{\begin{array}{l} b:\left|\frac{b_i-\tilde{b}_i}{\tilde{b}_i}\right| \leq \alpha \\ c:\left|\frac{c_i-\tilde{c}_i}{\tilde{c}_i}\right| \leq \alpha \end{array}\right\}, \alpha>0$$

where α is the "horizon of uncertainty" and is unbounded and unknown (Ben-Haim 2006). In this case α is the same for both *b* and *c*. The fractional-error model for *b* and *c* reduces to:

$(1-\alpha)\tilde{b}_i \leq b_i \leq (1+\alpha)\tilde{b}_i$ and $(1-\alpha)\tilde{c}_i \leq c_i \leq 1+\alpha)\tilde{c}_i$.

The greater my uncertainty, the higher the value of α; a value of 0.10 means I am 10% uncertain in my values of *b* and *c* and a value of 0.50 means I am 50% uncertain in my values of *b* and *c*.

To measure the performance of my process model, I first define the set of $M(\gamma)$ to be the number of properties that are excluded or included in error. To define $M(\gamma)$, I first consider the union of

the sets of properties selected, assuming my best estimates of b_i and c_i, $S(\tilde{\gamma})$, and the set of properties assuming complete knowledge of b_i and c_i, $S(\gamma)$. The union is the set of all properties selected regardless of uncertainty. I then consider the intersection between the two sets. The intersection gives me the set of properties that is included both with and without uncertainty. If I then remove the intersection from the union, I am left with only those properties that were included or excluded in error. So, for a given $\theta, M(\gamma) = \left|\left[S(\gamma) \cup S(\tilde{\gamma})\right] - \left[S(\gamma) \cap S(\tilde{\gamma})\right]\right|$. To account for the fact that different values of θ will select variable numbers of properties, I define my performance measure to be the percentage of properties in my selected set, $S(\gamma)$, to be have been misallocated. In other words I define my performance measure to be $D(\gamma) = M(\gamma)/S(\gamma)$x100. I would like the value of $D(\gamma)$ to be as small as possible, but I select a critical value D_c as the acceptable performance threshold. Values of $D(\gamma)$ larger than this critical value are unacceptable.

The robustness function for my set $S(\gamma)$ is given by:

$$\hat{\alpha}(\theta, D_c) = \max\left\{\alpha : \left(\max_{b,c \in U(\alpha)} D(\gamma, \theta)\right) \le D_c\right\}.$$

Thus, the equation addresses the question of how wrong, or uncertain, I can be in my estimates of b and c while still achieving an acceptable outcome, or minimizing the number of misallocations. The system of equations can be solved to explore the robustness of conservation budget allocations for a combination of values of the variables θ and D_c.

For comparison I also investigate the process model of selecting sets of areas based only on benefits such that the set of properties is $S(\tilde{b}, \beta) = \{s : b_i \ge \beta\}$. Like the method described above, I use the fractional-error info-gap model (Ben-Haim 2006; Regan et al. 2005) to describe the error of my best estimates of b_i, $\tilde{b}_i$:

$$U(\alpha, \tilde{b}, \tilde{c}) = \left\{b : \left|\frac{b_i - \tilde{b}_i}{\tilde{b}_i}\right| \le \alpha\right\}, \alpha > 0$$

where α is the "horizon of uncertainty" and is unbounded and unknown (Ben-Haim 2006). I formulate the performance of my process model in the same manner. I define the set of $M(b)$ to be the percentage of properties that are excluded or included in error. Or, for a given β,

$M(b) = \left| \left[S(b) \cup S(\tilde{b}) \right] - \left[S(b) \cap S(\tilde{b}) \right] \right|$ and $D(b) = M(b)/S(b)$x100. I select a critical value D_c – the acceptable performance threshold - for which a larger value of $D(b)$ is unacceptable.

The robustness function for my set $S(b)$ is given by:

$$\hat{\alpha}(\beta, D_c) = \max\left\{ \alpha : \left(\max_{b \in U(\alpha)} D(b, \beta) \right) \leq D_c \right\}.$$

Data

For each property, I calculate the benefit, b_i, using the definition given by Fuller et al. (2010):

$$b_i = \sum_{j=1}^{m} a_{ji} / r_j$$

where m is the number of vegetation types, a_{ji} is the area of vegetation type j in parcel i, and r_j is the proportion of vegetation type j remaining since clearing has occurred. I use the regional ecosystem mapping (~1300 regional ecosystems mapped across Queensland) (DERM 2009).

I define the cost c_i for each land parcel to be the acquisition cost to the government:

$$c_i = p_i + t_i$$

where p_i is the price of the land parcel, based on estimated sales value per property (Adams et al. 2011), and t_i is the transaction cost, which I assume to be $15,000 for all properties.

Model analysis

I assume a fixed budget of $120 million, set θ accordingly, and select my portfolio $S(\gamma,\theta)$. I use the term "budget-driven θ" for the θ value based on the budget of $120 million to indicate the situation in which a department is given a non-negotiable budget to purchase properties. I solve the system of equations for $\hat{\alpha}(\theta, D_c)$ and plot $\hat{\alpha}(\theta, D_c)$ against D_c, which is considered the robustness curve associated with a selected value of θ. The robustness of a selected value of θ is given by the x-value, or the value of α at which D_c exceeds the acceptable threshold of performance. Conversely, I can interpret robustness as the y-value, or number of misallocations (D_c) that can be accepted for a given value of α. Therefore, for different values of θ, the curve that is furthest to the right represents the most robust set of selections. Crossing of robustness curves indicates that, for a given critical value D_c, if the manager wishes to select the most robust strategy, then the selection of θ would change. Thus, whether θ is selected a priori or based on a

budget, if the selected critical value D_c is near a cross in robustness curves, then selection of θ should be considered carefully. A theoretical example is provided in Figure A2.1.

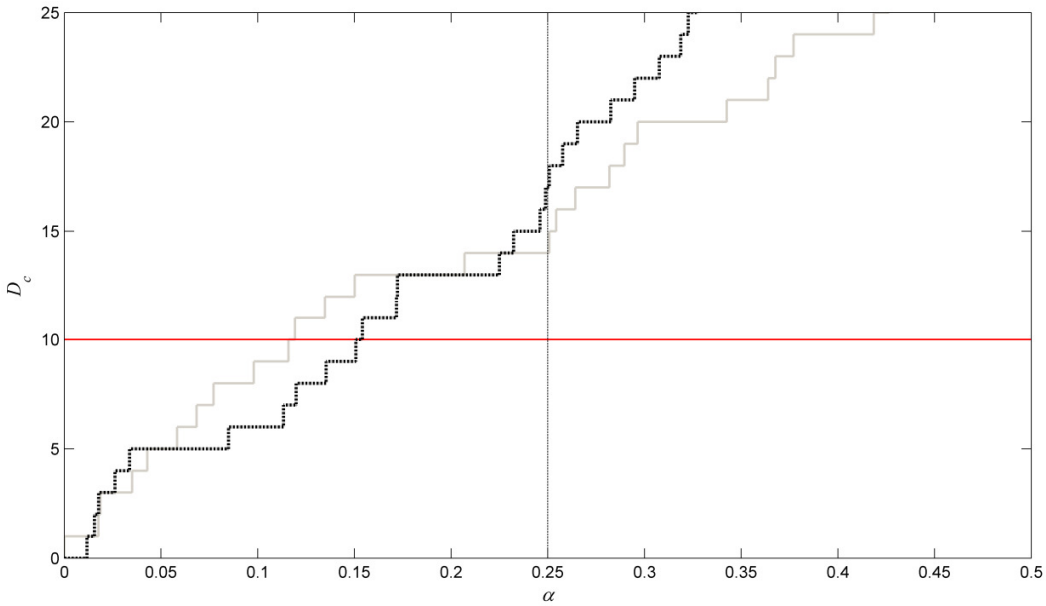


Figure A2.1 Example robustness curves for two values of θ using BCR ranking to select properties. The curve furthest to the right for a given critical value D_c represents the more robust set of conservation investments. A critical value threshold of $D_c = 10\%$ is shown in red. Where this line crosses each curve indicates how certain I must be in my estimates of b and c to ensure that I do not exceed the critical value of 10% misallocation of conservation investments. Conversely, if I know the uncertainty around my estimates of b and c, the robustness curve tells me what percentage of investments I can expect to be misallocated. The dashed vertical line at α=0.25 indicates the scenario in which I believe my data for benefits and costs are 75% accurate (or 25% inaccurate). At this level of uncertainty, for my two robustness curves, I can expect to have ~13% and ~16% of my portfolios misallocated.

Appendix 3 Chapter 4 Supplementary methods and figures

Species abundance models

Four models were compared for each species: Poisson (P), negative binomial (NB), zero-inflated Poisson (ZIP) and zero-inflated negative binomial (ZINB).

The Poisson distribution has a probability mass function of:

$$\Pr(N = x \mid \lambda) = \frac{e^{-\lambda}\lambda^{x}}{x!}, \; x = 0,1,2,\ldots,\infty$$

where λ is the mean, which can be interpreted as the mean number of events occurring in an area or time. In this case I interpret λ to be the number of fish counted in the 250 m^2 survey area or, in time units, the number of fish counted in 0.33 hrs. A limitation of the Poisson distribution is that the variance is equal to the mean. In biological data sets this is unlikely to be the case. In the case of over-dispersed data, where the sample variance is greater than the sample mean, the negative binomial distribution can be used. The negative binomial distribution can account for larger variances with the use of the additional parameter θ. The probability mass function of the negative binomial is:

$$\Pr(N = x \mid \lambda, \theta) = \frac{(x+\theta-1)!}{(\theta-1)!x!}\left(\frac{\theta}{\theta+\lambda}\right)^{-\theta}\left(\frac{\lambda}{\lambda+\theta}\right)^{x}, \; x = 0,1,2,\ldots,\infty$$

where λ is the mean, θ is the dispersion parameter and N is the number of individuals in the survey site.

Environmental predictors (reef type, presence of channel, exposure to tides and waves, depth, distance to shore, and protection status) were incorporated into the generalized linear regression and the log-linear transformation was applied:

$$\log(\lambda_i) = \gamma_0 + \sum_j \gamma_j x_i$$

where λ_i is the mean abundance at site *i,* γ_0 is the intercept coefficient, x_i are the predictor variables for site *i,* and γ_j are the predictor coefficients determined by maximum likelihood.

In zero-inflated mixture models the models are a mixture of a Bernoulli process (to model occupancy of sites) and Poisson or negative binomial process (to model site abundance). The mixture model assumes that some of the zeros occur due to the absence of a species at a site (captured by the Bernoulli process) and some of the zeros occur due to error in the count model (captured by the Poisson or negative binomial process). Therefore, the probability mass function for the zero-inflated Poisson process is:

$$\Pr(N_i = 0 \mid \lambda, \psi) = \psi + (1-\psi)e^{-\lambda}$$

$$\Pr(N_i > 0 \mid \lambda, \psi) = (1-\psi)\frac{e^{-\lambda}\lambda^x}{x!}, \text{ x = } 1,2,\ldots,\infty$$

where N is the actual site abundance, ψ is the probability that the site is not occupied and λ is the mean abundance. Similarly, the probability mass function for the zero-inflated negative binomial process is:

$$\Pr(N_i = 0 \mid \lambda, \theta, \psi) = \psi + (1-\psi)\left(\frac{\theta}{\theta+\lambda}\right)^{-\theta}$$

$$\Pr(N_i > 0 \mid \lambda, \theta, \psi) = (1-\psi)\frac{(x+\theta-1)!}{(\theta-1)!x!}\left(\frac{\theta}{\theta+\lambda}\right)^{-\theta}\left(\frac{\lambda}{\lambda+\theta}\right)^{x}, \text{ x = } 1,2,\ldots,\infty$$

where N is the actual site abundance, ψ is the probability that the site is not occupied, λ is the mean abundance and θ is the dispersion parameter of the negative binomial.

The log-linear transformation is applied to the Poisson and negative binomial components of the ZIP and ZINB. For the Bernoulli process a logistic regression is applied to predict occupancy:

$$\log it(\psi_i) = \beta_0 + \sum_j \beta_j x_i$$

where ψ_i is probability that the site i is not occupied, β_0 is the intercept coefficient, x_i are the predictor variables for site i and β_j are the predictor coefficients determined by maximum likelihood.

The predictor variables were selected based on available data representing biophysical and anthropogenic factors that influence abundance (Table A3.2). Note that all predictors were considered for both the Bernoulli, Poisson and negative binomial processes in the mixture models. However, the best subsets of predictors were selected for each of the processes in the mixture models (i.e. the sets of predictors selected for the Bernoulli processes could be different from the sets selected for the Poisson or negative binomial models).

Details of the model selected for each species are provided along with the AIC and proportion zeros in Table A3.3.

A standard measure of goodness of fit for models estimated with maximum likelihood is a chi-square test. However, none of the recent publications estimating count models such as Poisson or Zero-inflated Poisson report goodness of fit statistics. This is likely due to data restrictions. To run the chi-square test the expected count data must be divided into bins such that no bin has a zero count and at least 80% of bins have a count of at least 5. Due to the nature of my data (63 sites, with inflated zero counts) it was impossible to divide them to meet these requirements with enough degrees of freedom to conduct the chi-square test. I have therefore modelled the data by species and summed by family the expected and observed counts at each site. Histograms of the expected and observed abundance for selected families are provided below for visual inspection

(Figs A3.1-A3.5). The total abundances expected across the 63 sites are typically comparable to the observed abundances, indicating that the models are not underestimating abundance, but instead averaging the larger counts in the tail of the observed abundance distribution.

Table A3.1 List of transport and gear types from catch per unit effort surveys completed by trained Kubulau community representatives between July 2008 to February 2009. The largest extent covered is listed for each combination of transport and gear type. For combinations of transport and gear type in which only a point was indicated rather than a polygon, I set the area equal to the next closest combination of gear and transport. Transport types are all self-locomotive except for boats, which I assume to have a standard 25hp engine. A bilibili is a communal bamboo raft often pushed with a long paddle or used by swimming alongside. The maximum expected distance travelled from shore by transport is listed.

Transport	**Gear type**	**Largest extent covered (km^2)**	**Maximum expected distance travelled from shore (km)**
Walk	Gill net	1.651	0.5
Walk	Hand spear	0.989	0.5
Walk	Speargun	0.609	0.5
Walk	Hand line	1.763*	0.5
Swim	Speargun	0.180	1.0
Swim	Hawaiian sling	0.180	1.0
Bilibili	Gill net	1.918	3.0
Bilibili	Hand line	1.763*	3.0
Bilibili	Speargun	0.179	3.0
Bilibili	Hand spear	0.989§	3.0
Bilibili	Hawaiian sling	1.111†	3.0
Bilibili	Speargun	0.179	3.0
Boat	Gill net	1.752	Unlimited
Boat	Hand line	1.763	Unlimited
Boat	Speargun	1.111	Unlimited
Boat	Trolling	1.717	Unlimited
Boat	Hawaiian sling	1.111†	Unlimited
Boat	Hand spear	0.989§	Unlimited

* Data unavailable. Assumed the largest area used is the same as hand line and boat.
† Data unavailable. Assumed to be the same as speargun and boat.
§ Data unavailable. Assumed to be the same as hand spear and walk.

Table A3.2 Biophysical and anthropogenic predictors of fish species abundance and biomass.

Predictor	Variable name	Variable type
Reef Type	Fringing	Binary
	Barrier	Binary
	Patch	Binary
Channel	Channel	Binary
Exposure to tides	Exposed	Binary
Exposure to waves	Forereef	Binary
Depth ≤ 10m	Shallow	Binary
Distance to shore	Distance	Continuous
Protection status*	MPA	Binary
	Namena	Binary

* Protection status was broken down into MPA (Nasue and Namuri) and Namena. Namena was established prior to Nasue and Namuri and the effects on fish abundance are likely to be dissimilar to the other two MPAs due to this longer establishment period. Namena was therefore recorded separately for the purposes of modelling.

Table A3.3. Details of abundance models selected for each species.

Family	Species	Model	Equation	AIC	Proportion zeros
Acanthuridae	*Acanthurus auranticavus*	ZINB	log(λ) = 2.3864+1.1218mpa -2.0971 barrier - 3.8590 channel + 2.1867forereef logit(ψ) = 0.4797 -0.9739barrier	306.4	0.970
Acanthuridae	*Acanthurus grammoptilus*	ZINB	log(λ) = 2.6131-0.9214mpa +1.1146 barrier + 0.5012forereef logit(ψ) = 0.8412-0.9041barrier + 1.1382forereef	239	0.978
Acanthuridae	*Acanthurus lineatus*	NB	log(λ) = 0.1207-1.9014exposed + 2.6723forereef + -24.7096shallow	255.45	0.947
Acanthuridae	*Acanthurus nigrofuscus*	ZINB	log(λ) = 3.0388-1.5978shallow logit(ψ) = 1.9039-1.7157forereef	229	0.977
Acanthuridae	*Acanthurus pyroferus*	ZINB	log(λ) = 2.0468+0.9179namena -0.7134 mpa - 0.8365 patch logit(ψ) = 0.8787-1.4944forereef	269	0.744
Acanthuridae	*Acanthurus triostegus*	ZINB	log(λ) = 2.8032+1.3596mpa +0.5645 barrier - 0.6272exposed + 1.9735forereef logit(ψ) = 0.0002767 +2.3028657patch + 2.8906445forereef	128.3	1.000
Acanthuridae	*Ctenochaetus striatus*	ZINB	log(λ) = 4.47-0.44namena + 0.75barrier +0.67 channel -0.43 shallow logit(ψ) = -21.98+1.87 exposed +20.11 shallow	698.4	0.800
Acanthuridae	*Naso lituratus*	NB	log(λ) = 0.025+2.97namena -2.54 exposed + 2.96forereef -1.99 shallow -0.000103distance	288.7	1.000
Acanthuridae	*Naso unicornis*	ZINB	log(λ) = 0.72-2.29barrier + 4.52exposed + 4.83forereef logit(ψ) = 26.55-17.28barrier + 34.87exposed + 17.64forereef	292.6	0.897
Acanthuridae	*Zebrasoma scopas*	ZINB	log(λ) = 2.47-0.39namena + 0.43barrier -1.13 exposed +1.22 forereef logit(ψ) = -1.04+2.58exposed -20.12 forereef	491.8	1.000
Acanthuridae	*Zebrasoma veliferum*	ZINB	log(λ)= 3.07-1.38barrier -0.89patch -1.29 shallow logit(ψ) = 1.15-2.85barrier -1.69 shallow	292.8	0.917
Balistidae	*Balistapus undulatus*	ZINB	log(λ) = 1.2849+0.6105barrier + 1.1508forereef + 0.7409shallow logit(ψ) = -1.7154-18.9078forereef	432.2	1.000
Balistidae	*Balistoides viridescens*	ZIP	log(λ) = -1.2795+1.1974barrier + 1.6205shallow logit(ψ) = -0.5668 +3.2307 shallow	103.44	0.979
Balistidae	*Pseudobalistes flavimarginatus*	ZIP	log(λ) = 2.48+7.21channel -3.12 forereef -5.31 shallow logit(ψ) = 12.66 -13.42barrier -30.81 shallow	87.9	1.020

Balistidae	*Sufflamen bursa*	ZINB	log(λ) = 0.08+1.21barrier -17.34 exposed logit(ψ) = 2.39-2.32forereef	130.8	1.000
Balistidae	*Sufflamen chrysopterum*	ZINB	log(λ) = 1.20+13.93namena -13.44 exposed logit(ψ) = 1.73 -16.34exposed	106.77	1.000
Carangidae	*Caranx melampygus*	NB	log(λ) = -6.03+3.62patch + 2.66channel + 1.21exposed + 3.41forereef + 1.14shallow + 0.0001754distance	145.83	0.933
Haemulidae	*Plectorhinchus chaetodonoides*	ZIP	log(λ) = 0.78-2.24mpa -2.58 patch - 2.65exposed + 1.32shallow logit(ψ) = 1.71-1.79forereef	121.4	1.000
Lethrinidae	*Lethrinus obsoletus*	ZINB	log(λ) = -2.18+1.7554namena + 1.546mpa +0.9715 barrier + 5.0626patch + 2.594forereef logit(ψ) = -25.55+26.92patch +26.23 forereef + 12.76shallow	163.28	1.000
Lethrinidae	*Monotaxis grandoculis*	ZINB	log(λ) = 2.87+0.58namena +0.70 barrier + 1.37channel -1.57 exposed -0.66 shallow logit(ψ) = -3.80+2.64shallow	970.4	1.021
Lutjanidae	*Lutjanus bohar*	NB	log(λ) = 0.62+0.89namena -1.05 mpa + 2.12barrier + 3.48channel -1.91exposed - 1.49shallow	362.2	0.882
Lutjanidae	*Lutjanus fulviflamma*	NB	log(λ) = -23.05+22.18patch +1.84channel- 23.12 exposed +23.54 forereef +.0000758distance	196.2	1.000
Lutjanidae	*Lutjanus fulvus*	ZINB	log(λ) = -1.18-1.37mpa +3.12 barrier +4.12channel logit(ψ) = -0.36 +17.89exposed + 1.36shallow	179.86	1.000
Lutjanidae	*Lutjanus gibbus*	ZINB	log(λ) = 4.14+1.33channel -1.39 exposed -1.62 shallow logit(ψ) = 9.40-22.52barrier -20.62 channel - 8.52 shallow	486.8	1.000
Lutjanidae	*Lutjanus kasmira*	NB	log(λ) = -30.67+7.39namena -29.64 mpa + 30.52barrier + 29.20patch -0.00022distance	97.45	0.981
Lutjanidae	*Lutjanus semicinctus*	NB	log(λ) = 0.79+1.78barrier +1.22 patch + 2.27channel -0.80shallow-0.0000263 distance	378.16	1.000
Lutjanidae	*Macolor macularis*	ZIP	log(λ) = -1.90+1.79namena + 2.21barrier -2.70 exposed +4.61 shallow logit(ψ) = 0.14 -12.44channel + 12.29shallow	113.2	0.980
Lutjanidae	*Macolor niger*	ZINB	log(λ) = -1.70+1.11namena +0.51 mpa - 1.15barrier + 2.05exposed +5.08 forereef logit(ψ) = 0.79-17.52barrier + 30.56exposed - 20.49shallow	364.2	1.000
Scaridae	*Cetoscarus bicolor*	ZINB	log(λ) = 0.94+2.30barrier + 3.28channel - 2.06exposed -1.45forereef logit(ψ) = 0.55+11.28barrier -21.65 exposed -	282.8	0.913

			13.95 forereef		
Scaridae	*Chlorurus bleekeri*	ZINB	log(λ) = 2.88 -0.81exposed logit(ψ) = -0.74+3.52exposed -3.19 forereef	422.4	0.933
Scaridae	*Chlorurus microrhinos*	ZINB	log(λ) = -0.05+0.80namena -0.74mpa + 2.58barrier + 3.00channel -1.50 exposed logit(ψ) = 1.02-3.74forereef	315	0.926
Scaridae	*Chlorurus sordidus*	ZINB	log(λ) = 3.99-0.98namena + 0.74barrier - 0.72exposed -0.55 shallow logit(ψ) = 20.59+19.22shallow	629.4	0.800
Scaridae	*Hipposcarus longiceps*	ZINB	log(λ) = 5.33-2.16patch -2.77 exposed -3.16 forereef -2.86 shallow logit(ψ) = -0.27-0.50barrier	304	1.000
Scaridae	*Scarus altipinnis*	NB	log(λ) = 1.64-3.49shallow	171.8	0.980
Scaridae	*Scarus chameleon*	P	log(λ) = -19.38 -0.08mpa -21.29channel - 17.39 exposed -20.53 forereef - 0.000081distance	130.52	0.811
Scaridae	*Scarus dimidiatus*	ZINB	log(λ) = 2.75-2.60namena -2.13 channel -0.83 forereef logit(ψ) = 0.59+1.88barrier -2.67 forereef -1.90 shallow	278.6	1.000
Scaridae	*Scarus forsteni*	NB	log(λ) = -27.77-1.32mpa -55.96 channel - 26.71exposed +28.66 forereef - 0.000092distance+ 26.47fringing	93.2	1.000
Scaridae	*Scarus ghobban*	NB	log(λ) = 0.80-2.20mpa -1.24exposed - 1.01forereef	165.79	1.000
Scaridae	*Scarus niger*	ZINB	log(λ) = 1.27+0.76 barrier -1.94 exposed +1.31forereef logit(ψ) = 1.15-4.44forereef	352.4	1.000
Scaridae	*Scarus oviceps*	NB	log(λ) = 1.29 -1.73mpa + 1.86barrier -2.7 exposed -0.000207 distance	188.58	1.000
Scaridae	*Scarus psittacus*	NB	log(λ) = -2.76 -2.24mpa + 4.75barrier - 0.000123 distance	133.74	0.980
Scaridae	*Scarus rubroviolaceus*	NB	log(λ) = 0.45+1.75namena -2.40 mpa + 2.82barrier + 3.76channel -3.19shallow - 0.000184distance	216.43	1.057
Scaridae	*Scarus schlegeli*	ZINB	log(λ) = 3.06-0.68 namena -0.71shallow logit(ψ) = -1.04 -1.59forereef	456.2	1.000
Scaridae	*Scarus spinus*	NB	log(λ) = 0.47-1.47exposed +2.50 forereef -2.09 shallow -0.000074distance	274.68	0.967
Serranidae	*Cephalopholis argus*	NB	log(λ) = -1.66+2.05namena + 3.27barrier +3.76 channel -1.99 exposed -2.73shallow + - 0.0000695distance	202.11	1.000

Serranidae	*Cephalopholis urodeta*	ZINB	log(λ) = -1.50-0.37mpa +3.32 barrier +1.75 channel -0.67 exposed +1.25 shallow logit(ψ) = 1.26-2.45forereef	296.06	0.992
Serranidae	*Epinephelus Merra*	ZINB	log(λ) = -0.92-1.17namena + 2.84barrier + 1.92patch + 1.99channel logit(ψ) = -1.37-1.54barrier	301.8	0.958
Serranidae	*Plectropomus laevis*	ZIP	log(λ) = -0.17+1.46namena -2.93 exposed logit(ψ) = 11.48-26.75exposed -11.70forereef	90.42	1.000
Serranidae	*Plectropomus leopardus*	NB	log(λ) = -1.54+3.43namena +2.33 patch -31.53 exposed +3.96 forereef -0.000218distance	195.56	1.000
Siganidae	*Siganus doliatus*	ZINB	log(λ) = 3.20-1.53namena -0.63 mpa -1.13 shallow logit(ψ) = -1.76 +1.05barrier +3.05 exposed	368.8	1.000
Siganidae	*Siganus guttatus*	ZIP	log(λ) = 1.32 -1.22mpa -1.94 channel -16.97 exposed -1.42 shallow logit(ψ) = 0.167 +0.74barrier	125.12	1.000
Siganidae	*Siganus punctatus*	ZIP	log(λ) = 2.54 -1.18patch -1.43 exposed -1.17 forereef logit(ψ) = 1.63-1.65forereef	168.56	1.000
Siganidae	*Siganus stellatus*	ZINB	log(λ) = -16.67 -0.91namena + 1.36barrier +18.44 patch +17.07 forereef logit(ψ) = -0.05+2.32 patch	163.42	1.000
Siganidae	*Siganus uspi*	ZINB	log(λ) = 3.09 -1.31namena -1.39 patch -2.76 exposed -1.24shallow logit(ψ) = -13.91+12.21patch + 13.91shallow	348.2	0.900

Table A3.4 Biomass model by species

Family	Name	Equation	R^2
Acanthuridae	*Acanthurus auranticavus*	0.08687+0.1129barrier+.06988forereef	0.343
Acanthuridae	*Acanthurus grammoptilus*	0.05369	NA*
Acanthuridae	*Acanthurus lineatus*	0.1289	NA*
Acanthuridae	*Acanthurus nigrofuscus*	0.0452	NA*
Acanthuridae	*Acanthurus pyroferus*	0.05111	NA*
Acanthuridae	*Acanthurus triostegus*	0.0326	NA*
Acanthuridae	*Ctenochaetus striatus*	0.04438+0.02797barrier-0.0251exposed	0.193
Acanthuridae	*Naso lituratus*	0.2837	
Acanthuridae	*Naso unicornis*	0.2179+0.4223forereef	0.135
Acanthuridae	*Zebrasoma scopas*	0.00892- 0.01299namena+ 0.00812forereef + 0.00000081distance	0.256
Acanthuridae	*Zebrasoma veliferum*	0.083921	NA*
Balistidae	*Balistapus undulatus*	0.01805+0.03019namena+0.04716patch+ 0.05426forereef	0.183
Balistidae	*Balistoides viridescens*	1.231-1.1965shallow	0.171
Balistidae	*Pseudobalistes flavimarginatus*	0.6069+1.261forereef	0.237
Balistidae	*Sufflamen bursa*	0.03429	NA*
Balistidae	*Sufflamen chrysopterum*	0.03615	NA*
Carangidae	*Caranx melampygus*	0.9611+2.1545mpa	0.387
Haemulidae	*Plectorhinchus chaetodonoides*	2.1242	NA*
Lethrinidae	*Lethrinus obsoletus*	0.3533	NA*
Lethrinidae	*Monotaxis grandoculis*	0.25887-0.1894patch	0.124
Lutjanidae	*Lutjanus bohar*	0.09502+0.59435mpa+0.74919forereef	0.307
Lutjanidae	*Lutjanus fulviflamma*	.08177+.2529forereef	0.152
Lutjanidae	*Lutjanus fulvus*	0.04516+0.11536barrier	0.212
Lutjanidae	*Lutjanus gibbus*	0.10798+0.16574forereef	0.199
Lutjanidae	*Lutjanus kasmira*	0.06106+0.06775forereef- 0.0000015distance	0.573
Lutjanidae	*Lutjanus semicinctus*	0.17084+0.10365forereef	0.176
Lutjanidae	*Macolor macularis*	0.940604	NA*
Lutjanidae	*Macolor niger*	0.05885+0.4562forereef	0.136
Scaridae	*Cetoscarus bicolor*	0.1659+0.9739forereef+ 0.0000172distance	0.392
Scaridae	*Chlorurus bleekeri*	0.4115	NA*
Scaridae	*Chlorurus microrhinos*	0.4849+0.3054forereef+ 0.0000183distance	0.242
Scaridae	*Chlorurus sordidus*	0.09651+0.14509forereef	0.333
Scaridae	*Hipposcarus longiceps*	0.4333+0.7574forereef	0.291
Scaridae	*Scarus altipinnis*	0.2916	NA*
Scaridae	*Scarus chameleon*	0.1182	NA*

Scaridae	*Scarus dimidiatus*	0.1786+0.1425barrier-0.0000073distance	0.385
Scaridae	*Scarus forsteni*	0.253	NA*
Scaridae	*Scarus ghobban*	0.5663+0.3324forereef	0.135
Scaridae	*Scarus niger*	0.15894+0.11455barrier	0.221
Scaridae	*Scarus oviceps*	0.10186+0.07587barrier	0.145
Scaridae	*Scarus psittacus*	0.41228-0.2559exposed	0.298
Scaridae	*Scarus rubroviolaceus*	3.254-1.5032barrier-3.0765patch	0.427
Scaridae	*Scarus schlegeli*	0.2877	NA*
Scaridae	*Scarus spinus*	0.3271+0.0571namena-0.2375patch-0.1985forereef- 0.000003distance	0.625
Serranidae	*Cephalopholis argus*	0.45643-0.29842exposed	0.080
Serranidae	*Cephalopholis urodeta*	0.11417	NA*
Serranidae	*Epinephelus Merra*	0.05	NA*
Serranidae	*Plectropomus laevis*	4.6679	NA*
Serranidae	*Plectropomus leopardus*	0.1436+0.495forereef+0.9421namena+0.3212mpa + 0.2777fringing	0.572
Siganidae	*Siganus doliatus*	0.07095-0.03856patch	0.147
Siganidae	*Siganus guttatus*	0.1279	NA*
Siganidae	*Siganus punctatus*	0.04913+0.16248namena+0.10058barrier	0.447
Siganidae	*Siganus stellatus*	0.066+0.1465namena+0.1277shallow	0.486
Siganidae	*Siganus uspi*	0.066+0.0585barrier-0.000002distance	0.145

*No significant predictors. Average biomass was used.

Table A3.5. Predictor variables for percentage catch models.

Predictors	Description
time	The amount of time spent fishing per fisher, determined from CPUE data
$time^2$	Square of the amount of time spent fishing per fisher
fisher	Number of fishers fishing together, determined from CPUE data
area	Area in km^2
ab	Observed abundance of target species
$abperkm^2$	Observed abundance of target species per km^2

Table A3.6. Details of percentage catch models by gear type. Model details and R^2 values are given as well as the genera included, based on CPUE surveys.

Gear type	Model	R^2	Genera included
Gill net	0.0499 - 0.002 fisher + 0.00035 time - 0.000003 abperkm^2 - 0.0481 area	0.617	*Balistoides, Caranx, Cephalopholis, Chlorurus, Lethrinus, Lutjanus, Scarus, Siganus, Macolor*
Hand line	0.0172 - 0.00259 time + 0.000017 time^2 - 0.000001 abperkm^2	0.516	*Balistapus, Cephalopholis, Ctenochaetus, Epinephelus, Lethrinus, Lutjanus, Plectropomus*
Hand spear	0.0886 - 0.0261 fisher + 0.00160 time - 0.000012 time^2 - 0.000002 ab	0.948	*Balistoides, Chlorurus, Lethrinus, Lutjanus, Scarus, Siganus*
Hawaiian sling	0.00195	NA*	*Chlorurus, Ctenochaetus, Scarus, Siganus*
Speargun	0.0706 + 0.000393 time - 0.000001 time^2 - 0.00505 fisher - 0.0338 areakm^2 - 0.000003 abperkm^2	0.396	*Acanthurus, Caranx, Cetoscarus, Chlorurus, Hipposcarus, Lethrinus, Lutjanus, Macolor, Monotaxis, Naso, Plectropomus, Scarus, Siganus*
Trolling	0.0015	NA*	*Balistoides, Caranx, Cephalopholis, Chlorurus, Epinephelus, Plector, Scarus, Siganus*

*Sample size was too small, average catch percentage was used

Figure A3.1. Histogram of expected and observed abundance across the 63 sites for the family Acanthuridae.

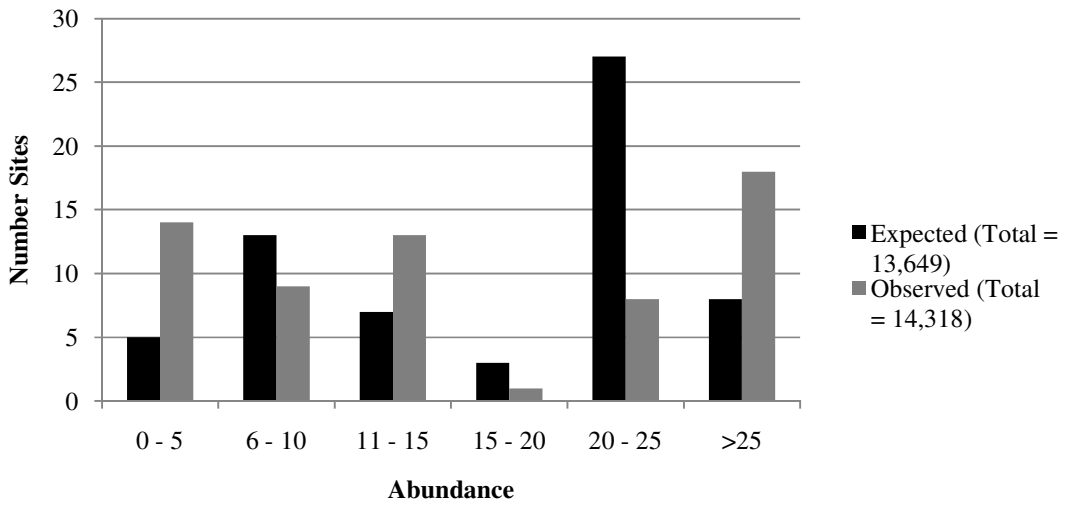


Figure A3.2. Histogram of expected and observed abundance across the 63 sites for the family Carangidae.

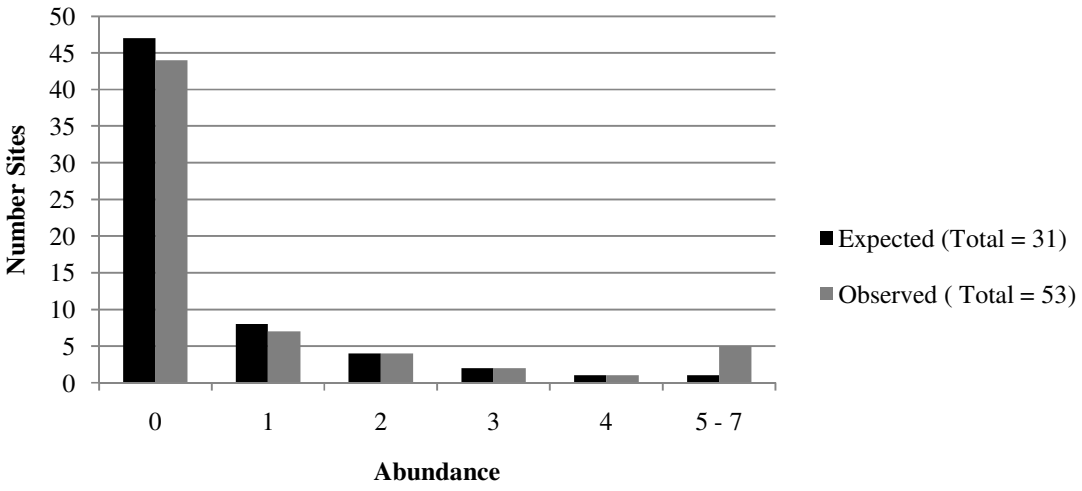


Figure A3.3. Histogram of expected and observed abundance across the 63 sites for the family Lutjanidae.

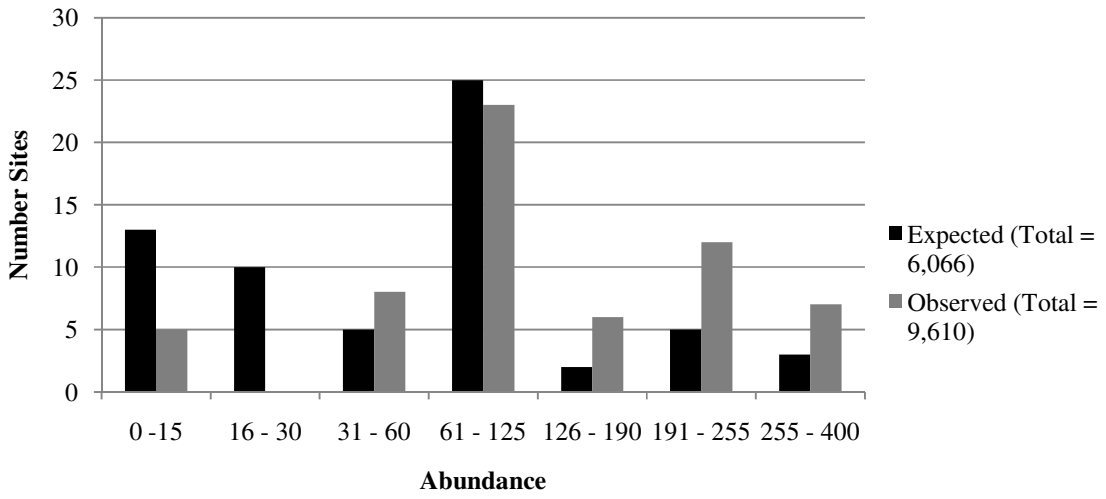


Figure A3.4. Histogram of expected and observed abundance across the 63 sites for the family Scaridae.

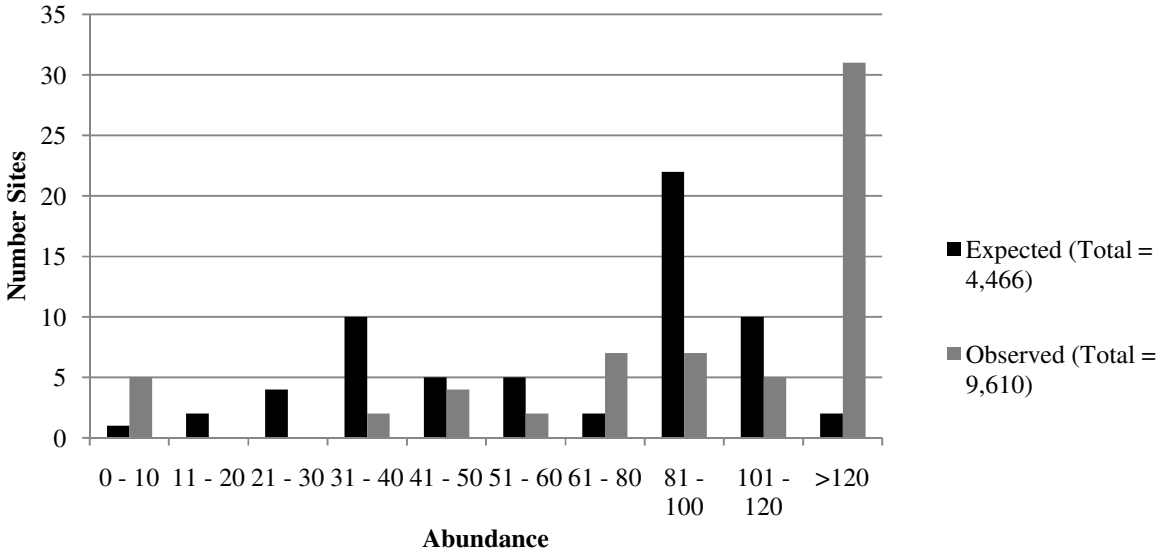
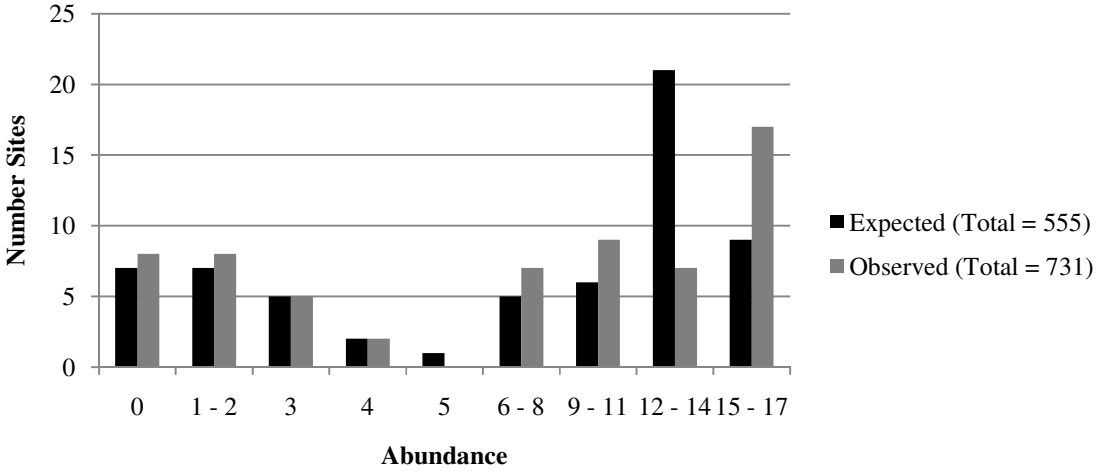


Figure A3.5. Histogram of expected and observed abundance across the 63 sites for the family Serranidae.



Appendix 4 Survey used for chapters 6 and 7

The project seeks to understand your preferences for dealing with conservation matters on your land and the financial, biological and political reasons for this. As such, we are trying to reach all landholders in the Daly Catchment (extending from the coast, south-west of Darwin, about 250 km inland) with land parcels of at least 10ha.

As you are aware, the clearing moratorium in the Daly area as well as the land clearing cap has maintained much of the native vegetation. However, changing climate and fire regimes, as well as increased invasive weeds and feral animals pose a large threat to native vegetation and are causing losses of native vegetation and animals. In order to maintain healthy native systems your aid is necessary to control invasive species and continue healthy fire regimes.

We hope that our survey will help answer important questions such as:

- What would best reflect your preferences for conservation – payment schemes to support conservation management agreements or to establish conservation covenants?
- What support you would need to be able to engage in the preferred conservation actions?

We have attached the questionnaire to this letter. We would be very grateful if someone in the household could fill it in (only one response is needed), RETAIN THIS COVER LETTER (so that your response remains confidential) and then mail it back to us in the enclosed reply-paid envelope. As noted in our earlier letter, all information which we collect will be kept **strictly confidential**. No information will be attributed to any single person or household, and results will only be released in aggregate form. Responses to the survey will be stored separately from the names and addresses of households, so that no link can be made between them.

Should you have any queries about the project or if you are interested in seeing the results of our survey please do not hesitate to contact me.

On behalf of the JCU research team, thank you in advance for your assistance.

Yours sincerely,

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CURRENT LAND USE AND MANAGEMENT

First, we would like to find out about how you use your land and what your current land use management practices are.

1. Are you the owner of the property or the manager?
☐ Owner ☐ Other *(please specify)*.....

2. If you are the owner or manager of multiple properties, how many properties are you referring to in your responses for this survey *(please indicate the number of properties, and if available something that would identify those properties such as the address or the 'laiskey' of the properties).*
Number (and address or 'laiskey')*)*.....

3. Are you willing to answer a survey individually for each of the properties that you indicated you manage in question 2?
- ☐ I would like a separate survey to complete for each property, please send me additional surveys.
- ☐ I would like to answer the survey in aggregate for all my properties, do not send me additional surveys.

If you have indicated that you will answer a survey for each property, please indicate the address or 'laiskey' of the property you refer to for the rest of the survey .....

4. How long have you (or your family) been running this property? *Please indicate years.*
Years.....

5. What is your age?
☐ <30 ☐ 30-39 ☐ 40-49 ☐ 50-59 ☐ 60-69 ☐ >70

6. What is your gender?
☐ Male ☐ Female

7. What tenure is the property under?
☐ freehold ☐ leasehold ☐ other *(please specify)*.....

8. How many people are working on the property full time and part time (including contract)?
How many are family members (including yourself)?

In total: Full time..... Part time..... Part time hrs/wk *(Average hrs/week per part time staff)* .....

Family: Full time..... Part time..... Part time hrs/wk *(Average hrs/week per part time staff)* .....

9. In general, would you like to see your land maintained:
☐ In its current state ☐ In a more natural state ☐ In a more developed state

10. In general would you like to see the Daly catchment as a whole managed at:
☐ In its current state ☐ In a more natural state ☐ In a more developed state

11. What is the approximate size of your property? *Please indicate the area measure you use.*
Size.....hectares acres square km square miles

12. Approximately how much of your property is used for grazing? *Please indicate the area measure you use.*
Grazing area.....hectares acres square km square miles

And of the land used for grazing approximately how much occurs on land that is (use the same area measure used above):
Bush and unimproved country.....
Cleared land with natural pasture.....
Cleared land with introduced pasture or fodder crops.....

13. Approximately how much of your property is used for intensive agricultural use (such as dry land, irrigated and fallow crops or other horticultural uses such as plantations)? *Please indicate the area measure you use.*

Area.....hectares acres square km square miles
And of the land used for agriculture approximately how much area is used for each of the following types of intensive use (use the same area measure used above):
Dry land Crops.....
Irrigated Crops.....
Follow Crops.....
Forest Plantation.....

14. Approximately how much of your property is used for tourism? *Please indicate the area measure you use.*

Area.....hectares acres square km square miles
And of the land used for tourism how much area is used for each of the following types of intensive use (use the same area measure used above):
Native vegetation.....
Land unsuitable for any agricultural use.....
Land already cleared and used.....

15. Approximately how much of your property is used for other uses (e.g. residential or other industries not already mentioned)? *Please indicate the area measure you use.*

Area.....hectares acres square km square miles
Please list how land is used..... .....

16. Do any of the following pose threats to your land? If not, please indicate 0 for "Not Present". If present please indicate the prominence of each threat with 1 being low threats and 5 being a large threat to the entire property:

	Not Present	Threatens only some of property		Threatens about half of the property		Threatens entire property
	0	1	2	3	4	5
Uncontrolled fires	☐	☐	☐	☐	☐	☐
Declared Weeds* such as *Mimosa pigra*	☐	☐	☐	☐	☐	☐
Other weeds	☐	☐	☐	☐	☐	☐
Feral animals (pigs, buffalo etc.)	☐	☐	☐	☐	☐	☐
Unauthorised Access (hunting etc.)	☐	☐	☐	☐	☐	☐

*the Northern Territory declared weed list is available at:
http://www.nt.gov.au/nreta/natres/weeds/legislation.html

We know that there are laws which require you to undertake certain actions to manage your property. We are interested in asking you more about how much it costs you and how long it takes you to undertake these actions to meet these obligations. Examples of actions you may be required to take to maintain your land are:

Minimum requirements	• Control weeds • Control feral animals • Control erosion • Minimise fire hazards

We would also like to know about any extra environmental actions that you take that go above and beyond your legislative obligations. For example this could be to prevent and mitigate threats to natural values before they are actually being damaged. Actions that may be beyond requirements are, for example:

Actions beyond minimum requirements	• Fencing to protect rainforest patches and watercourses • Revegetation of threatened plants • Weed control for non declared weed species • Strategic and extensive feral animal control efforts • Patch burning in bush areas to minimise risk of large fires and promote regeneration of native species

17. How would you describe the time/effort required to meet the minimum required land management obligations?

- ☐ Requires little extra work
- ☐ Is a moderate amount of hours and dollars
- ☐ Is too demanding, both in hours and dollars, to meet standards

18. On average how many days a month do you dedicate to managing your land to meet these required management obligations?

☐ <1 ☐ 1-2 ☐ 3-7 ☐ >7

19. On average how much do you spend in a year on managing your land to meet these management obligations? Please answer in dollars for the itemized list below:

Item	Amount ($) and any details you can provide
Supplies (regular operation costs such as fuel, pesticides and fencing materials such as timber and wire)	
Labor (wages and salaries that reflect your time, your employee's time and hired services such as culling of feral animals)	
Other cost (please specify, e.g. replacement of worn-out equipment and investment in new facilities)	

20. How often do you undertake the following to manage your land:

	Frequency				
	Never	Weekly	Monthly	Quarterly	Annually
Back burn away from areas that are fire sensitive	☐	☐	☐	☐	☐
Control weedy grasses (such as Mission Grass) that increase fuel loads	☐	☐	☐	☐	☐
Control other type of weeds with fire	☐	☐	☐	☐	☐
"Cool burn" early in dry season or after first wet storm to break up fuel layer (indicate whether you undertake (annually) or not (never))	☐				☐
Control weeds mechanically	☐	☐	☐	☐	☐
Control weeds with chemicals	☐	☐	☐	☐	☐
Use integrated or biological pest control to remove weeds	☐	☐	☐	☐	☐
Cull feral animals	☐	☐	☐	☐	☐

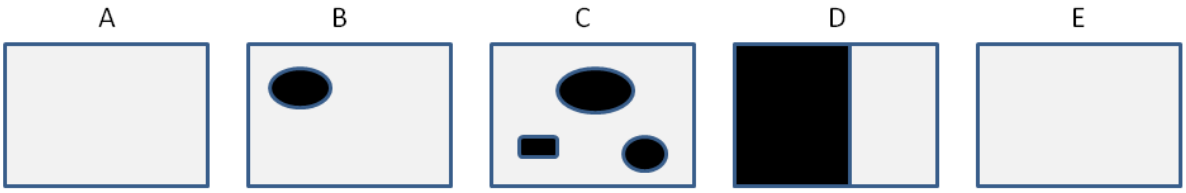
21. We understand that a common method for protecting land against feral animals is to put up fencing. Please indicate the percentage of your land that you have fenced off to protect against feral animals: .....

22. If there are actions you take to manage your land beyond legislative obligations please list them and the approximate cost of undertaking such actions:

Item	Amount ($) and any details you can provide including breakdown of labor, operational costs such as fuel or other supplies, maintenance of existing facilities or investment in new facilities
For example, you may have fenced off rainforest patches on your property	S$ in fencing supplies T$ in paid labor to construct fences Y hrs supervising and aiding in construction of fences Z hrs annually in fence maintenance

23. Aside from areas that you may be required to leave uncleared such as riparian strips along creeks or wetlands, do you have native vegetation or other natural features that you are managing, including required management actions and environmental actions beyond the minimum required? What spatial configuration below best describes the placement of these areas on your land? *Please refer to the diagrams and indicate what spatial configuration is most representative. Note grey indicates your property and black indicates areas being managed. If you feel that A – D are not applicable please use E to draw the configuration that applies to your property.*

- ☐ A) None
- ☐ B) Limited and patchy
- ☐ C) Extensive and patchy
- ☐ D) Extensive and continuous
- ☐ E) None of these are applicable, I will draw my own in



Now, we would like to ask about your familiarity with different land management options. For your reference, conservation covenants and conservation management agreements are described below:

A **conservation covenant** is a legally binding agreement (similar to a power line easement) that is held with the land title. The covenant is voluntarily entered into, and includes land management restrictions that are aimed to protect special natural values on the property (such as a population of threatened species). Land uses such as grazing may be compatible with the covenant depending on the requirements of the special natural values the covenant is to protect. Covenants can cover either all or a specific portion of a land title. There are no native title implications as the agreement is not a subdivision and tenure does not change.

A **conservation management agreement (CMA)** is a contract or binding agreement regarding the use and management of a portion of land that has special natural values. A CMA specifies conservation actions and or outcomes that you agree. *A CMA differs from a conservation covenant in that it is not placed against the land title and thus does not bind future owners of the property if it is to change hands.*

24. Please indicate your familiarity with each of the management options, from "not at all" to "very":

	Not familiar				Very familiar
	1	2	3	4	5
Current land management obligations	☐	☐	☐	☐	☐
Conservation Management Agreement	☐	☐	☐	☐	☐
Conservation Covenant	☐	☐	☐	☐	☐

We would like to ask you a little bit more about what capacity, infrastructure, or financial support you need to undertake increased conservation management on your land.

If government funding was available to run conservation schemes the payments provided to you, under a legal agreement, to support specific conservation actions may be in many forms such as providing:

- People (e.g. **Indigenous Rangers or Conservation Managers**) that will undertake management actions such as controlled burning on your land
- **Supplies** such as chemicals for weed control or fencing materials
- Public recognition (such as **Conservation Certification**) for your property
- **Offsets** such as increased clearing rights
- **Financial payments**

25. Please indicate your interest in each of the management options, from "would not consider" to "very":

	Would not consider 0	1	2	3	4	Would be very interested 5
Indigenous Rangers or Conservation Managers	☐	☐	☐	☐	☐	☐
Supplies	☐	☐	☐	☐	☐	☐
Conservation Certification	☐	☐	☐	☐	☐	☐
Offsets	☐	☐	☐	☐	☐	☐
Financial Support	☐	☐	☐	☐	☐	☐

CHOICE SETS

Finally, we would like to find out more about your preferences with respect to conservation arrangements and compensation for undertaking conservation on your property.

We will do this by presenting you with 10 different HYPOTHETICAL situations. In each situation, you will be told what the HYPOTETICAL aim/goal of the government is - and will be shown a picture of how this would affect your HYPOTHETICAL property. You will then be given a set of options, namely to:

a) agree to change your property, using a CONSERVATION CONVENANT – and receiving the compensation specified in the hypothetical example

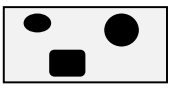
b) agree to change your property, using a CONSERVATION MANAGEMENT AGREEMENT – receiving the level of compensation specified in the example; or

c) sell the entire property – at the going market rate.

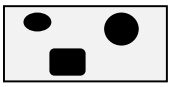
For each of the 10 different hypothetical situations, we ask you to indicate which of the options (agree to a covenant; negotiate a management agreement; or sell) you would select. *To indicate your choice, simply tick the box next to the option you select. Please make sure to indicate your selection for each of the 10 sets and consider only the options presented in a set to determine your preference.*

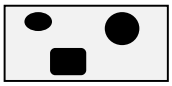
Example choice set:

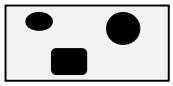
Imagine that the government would like you to set aside two more 'patches' of land on your property for conservation purposes. This would change the configuration of your land

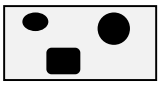
From:  **where there is one small patch set aside for conservation**	***To:***  **where there are several patches set aside for conservation**	
Would you choose to	**Payment (as a % of Total Costs)**	**Choice** ✓
Accept a **Conservation covenant**, that would require you to pay for the survey costs, and then spend 1-2 days per month 'managing' the extra conservation areas	*and* receive compensation for **50%** of all costs	☑
Or Accept a **Conservation management agreement**, that would require you to purchase some extra supplies (e.g. fencing) and labour (to put the fences in) and that would require you to spend an extra 1-2 days per month 'managing' the areas	*and* receive compensation for **50%** of all costs	☐
Or **Sell your entire property at market value**	Market value	☐

You would consider the options presented here and select ONE of the four options by ticking the appropriate box. If, you prefer the conservation covenant, then you would tick the second box.

1. Imagine that the government would like you to set aside two more 'patches' of land on your property for conservation purposes. This would change the configuration of your land		
From:  **where there is one small patch set aside for conservation**	***To:***  **where there are several patches set aside for conservation**	
Would you choose to	**Payment (as a % of Total Costs)**	**Choice** ✓
Accept a **Conservation covenant**, that would require you to pay for the survey costs, and then spend 1-2 days per month 'managing' the extra conservation areas	*and* receive compensation for **0%** of all costs	☐
Or Accept a **Conservation management agreement**, that would require you to purchase some extra supplies (e.g. fencing) and labour (to put the fences in) and that would require you to spend an extra 1-2 days per month 'managing' the areas	*and* receive compensation for **0%** of all costs	☐
Or **Sell your entire property at market value**	Market value	☐

2. Imagine that the government would like you to set aside several 'patches' of land on your property for conservation purposes. This would change the configuration of your land		
From:  **where there is no land set aside for conservation**	***To:***  **where there are several patches set aside for conservation**	
Would you choose to	**Payment (as a % of Total Costs)**	**Choice** ✓
Accept a **Conservation covenant**, that would require you to pay for the survey costs, and then spend 1-2 days per month 'managing' the extra conservation areas	*and* receive compensation for **150%** of all costs	☐
Or Accept a **Conservation management agreement**, that would require you to purchase some extra supplies (e.g. fencing) and labour (to put the fences in) and that would require you to spend an extra 1-2 days per month 'managing' the areas	*and* receive compensation for **100%** of all costs	☐
Or **Sell your entire property at market value**	Market value	☐

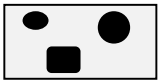
3. Imagine that the government would like you to set aside several 'patches' of land on your property for conservation purposes. This would change the configuration of your land		
From:  **where there is no land set aside for conservation**	***To:***  **where there are several patches set aside for conservation**	
Would you choose to	**Payment (as a % of Total Costs)**	**Choice** ✓
Accept a **Conservation covenant**, that would require you to pay for the survey costs, and then spend 1-2 days per month 'managing' the extra conservation areas	*and* receive compensation for **100%** of all costs	☐
Or Accept a **Conservation management agreement**, that would require you to purchase some extra supplies (e.g. fencing) and labour (to put the fences in) and that would require you to spend an extra 1-2 days per month 'managing' the areas	*and* receive compensation for **0%** of all costs	☐
Or **Sell your entire property at market value**	Market value	☐

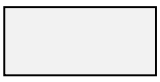
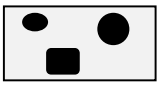
4. Imagine that the government would like you to set aside a continuous section of land on your property for conservation purposes. This would change the configuration of your land		
From:  **where there are several patches set aside for conservation**	***To:***  **where there is one continuous area set aside for conservation**	
Would you choose to	**Payment (as a % of Total Costs)**	**Choice** ✓
Accept a **Conservation covenant**, that would require you to pay for the survey costs, and then spend 1-2 days per month 'managing' the extra conservation areas	*and* receive compensation for **150%** of all costs	☐
Or Accept a **Conservation management agreement**, that would require you to purchase some extra supplies (e.g. fencing) and labour (to put the fences in) and that would require you to spend an extra 1-2 days per month 'managing' the areas	*and* receive compensation for **0%** of all costs	☐
Or **Sell your entire property at market value**	Market value	☐

5. Imagine that the government would like you to set aside one 'patch' of land on your property for conservation purposes. This would change the configuration of your land

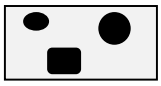
From:  **where there is no land set aside for conservation**	***To:***  **where there is one small patch set aside for conservation**	
Would you choose to	**Payment (as a % of Total Costs)**	**Choice** ✓
Accept a **Conservation covenant**, that would require you to pay for the survey costs, and then spend 1-2 days per month 'managing' the extra conservation areas	*and* receive compensation for **50%** of all costs	☐
Or Accept a **Conservation management agreement**, that would require you to purchase some extra supplies (e.g. fencing) and labour (to put the fences in) and that would require you to spend an extra 1-2 days per month 'managing' the areas	*and* receive compensation for **50%** of all costs	☐
Or **Sell your entire property at market value**	Market value	☐

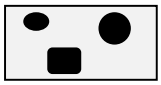
6. Imagine that the government would like you to set aside a continuous section of land on your property for conservation purposes. This would change the configuration of your land

From:  **where there are several patches set aside for conservation**	***To:***  **where there is one continuous area set aside for conservation**	
Would you choose to	**Payment (as a % of Total Costs)**	**Choice** ✓
Accept a **Conservation covenant**, that would require you to pay for the survey costs, and then spend 1-2 days per month 'managing' the extra conservation areas	*and* receive compensation for **150%** of all costs	☐
or Accept a **Conservation management agreement**, that would require you to purchase some extra supplies (e.g. fencing) and labour (to put the fences in) and that would require you to spend an extra 1-2 days per month 'managing' the areas	*and* receive compensation for **50%** of all costs	☐
or **Sell your entire property at market value**	Market value	☐

7. Imagine that the government would like you to set aside several 'patches' of land on your property for conservation purposes. This would change the configuration of your land		
From:  **where there is no land set aside for conservation**	***To:***  **where there are several patches set aside for conservation**	
Would you choose to	**Payment (as a % of Total Costs)**	**Choice** ✓
Accept a **Conservation covenant**, that would require you to pay for the survey costs, and then spend 1-2 days per month 'managing' the extra conservation areas	*and* receive compensation for **100%** of all costs	☐
or Accept a **Conservation management agreement**, that would require you to purchase some extra supplies (e.g. fencing) and labour (to put the fences in) and that would require you to spend an extra 1-2 days per month 'managing' the areas	*and* receive compensation for **100%** of all costs	☐
or **Sell your entire property at market value**	Market value	☐

8. Imagine that the government would like you to set aside a continuous section of land on your property for conservation purposes. This would change the configuration of your land		
From:  **where there is no land set aside for conservation**	***To:***  **where there is one continuous area set aside for conservation**	
Would you choose to	**Payment (as a % of Total Costs)**	**Choice** ✓
Accept a **Conservation covenant**, that would require you to pay for the survey costs, and then spend 1-2 days per month 'managing' the extra conservation areas	*and* receive compensation for **50%** of all costs	☐
or Accept a **Conservation management agreement**, that would require you to purchase some extra supplies (e.g. fencing) and labour (to put the fences in) and that would require you to spend an extra 1-2 days per month 'managing' the areas	*and* receive compensation for **100%** of all costs	☐
or **Sell your entire property at market value**	Market value	☐

9. Imagine that the government would like you to set aside a continuous section of land on your property for conservation purposes. This would change the configuration of your land		
From:  **where there are several patches set aside for conservation**	***To:***  **where there is one continuous area set aside for conservation**	
Would you choose to	**Payment (as a % of Total Costs)**	**Choice** ✓
Accept a **Conservation covenant**, that would require you to pay for the survey costs, and then spend 1-2 days per month 'managing' the extra conservation areas	*and* receive compensation for **100%** of all costs	☐
or Accept a **Conservation management agreement**, that would require you to purchase some extra supplies (e.g. fencing) and labour (to put the fences in) and that would require you to spend an extra 1-2 days per month 'managing' the areas	*and* receive compensation for **50%** of all costs	☐
or **Sell your entire property at market value**	Market value	☐

10. Imagine that the government would like you to set aside several 'patches' of land on your property for conservation purposes. This would change the configuration of your land		
From:  **where there is no land set aside for conservation**	***To:***  **where there are several patches set aside for conservation**	
Would you choose to	**Payment (as a % of Total Costs)**	**Choice** ✓
Accept a **Conservation covenant**, that would require you to pay for the survey costs, and then spend 1-2 days per month 'managing' the extra conservation areas	*and* receive compensation for **0%** of all costs	☐
or Accept a **Conservation management agreement**, that would require you to purchase some extra supplies (e.g. fencing) and labour (to put the fences in) and that would require you to spend an extra 1-2 days per month 'managing' the areas	*and* receive compensation for **150%** of all costs	☐
or **Sell your entire property at market value**	Market value	☐

GENERAL HOUSEHOLD INFORMATION

Lastly, we would like to know a little bit more about your household. This information will be kept strictly confidential and is used to ensure that we have collected information from a wide variety of households.

Please tick appropriate box for each question.

1. What is the total, combined, annual (taxable) income of ALL the people who 'normally' live in your house?

☐ <$20,000 AUS Dollars
☐ $20,000-$40,000 AUS Dollars
☐ $40,000-$60,000 AUS Dollars
☐ $60,000-$80,000 AUS Dollars
☐ $80,000-$100,000 AUS Dollars
☐ $100,000-$120,000 AUS Dollars
☐ $120,000-$140,000 AUS Dollars
☐ $140,000-$160,000 AUS Dollars
☐ > $160,000 AUS Dollars

2. Please indicate the percentage of your income that you derive from enterprises on your property and how much of your income is from work off of the land:

On land enterprises: .....
Off farm income: .....

3. What enterprises do you have on your property? Please indicate the percentage of your on farm income that you derive from these enterprises. *Please tick appropriate box(es)*

	Percent Income				
	0%	25%	50%	75%	100%
Cattle Grazing on:					
Bush and Unimproved Country	☐	☐	☐	☐	☐
Cleared Land with natural pasture	☐	☐	☐	☐	☐
Cleared Land with Introduced Pasture or Fodder Crops	☐	☐	☐	☐	☐
Intensive Use for:					
Dry land Crops	☐	☐	☐	☐	☐
Irrigated Crops	☐	☐	☐	☐	☐
Fallow Crops	☐	☐	☐	☐	☐
Forest Plantation	☐	☐	☐	☐	☐
Tourism					
Property Stay	☐	☐	☐	☐	☐
Access to Tour Provider	☐	☐	☐	☐	☐
Other (specify: _____)	☐	☐	☐	☐	☐
Other (Please Specify)					
	☐	☐	☐	☐	☐