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Managing natural capital assets and ecosystem services under global change

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Abstract

Human activities are placing increasing pressure on Earth's systems and finite natural resources. Climate change alters the provision of ecosystem services and natural capital, so innovative strategies are needed to adapt to these impacts. However, the formulation and implementation of such strategies is hindered by the substantial uncertainty involved in projections of climate change and the impacts this will have. This is confounded by the uncertain impacts of other drivers of change (such as varying demand and commodity prices), which can alter the demand for ecosystem service provision. To add to this challenge, ecosystem services and natural capital assets are not independent of each other, so policies targeting the provision of an individual ecosystem service (such as food production) need to consider the potential impacts they may have on other ecosystem services. I addressed this problem by developing and evaluating strategies to manage multiple ecosystem services under uncertain global drivers of change.

In ***chapter 2*** I conducted a systematic literature review of climate change impacts on ecosystem services and found that the impact of climate change on most types of services was predominantly negative, but varied across services, drivers of change, and assessment methods. Although uncertainty was usually incorporated into assessments, there were substantial gaps in the sources of uncertainty included. In addition relatively few studies integrated decision making, and even fewer studies aimed to include multiple drivers in decisions or identify solutions that were robust to uncertainty.

I then addressed decision making under climate change using a case study of conservation planning for coastal wetlands and the ecosystem services they provide under sea level rise in ***chapters 3*** and ***4***. The expansion of coastal developments can prevent potential landward wetland migration, exacerbating wetland loss as sea levels rise. Pre-emptive planning to set aside key coastal areas for wetland migration is therefore critical for the long term preservation of species habitat and ecosystem services. In ***chapter 3*** I show that the opportunity cost of preserving wetlands is likely to be much higher under sea level rise than under current sea levels. Nonetheless, payments for ecosystem services were able to alleviate these costs, but even this was hampered with higher rates of sea level rise.

I then explicitly incorporated uncertainty in sea level rise projections and modelling of wetland change into a novel problem formulation in ***chapter 4***. I integrated a risk-sensitive resource allocation framework from economics, Modern Portfolio Theory, with a conservation planning

framework. This approach allows the selection of a complementary set of connected sites that met a set of conservation objectives whilst hedging the risk of different climate change scenarios and associated uncertainties. I found that planning for specific projections of sea level rise was a relatively high risk strategy, even when planning for the most severe impacts, compared to the risk-sensitive planning approach.

Where multiple ecosystem services trade-off against each other, management strategies are needed to balance the relative provision of each ecosystem service, whilst also accounting for different global change scenarios. I exemplified this situation in ***chapter 5*** by using an integrated modelling approach to assess the impact of climate change, fire, and global economic drivers on the profitability and effectiveness of management actions for livestock production and greenhouse gas regulation in the tropical savannas of northern Australia. Emerging strategies, such as changing fire management practices or nitrate supplementation, were able to reduce greenhouse gas emissions, but they came with financial costs. However, the growing urgency to abate emissions under some global change scenarios resulted in prices for carbon that compensated for these costs in some cases.

I conclude that innovative methods are vital to successfully adapt the management of ecosystem services to the impacts of climate change and associated complexities. Although the application of such approaches are challenging, ignoring the future impacts of global change can result in the inefficient allocation of resources for climate adaptation and suboptimal management outcomes. Ideally, decision making should also incorporate deep uncertainty and ecosystem service flows to beneficiaries. However, no individual assessment or project can include every complexity, so future research should focus on which drivers, processes, and uncertainties should be prioritised for inclusion in decision making.

Declaration by author

This thesis is composed of my original work, and contains no material previously published or written by another person except where due reference has been made in the text. I have clearly stated the contribution by others to jointly-authored works that I have included in my thesis.

I have clearly stated the contribution of others to my thesis as a whole, including statistical assistance, survey design, data analysis, significant technical procedures, professional editorial advice, and any other original research work used or reported in my thesis. The content of my thesis is the result of work I have carried out since the commencement of my research higher degree candidature and does not include a substantial part of work that has been submitted to qualify for the award of any other degree or diploma in any university or other tertiary institution. I have clearly stated which parts of my thesis, if any, have been submitted to qualify for another award.

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Published:

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Contributor	***Statement of contribution***
R.K. Runting (Candidate)	Conceptualised paper (50%) Reviewed Literature (65%) Analysed data (100%) Wrote and edited paper (80%)
B.A. Bryan	Conceptualised paper (20%) Reviewed Literature (4%) Wrote and edited paper (2%)
L.E. Dee	Reviewed Literature (4%) Wrote and edited paper (3%)
F.J.F. Maseyk	Reviewed Literature (4%) Wrote and edited paper (2%)
L. Mandle	Reviewed Literature (4%) Wrote and edited paper (2%)
P. Hamel	Reviewed Literature (4%) Wrote and edited paper (2%)
K.A. Wilson	Reviewed Literature (4%) Wrote and edited paper (2%)
K. Yetka	Reviewed Literature (7%)
H.P. Possingham	Conceptualised paper (10%) Wrote and edited paper (2%)
J.R. Rhodes	Conceptualised paper (20%) Reviewed Literature (4%) Wrote and edited paper (5%)

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R.K. Runting (Candidate)	Conceptualised paper (50%) Built models (90%) Analysed data (100%) Wrote and edited paper (80%)
C.E. Lovelock	Conceptualised paper (20%) Built models (5%) Wrote and edited paper (5%)
H.L. Beyer	Built models (5%) Wrote and edited paper (5%)
J.R. Rhodes	Conceptualised paper (30%) Wrote and edited paper (10%)

Runting, RK, Meijaard, E, Abram, NK, Wells, JA, Gaveau, DLA, Ancrenaz, M, Possingham, HP, Wich, SA, Ardiansyah, F, Gumal, MT, Ambu, LN, & Wilson, KA. 2015. Alternative futures for Borneo show the value of integrating economic and conservation targets across borders. *Nature Communications*. 6:6819. – incorporated as Appendix A

Contributor	***Statement of contribution***
R.K. Runting (Candidate)	Conceptualised paper (40%) Reviewed literature (50%) Produced spatial data (65%) Built models (100%) Analysed data (100%) Wrote and edited paper (55%)
E. Meijaard	Conceptualised paper (20%) Reviewed literature (10%) Wrote and edited paper (10%)

N.K. Abram	Conceptualised paper (10%) Reviewed literature (5%) Produced spatial data (15%) Wrote and edited paper (4%)
J.A. Wells	Conceptualised paper (10%) Reviewed literature (10%) Wrote and edited paper (4%)
D.L.A Gaveau	Produced spatial data (15%) Wrote and edited paper (2%)
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H.P. Possingham	Conceptualised paper (10%) Wrote and edited paper (2%)
S.A. Wich	Produced spatial data (5%) Wrote and edited paper (2%)
F. Ardiansyah	Reviewed literature (5%) Wrote and edited paper (2%)
M.T. Gumal	Reviewed literature (5%) Wrote and edited paper (2%)
L.N. Ambu	Wrote and edited paper (2%)
K.A. Wilson	Conceptualised paper (10%) Reviewed literature (10%) Wrote and edited paper (13%)

Contributions by others to the thesis

This thesis consists of five manuscripts that have either been published or are intended for submission for publication, with myself as the lead author. Chapters 2, 3 and Appendix A have been published, and chapters 4 and 5 will be submitted for publication in due course. Chapters 2-5 and all appendices are written plural first-person pronoun “we”/”our”, to reflect the contributions from others. In chapters 1 and 6, I use the singular first-person pronoun “I”/”my” as these were written entirely by me (with editorial input from my supervisors).

Chapter 1

This chapter was written by myself, with editorial input from Jonathan Rhodes, Brett Bryan and Hugh Possingham.

Chapter 2

Contributions to this chapter are detailed in the preceding “Publications included in this thesis” section.

Chapter 3

Contributions to this chapter are detailed in the preceding “Publications included in this thesis” section.

Chapter 4

This chapter is being prepared for submission to *Global Environmental Change*. The idea for the manuscript was conceptualised by Jonathan Rhodes and myself. I developed the problem formulation, with advice from Yann Dujardin, Jonathan Rhodes, and Hawthorne Beyer. Catherine Lovelock provided advice and data for the parameterisation of the SLAMM model. I conducted all analyses, with feedback on results from Jonathan Rhodes, Hawthorne Beyer, Brett Bryan and Yann Dujardin. I wrote the chapter, with editorial input from Hawthorne Beyer, Jonathan Rhodes, Catherine Lovelock, and Brett Bryan.

Chapter 5

This chapter is being prepared for submission to *Ecosystem Services*. The idea for the manuscript was conceptualised by Brett Bryan, Jonathan Rhodes and myself. The modelling of fire severity and intensity was conducted by Martin Nolan, with input from Brett Bryan and myself. The modelling of pasture production was undertaken by Darran King and myself, with input from Brett Bryan and

Raymundo Marcos-Martínez. Javier Navarro provided baseline economic data. All other modelling and analysis was conducted by myself, with advice from Brett Bryan, Martin Nolan, Darran King, Raymundo Marcos-Martínez, Jeff Connor and Jonathan Rhodes. The chapter was written by myself, with editorial input from Brett Bryan, Jonathan Rhodes, Jeff Connor, Marin Nolan, and Javier Navarro.

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Appendix A

Contributions to this chapter are detailed in the preceding "Publications included in this thesis" section.

Statement of parts of the thesis submitted to qualify for the award of another degree

None.

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Keywords

Ecosystem services, climate change adaptation, sea level rise, carbon sequestration, conservation planning, coastal wetlands, savanna, livestock, uncertainty, global change.

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Table of Contents

Figures and Tables

Figures:

Tables:

Abbreviations

AUD – Australian Dollars

CanESM - Canadian Earth System Model

CL – Conventional logging

DEM – Digital Elevation Model

DERM – Queensland Government Department of Environment and Resource Management

DAFF – Queensland Government Department of Agriculture, Fisheries and Forestry

GCM – General Circulation Model

GHG – Greenhouse Gas

GVP – Gross Value of Production

ILP – Integer Linear Programming

IPCC – Intergovernmental Panel on Climate Change

ITP – Industrial timber plantations

IUCN – International Union for the Conservation of Nature

LiDAR – Light Detection and Ranging

MIROC5 - Model for Interdisciplinary Research on Climate version 5

MPI-ESM-LR - Max Planck Institute – Earth System Model – Low Resolution

NPV – Net Present Value

PES – Payments for Ecosystem Services

QLD – Queensland

RBA – Reserve Bank of Australia

RCP – Representative Concentration Pathway

RIL – Reduced impact logging

RMSE – Root mean square error

SLAMM – Sea Level Affecting Marshes Model

WWF – World Wide Fund for Nature

1 Introduction

This thesis develops and evaluates strategies to manage multiple ecosystem services under global change. Natural capital assets encompass the soil, water, atmosphere and ecosystems, and provide flows of goods and services of benefit to humans (referred to as ecosystem services) (Daily 1997). Yet the activities of humans are having a substantial impact on Earth's systems (Steffen *et al* 2015). Climate and land use change alters the provision of ecosystem services and natural capital (Nelson et al. 2013), so innovative strategies are needed to adapt to these impacts (Poiani *et al* 2010). However, the formulation and implementation of such strategies is hindered by the substantial uncertainty involved in projections of climate change and the impacts this will have (Hallegatte 2009). This is also confounded by the uncertain impacts of other drivers of change (such as varying demand and commodity prices), which can alter the demand for ecosystem service provision (Bryan 2013). Quantifying these effects is not only important for determining the range of impacts on ecosystem services, but is especially important in the context of decision making (Polasky *et al* 2011). Ignoring these effects could result in misleading assessments of the impacts of climate change, or sub-optimal decision making outcomes.

To determine the effectiveness of management actions aimed at preserving natural capital assets and ecosystem services, it is necessary to understand the relationships between these assets and services, and how they are affected by external drivers. For instance, biodiversity underpins and interacts with essential ecosystem functions that support human activities (Mace *et al* 2012), but at the same time, human population and economic growth, coupled with climatic change and natural resource depletion, are likely to place increasing demands on the Earth's finite natural resources and ecosystems (Foley *et al* 2005, Liu *et al* 2007). These interactions among services and drivers can have a significant impact on the effectiveness of decisions concerning their management (Carpenter *et al* 2009). I develop a conceptual framework (Figure 1.1) to describe the relationship between natural capital and ecosystem services, and how they are affected by global drivers and management strategies.

This review and synthesis section is divided into five sub-sections to describe the conceptual framework (Figure 1.1). The first section, 'natural capital and ecosystem services', defines these terms, whilst discussing the linkages between them. The second section 'external drivers' discusses the impact of climate change and economic drivers on natural capital and the supply of ecosystem services. The 'multiple objectives, trade-offs, and co-benefits' discusses the issues arising from multiple competing objectives. The section on ‘management strategies’ describes the dominant policy options for managing natural capital and ecosystem services. The final section identifies the key research gaps and outlines the objectives of this thesis.

1.1 LITERATURE REVIEW AND SYNTHESIS

1.1.1 Natural capital and ecosystem services

Natural capital is defined as ‘the stock of natural resources or environmental assets (such as soil, water, atmosphere, and ecosystems) that provide a flow of useful goods or services, now or in the future’ (*sensu* De Groot et al. 2003; Van Dieren 1995; Pearce & Turner 1990; Daly 1994). Whilst natural capital can include abiotic components (such as minerals), much of the flow of goods and services are derived from ecosystems. The benefits that flow from the biotic components of natural capital are known as ecosystem services (Costanza et al. 1997). For example, the aerial root systems of mangroves (the natural capital asset) provide a sheltered environment that serves as a nursery habitat for many commercially important fish species (the ecosystem service) (Nagelkerken *et al* 2008). Other examples of ecosystem services include greenhouse gas regulation, erosion prevention, the provision of food and fibre, temporary storage of flood waters by wetlands, assimilation of wastes, and many others (Costanza et al. 2006). This thesis will focus on both the biotic components of natural capital, and the ecosystem services they provide (Figure 1.1).

However, the distinction between natural capital assets and ecosystem services is not always clear. For example, Hawken et al. (1999) state that natural capital is “… made up of resources, living systems, and ecosystem services”. Alternatively, the Millennium Ecosystem Assessment (2005) includes a ‘supporting’ ecosystem service category which are “… necessary for the production of all other ecosystem services”, but do not provide direct benefits to humans. The ‘supporting services’ category was replaced by ‘habitat services’ in the more recent report by The Economics of Ecosystems and Biodiversity initiative (TEEB 2010b), although these ‘habitat services’ also do not provide direct benefits to humans. Here, I consider biodiversity (along with ecosystems and habitat for species) to be separate from ecosystem services, which is in line with the definitions provided above, and along with many other studies (Benayas et al. 2009; Worm et al. 2006; Goldman et al. 2008; Maynard et al. 2010; Costanza et al. 1997; Costanza et al. 2006).

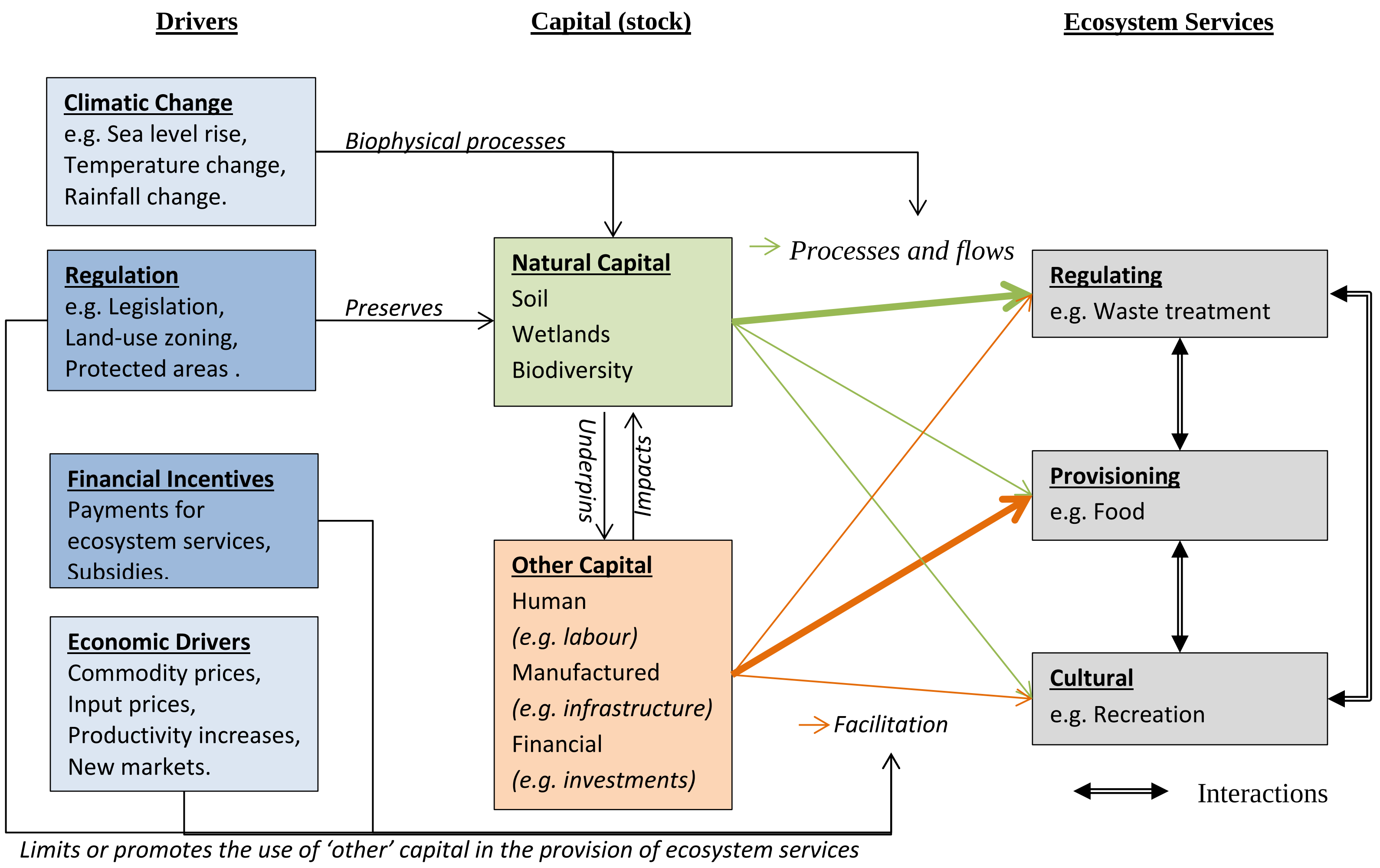


Figure 1.1 | A conceptual framework for the relationship between natural capital, ecosystem services, global change, and management actions (dark blue). The list of drivers is not exhaustive: other drivers, such as food web dynamics or voluntary incentives, can also influence the provision of ecosystem services.

Whilst natural capital is clearly necessary for the provision of ecosystem services, many of these services also require inputs from other types of capital to facilitate their use by humans (Fisher et al. 2008) (Figure 1.1). For example, food production not only requires natural capital assets (such as soil), but also uses human capital (i.e. labour) and manufactured capital (i.e. machinery to harvest crops). Natural capital underpins these other types of capital (such as human, manufactured and financial capital), by providing the essential Earth systems functions in which they operate (Hawken *et al* 1999, Chiesura and de Groot 2003, De Groot *et al* 2003) (Figure 1.1). These other types of capital can have a detrimental impact on natural capital (e.g. land clearing), but may also enhance natural capital through activities such as ecosystem restoration (Haines-Young *et al* 2006).

1.1.2 Global drivers of change

Human activities are placing increasing stress on natural systems through multiple pathways, including agricultural expansion, natural resource depletion, and accelerating climatic change (Steffen *et al* 2015, Maxwell *et al* 2016). These anthropogenic drivers operate across all spatial scales from global (e.g. climate change) to local (e.g. point source pollution) and are often interrelated (Liu *et al* 2015a). For example, global increases in prices for wildlife, alongside growing relative poverty, can drive local poaching efforts and subsequent population deciles for high-value species (Challender and MacMillan 2014). Whilst the focus of this thesis is on global change, specifically climatic and economic drivers (Figure 1.1), considering the interactions with other key drivers is still important in many contexts.

Climate change can impact the distribution of natural capital assets, whilst also altering the biophysical processes that produce ecosystem services (Harley *et al* 2006, Mooney *et al* 2009) (Figure 1.1). For example, climate change can cause sea levels to rise, which alters the distribution of coastal wetlands (a natural capital asset) (Craft *et al* 2009, Aiello-Lammens *et al* 2011, Traill *et al* 2011, Runting *et al* 2013). These wetlands can be lost if their tolerance for inundation is exceeded, but they can also be replaced by other wetlands, or migrate landward in the absence of steep gradients in topography or anthropogenic barriers, such as built structures (Traill *et al* 2011). Climate change can also impact the waste assimilation capacity of freshwater and marine ecosystems, leading to an increase in harmful cyanobacterial blooms (Paerl and Paul 2012). Alternatively, elevated atmospheric CO_2 concentrations can alter soil microbial communities, which can affect nitrogen availability, leading to a decrease in agricultural yields (although these yields may be maintained with higher rates of nitrogen fertiliser application) (Jackson et al. 2008).

Consequently, when managing natural capital assets, it is important to consider both the potential change in distribution of these assets, along with impacts on the processes that provide ecosystem services.

Given the influence of human, manufactured and financial capital in facilitating the provision of ecosystem services (Figure 1.1), it is also important to consider the economic factors that drive the relative allocation of these inputs. For example, population growth increases the demand for agricultural commodities (Foley et. al 2005), which may facilitate the expansion or intensification of the food provision ecosystem service. Alternatively, rises in the cost of farm inputs reduces the profitability of farming enterprises (*ceteris paribus*) which may lead to a decline in this service (Bryan 2013). Whilst these drivers mainly affect provisioning services (such as timber, fibre or food production), external economic drivers can also impact the supply of other ecosystem services. New and emerging markets may increase the provision of the ecosystem service it is trading, particularly if the market involves direct payment for service provision (Kinzig *et al* 2011). For example, the carbon market is likely to increase the area of plantations to supply carbon credits, if carbon is priced sufficiently high (Hunt 2008a). Whilst such markets can deliver co-benefits in terms of other non-marketed ecosystem services, they can also drive trade-offs. For example, given that monoculture plantations are more cost-effective at storing carbon than biodiverse plantations, the carbon market may have a negative impact on biodiversity (Lindenmayer *et al* 2012). Consequently, it is critical that we understand how the cumulative impact of multiple drivers affects natural capital, ecosystem services and the relationships among them, so that we can effectively manage these assets.

1.1.3 Multiple objectives, trade-offs, and co-benefits

When making decisions for preserving natural capital and ecosystem services, it is unusual to have only one objective, particularly if the interests of diverse stakeholders are included (Lahdelma *et al* 2000, Berkes 2007). The preferences and goals of different stakeholder groups are often divergent (King *et al* 2015), which can lead to decision makers seeking to achieve the provision of multiple competing ecosystem services. In the context of land use planning, simultaneously providing the desired level of these ecosystem services in the landscape may not always be possible, due primarily to constraints in the amount of land available. This may result in compromises between objectives (Krcmar *et al* 2005), or dominance of the most profitable ecosystem services (such as food production) to the detriment of others (Foley *et al* 2005).

Ecosystem services and natural capital assets are rarely perfectly correlated across the landscape (Anderson *et al* 2009), so any decisions involving multiple ecosystem services are likely to involve some degree of trade-offs between services. Trade-offs can be driven by a variety of different processes, depending on the individual ecosystem services in question (Bennett *et al* 2009, Howe *et al* 2014). For example, increasing the ecosystem service of food production by intensifying nitrogen fertiliser or pesticide application in catchments draining to the Great Barrier Reef, Australia, causes declines in water quality, which can subsequently impact coral reef ecology (Waterhouse *et al* 2012). This can in turn cause declines in recreational (e.g. reef tourism) and fisheries ecosystem services, which are both dependent on the quality of coral reefs (Butler *et al* 2013).

Where the relationships in ecosystem services stemming from natural capital are highly correlated, management objectives should ideally be focused on producing co-benefits (Chan *et al* 2011). For example, increasing the area of mangroves will also lead to increases in the waste assimilation, carbon sequestration, storm protection and fisheries maintenance services they provide (Barbier *et al* 2008). Alternatively, while planting for erosion control (i.e. on steep slopes) can be broadly beneficial for biodiversity (Brambilla *et al* 2017), these co-benefits can be reduced if non-native species are used (Cao *et al* 2009). Ideally, multiple objectives would be considered when choosing the composition of plantings (Talema *et al* 2017), or designing a payment scheme to incentivise plantings (Bryan *et al* 2016b). Consequently, even where co-benefits can arise from the management of a natural capital asset or ecosystem services, it can still be important to consider multiple objectives.

1.1.4 Management strategies

Strategies to manage natural capital assets and ecosystem services under global change can be divided into three broad categories; regulation (including public acquisition), financial incentives, and voluntary incentives (such as awareness raising and education) (Bengston *et al* 2004, Cocklin *et al* 2007, Ulvevadet and Hausner 2011). This thesis will focus on regulation and financial incentives (Figure 1.1). A common regulatory instrument to preserve natural capital (including biodiversity) is the designation and management of protected areas (Margules and Pressey 2000). If designed strategically, these reserve systems have the potential to be robust to the impacts of climate change (Carvalho *et al* 2011a, Thomas *et al* 2012). Similarly, sophisticated methods for broader land use zoning can account for the achievement of multiple objectives, including natural capital and ecosystem services (Pourebrahim *et al* 2011, Bateman *et al* 2013). For example, in urban planning, permitting high density residential development in a concentrated area can spare land to provide biodiversity and ecosystem services that might otherwise be lost to urban and peri-urban sprawl (Sushinsky *et al* 2013, Stott *et al* 2015). Alternatively, specific regulations can be required to address complex issues by restricting or permitting particular practices. For example, regulations have been implemented to limit nitrogen fertiliser application on sugarcane farms in catchments draining to the Great Barrier Reef, in order to preserve the reef quality (van Grieken *et al* 2013). Likewise, many tropical countries restrict the amount of timber extracted from forests through mandated cutting cycles, minimum felling diameters, and/or per-unit-area harvest intensities (Zimmerman and Kormos 2012)." These regulatory instruments may be used in isolation, but they are increasingly being complemented by incentive-based mechanisms (Moon and Cocklin 2011).

Financial incentives, which can include payments for ecosystem services and stewardship payments, are considered vital to secure the participation of production-based landholders in conservation programs on private land (Moon and Cocklin 2011). Although such payments are generally considered financial incentives, it is important to recognise that these payment schemes are ultimately driven by regulation and policy, such as China's US$50 billion scheme to pay farmers to restore natural ecosystems (Ouyang *et al* 2016). Environmental stewardship services, such as biodiversity protection or water quality enhancement, are often undersupplied by rural landholders due to weak market signals (Mann and Wüstemann 2008). Stewardship payments reward pro-environmental management actions (such as constraining farm inputs or changing farming practices) though direct monetary transfers and/or indirect credit or tax concessions (Hajkowicz and Collins 2009). On the other hand, payments for ecosystem services schemes are usually directly linked to the provision of a particular ecosystem service (such as carbon sequestration or hydrological services) or bundles of services (Farley & Costanza 2010). They have emerged as a way to address trade-offs that arise when some services have a market price (such as food and fibre) and others do not (e.g. scenic amenity or hydrological services) (Wunder 2007). In these cases, it is particularly relevant to consider the impact of external economic drivers, as changes in global food demand or the cost of farm inputs could alter the viability of these schemes (Bryan 2013).

Incorporating climate change into management decisions for ecosystem services will inevitably involve dealing with uncertainty. There is considerable uncertainty in the projections of climate change (IPCC 2014), which can impact natural capital assets, ecosystem services, and the relationships among them (Scholes 2016). Any management decision is further complicated by uncertain projections of other drivers of change (such as varying commodity or land prices), which can also alter the supply or demand for ecosystem services (Bryan, 2013). Other uncertainties, such as those related to the modelling or measurement of ecosystem services, are also potentially important to consider (Refsgaard *et al* 2007, Hamel and Bryant 2017). Designing policy and management strategies that are robust to these uncertainties and simultaneously achieve multiple objectives is a substantial undertaking (Polasky *et al* 2011). However, it is not insurmountable: these problems can potentially be solved through decision theoretic approaches, such as robust optimisation (Bertsimas and Sim 2004), info-gap theory (Regan *et al* 2005), Modern Portfolio Theory (Ando and Mallory 2012a), and threshold approaches (Lempert and Collins 2007). Yet the application of these methods to decision making for ecosystem services under climate change has been limited.

1.2 OBJECTIVES AND SIGNIFICANCE

The main innovation of this thesis is in developing and evaluating strategies to manage multiple ecosystem services under uncertain global change. Despite the importance of this topic, it remains understudied: there are no quantitative syntheses of the impact of global change on ecosystem services; competing objectives are often ignored when designing strategies; the impacts of climate change and global economic drivers are frequently overlooked; and there is a dearth of information on appropriate management strategies in this context. I review existing literature, in addition to undertaking original research chapters, in order to address these gaps (Figure 1.2). Specifically, four separate objectives are addressed:

1. To determine how climate change and other drivers have been incorporated into ecosystem service assessments and decisions (***chapter 2***).
2. To determine the extent to which the costs of strategies to preserve natural capital assets are affected by climate change and payments for ecosystem services (***chapter 3***).
3. To develop an approach to preserve natural capital assets and ecosystem services that are robust to the uncertain impacts of climate change (***chapter 4***).
4. To assess the costs and effectiveness of actions to manage ecosystem services under multiple global drivers (***chapter 5***).

To achieve objective one, I have undertaken a systematic literature review of how climate change and other drivers have been incorporated into ecosystem service assessments and decisions (Runting *et al.*, 2016, ***chapter 2***). Despite growing literature on the impacts of climate change on ecosystem services, no quantitative syntheses exist. Hence, we lack an overarching understanding of the impacts of climate change, how they are being assessed, and the extent to which other drivers, uncertainties, and decision making (i.e., actions, policies or other interventions) are incorporated. This systematic review determines the impacts of climate change on ecosystem services, whilst also establishing the methods used, the other drivers included, and how the outcomes of assessments are being incorporated into decision making.

This review is followed by three original research chapters orientated around two themes (that correspond to two different geographies): (i) protecting ecosystem services by planning for coastal wetland migration under sea level rise; and (ii) integrating multiple drivers to assess management actions for ecosystem services in rangelands (Figure 1.2). Coastal ecosystems and the services they provide are particularly vulnerable to climate change, primarily due to sea level rise (Lovelock *et al* 2015). Consequently, the application of emerging climate adaptation strategies to these coastal systems is vital to ensure the continued supply of services (theme 1) (Ruckelshaus *et al* 2013). Likewise, the capacity of rangelands to maintain livestock production is likely to be impacted by changes in temperature, rainfall, and fire (Lohmann *et al* 2012). At the same time, changing livestock and carbon prices, could affect the viability of these production systems and potential emission abatement actions (theme 2) (Thornton 2010).

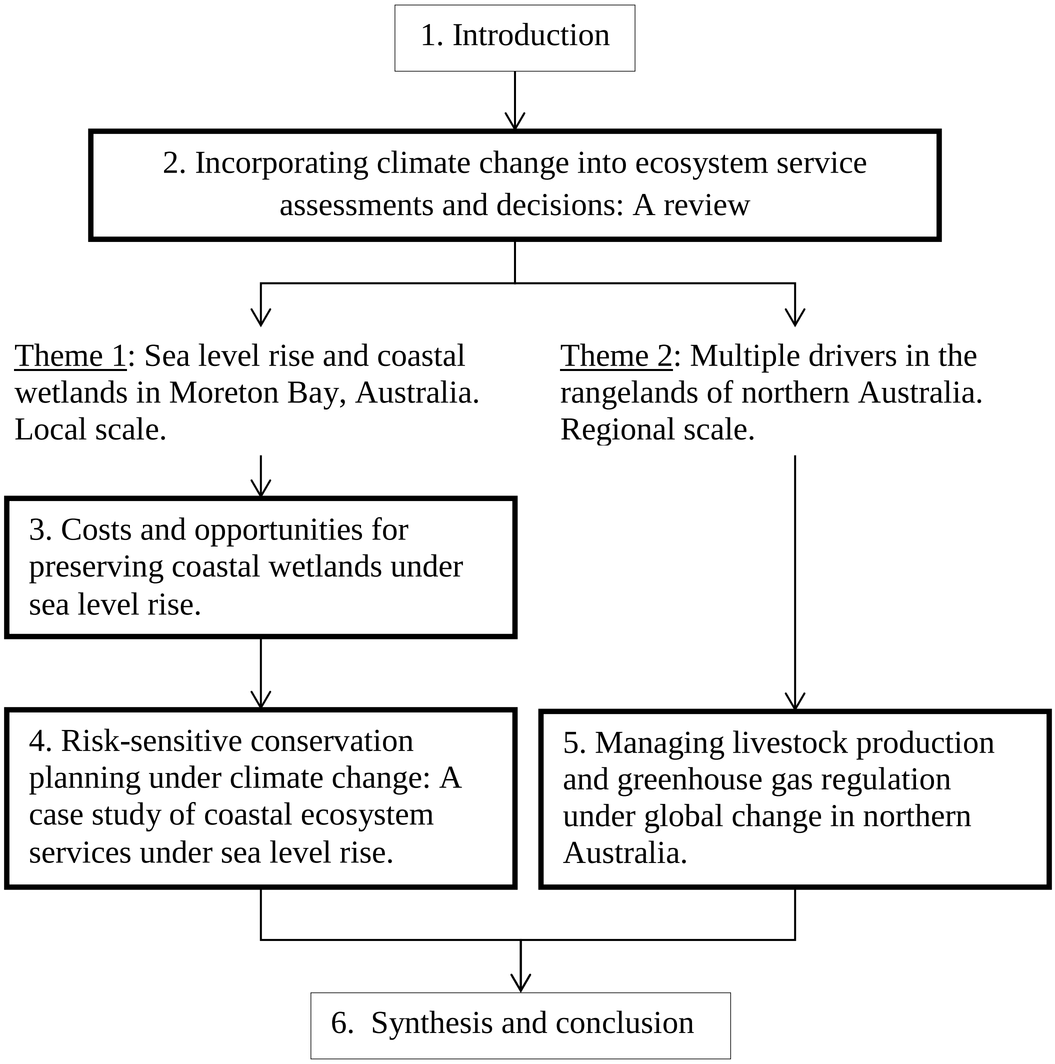


Figure 1.2 | Flowchart of thesis structure. Boxes indicate chapters and boxes outlined in bold indicate chapters that contain analyses.

Where the ecosystem services provided by natural capital assets are highly correlated, it is beneficial to preserve these assets to ensure the continued provision of services (Naidoo et al. 2008; Turner et al. 2007). Whilst this seems straight-forward, the combination of high opportunity costs and uncertain impacts of climate change can make this process challenging. The first theme explores cost-effective strategies to manage natural capital assets and ecosystem services under climate change, using the coastal ecosystem services of Moreton Bay, Australia, as a case study. ***Chapter 3*** addresses objective two and focuses on the costs of conservation planning for coastal wetlands as their distribution changes under sea level rise. Here, I also consider the potential for payments for carbon sequestration and fisheries maintenance to reduce these costs. The next chapter (objective 3, ***chapter 4***) further explores the issue of preserving coastal wetlands (and the ecosystem services they provide) when there are multiple uncertainties. Here I develop an approach to preserve these assets and services that are both cost-effective and robust to the range of uncertainties. Whilst previous studies have dealt with coastal planning under sea level rise (e.g. Abel et al. (2011), Erwin (2008) and Runting et al. (2013)) they have not identified a method for preserving wetlands that is both cost-effective and robust to different climate change projections.

Where there are trade-offs among ecosystem services, management strategies are needed to balance the relative provision of each service (Rodríguez *et al* 2006), whilst also accounting for different global change scenarios (Bryan 2013). The combined impact of climate change and global economic drivers has rarely been considered for ecosystem services in any system (see Connor *et al.*, (2015) and Bryan *et al.*, (2016) for exceptions), and never for livestock production and greenhouse gas regulation in tropical savannas. The second theme (objective 4, ***chapter 5***) addresses this by evaluating strategies for managing these antagonistic services in northern Australia's tropical savannas under global change scenarios. Here I use an integrated modelling approach to assess the impact of climate change, fire, and global economic drivers on the profitability and effectiveness of management strategies.

When assessing, mapping or managing ecosystem services, the potential ramifications of global change are often overlooked (Ziervogel and Ericksen 2010), despite climate and economic drivers having a potentially large effect on the management outcome (Bryan 2013). Allocating land uses or management to achieve multiple objectives is a challenging task, particularly when ecosystem services are competing (Kiker *et al* 2005) or are subject to the impacts and uncertainties of global drivers (Polasky *et al* 2011). In this thesis, I address these gaps by developing and evaluating management approaches that deal with multiple ecosystem services and the impacts of global drivers of change.

2 Incorporating climate change into ecosystem service assessments and decisions: A review

This chapter is reproduced from the following paper, with some alterations to formatting and structure:

Runting, RK, Bryan, BA, Dee, LE, Maseyk, FJF, Mandle, L, Hamel, P, Wilson, KA, Yetka, K, Possingham, HP, & Rhodes, JR. 2017. Incorporating climate change into ecosystem service assessments and decisions: A review. *Global Change Biology*. 23(1): 28–41. dx.doi.org/10.1111/gcb.13457

2.1 ABSTRACT

Climate change is having a significant impact on ecosystem services, and is likely to become increasingly important as this phenomenon intensifies. Future impacts can be difficult to assess as they often involve long time scales, dynamic systems with high uncertainties, and are typically confounded by other drivers of change. Despite a growing literature on climate change impacts on ecosystem services, no quantitative syntheses exist. Hence, we lack an overarching understanding of the impacts of climate change, how they are being assessed, and the extent to which other drivers, uncertainties, and decision making are incorporated. To address this, we systematically reviewed the peer-reviewed literature that assesses climate change impacts on ecosystem services at sub-global scales. We found that the impact of climate change on most types of services was predominantly negative (59% negative, 24% mixed, 4% neutral, 13% positive), but varied across services, drivers, and assessment methods. Although uncertainty was usually incorporated, there were substantial gaps in the sources of uncertainty included, along with the methods used to incorporate them. We found that relatively few studies integrated decision making, and even fewer studies aimed to identify solutions that were robust to uncertainty. For management or policy to ensure the delivery of ecosystem services, an integrated approach that incorporates multiple drivers of change and accounts for multiple sources of uncertainty is needed. This is undoubtedly a challenging task, but ignoring these complexities can result in misleading assessments of the impacts of climate change, sub-optimal management outcomes, and the inefficient allocation of resources for climate adaptation.

2.2 INTRODUCTION

Climate change is having a significant impact on ecosystem services, and these impacts are likely to increase as this phenomenon intensifies (Mooney *et al* 2009). However, the impacts of climate change on ecosystem services can be difficult to assess as impacts often change over long time scales with high uncertainties (IPCC 2014). Regional variation in climate drivers and pressures can create further challenges when assessing and managing their impacts (van Vuuren *et al* 2007). Despite these challenges, integrating climate change and other drivers into assessments of ecosystem service provision is vital, because efforts to ensure supply of ecosystem services which ignore these impacts could lead to perverse outcomes. For instance, designing a coastal reserve system that ignored the impacts of sea level rise could lead to a decline in coastal wetlands and the ecosystem services they provide in the long run (Runting *et al.* 2017b). To add to this challenge, future drivers of change of ecosystem services are not limited to the biophysical aspects of climate change but also include socio-economic changes occurring in parallel, such as increases in population, food demand, and technology, as well as changes in policy and institutions (Millennium Ecosystem Assessment 2005) (Figure 2.1).

Assessing the impact of the different attributes of climate change on ecosystem services (e.g., changes in precipitation, temperature, CO_2, and sea level rise) individually is informative but does not necessarily capture all the information needed for a comprehensive assessment. It is important to consider the impact of multiple attributes of climate change simultaneously within the socio-economic context that together drive the relative supply of and demand for ecosystem services. To illustrate, climate change may decrease agricultural production through declines in rainfall, increases in evaporative demand, and shorter growing seasons, despite the positive effects of CO_2 fertilization on productivity (Rosenzweig *et al* 2014). However, increases in global population and demand for agricultural commodities may facilitate agricultural expansion or intensification (Foley et. al 2005), which could result in an overall increase in food provision. Because of these complex interactions, assessing the relative and cumulative impact of these drivers is essential for a thorough understanding of ecosystem service change.

It is also important to incorporate the impacts of key local drivers of change, alongside global drivers such as climate change, as this could impact both the outcome of the assessment and how the service is managed (Figure 2.1). For example, efforts to secure freshwater supply in South

Africa's fynbos ecosystem in a drying climate may be thwarted by invasive alien woody plant species, as these species have higher rates of evapotranspiration than the native fynbos plants (Pejchar and Mooney 2009). After considering these key impacts, policy to secure freshwater supply in the region is now focused on the removal of these invasive species (Buch and Dixon 2009). Furthermore, both local and external drivers may alter the relationships between services, particularly where each service reacts differently to the same driver (Bennett *et al* 2009). Identifying and incorporating these key drivers of change in ecosystem services is essential for designing context appropriate management strategies.

However, even if all major drivers are incorporated into ecosystem service assessments, there may still be considerable uncertainty associated with the results. First, there is substantial uncertainty involved in projections of climate change and its potential impacts (IPCC 2014). This is further confounded by the uncertainty in the magnitude of other drivers of change (such as varying demand and commodity prices), which can also alter the demand for and supply of ecosystem services (Bryan, 2013) (Figure 2.1). Other potential uncertainties, such as those associated with the measurement or modelling of ecosystem services, may also be important to consider (Hamel & Bryant, In review). Quantifying this uncertainty is not only important for determining the range of impacts on ecosystem services but is especially important to include in designing robust policy and management strategies.

Despite a growing number of studies assessing the impacts of climate change on ecosystem services, there are no quantitative syntheses of this information. Consequently we lack a broad understanding of these impacts, how they are being assessed, and the extent to which other drivers, uncertainties, and decision making are included. To address these gaps, we systematically reviewed the peer-reviewed literature that assesses climate change impacts on ecosystem services at sub-global scales. This allowed us to quantify the impacts of climate change and other drivers on ecosystem services, and determine how these impacts were measured or modelled. In doing so, we determine how uncertainty was incorporated in these assessments, and the extent to which decision making (actions, policies, or other interventions) was considered. We also identify gaps in the literature relating to the contexts of the assessments, and recommend key directions for future research.

2.3 METHODS

To address these aims, we designed a conceptual framework to structure our literature review (Figure 2.1). Climate change, along with a range of other drivers and decisions, can impact ecosystem service provision. Non-climate drivers of change (e.g., land use change) can vary in scale from local drivers (which originate within or proximate to the study site) to external drivers (which operate at a scale larger than the study site). Whether a particular driver is local or external can depend on the scale and context of the study. For instance, commodity prices for food and raw materials are set globally for crops like wheat, corn, or cotton, but set locally for some non-timber forest products such as some medicinal plants, forage, and resin (Shackleton *et al* 2007). Additionally, a driver that is external at the patch scale (e.g., fertilizer run-off) may be within the study area at regional or national scales. These drivers of change are often interrelated as external drivers can influence local ones, such as global commodity prices influencing local land use change. Decisions made at the local scale can directly improve ecosystem service provision or influence local drivers, but they generally do not have a significant impact on the magnitude of external drivers. Decision making can also occur well outside the location and scale of the study area (e.g., the national and global level decision making inherent in the IPCC emissions scenarios (IPCC 2013)), but here we focus on the decisions that can be made by local and regional actors to *adapt* to the impacts.

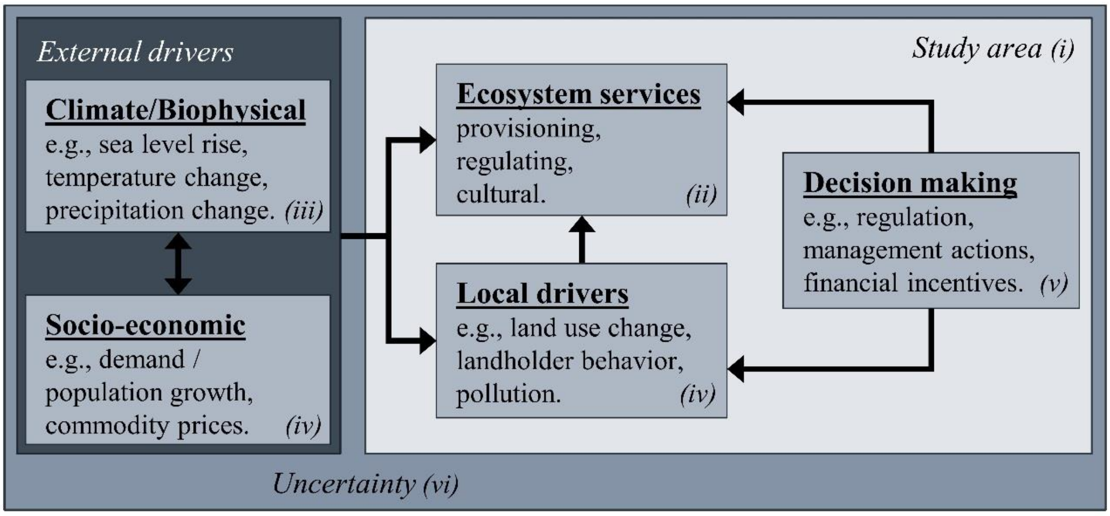


Figure 2.1 | A simplified conceptual framework illustrating how drivers of change impact ecosystem services. Ecosystem service provision is affected by climate change and other drivers (from global to local), along with decisions relating to their management. These decisions address the ecosystem service directly (e.g., through site-based management) or indirectly (by influencing local drivers). Uncertainty is inherent in all components of the framework and their interactions. This framework was used to structure our systematic literature review, with roman numerals indicating how each component relates to specific sections of the data extraction process (Table 1).

We compiled a set of peer-reviewed journal articles on ecosystem services and climate change (Figure 2.2). A list of research articles published between 1990 and 2014 was generated using selective key-words under "TOPIC" in the database of ISI Web of Science Core Collection. Articles published in 2014 were only included if they appeared in the database before November 2014. We applied the search: ("ecosystem service*" OR "ecosystem good*") AND (climat* NEAR chang*). The key word search was constrained to general terms in order to produce a representative sample of the literature (rather than a comprehensive list). Using "ecosystem service" OR "ecosystem good" omitted studies that assessed an ecosystem service, but did not identify it as such (e.g., food production, biofuels, health benefits). Studies that did not use the term "ecosystem service" would be unlikely to follow an ecosystem service framework, so comparing them to our conceptual framework (Figure 2.1) would have potentially exaggerated research gaps (such as incorporating drivers other than climate change and decision making). Additionally, including more specific terms such as "crops" or "fisheries" would bias the results towards these services and return an impractical number of papers, so specific key words such as these were excluded. We applied a similar approach to climate change phenomena (e.g., we did not include additional terms like "sea level rise" or "global warming") for the same reasons. These general search terms returned 1,567 papers (Figure 2.2).

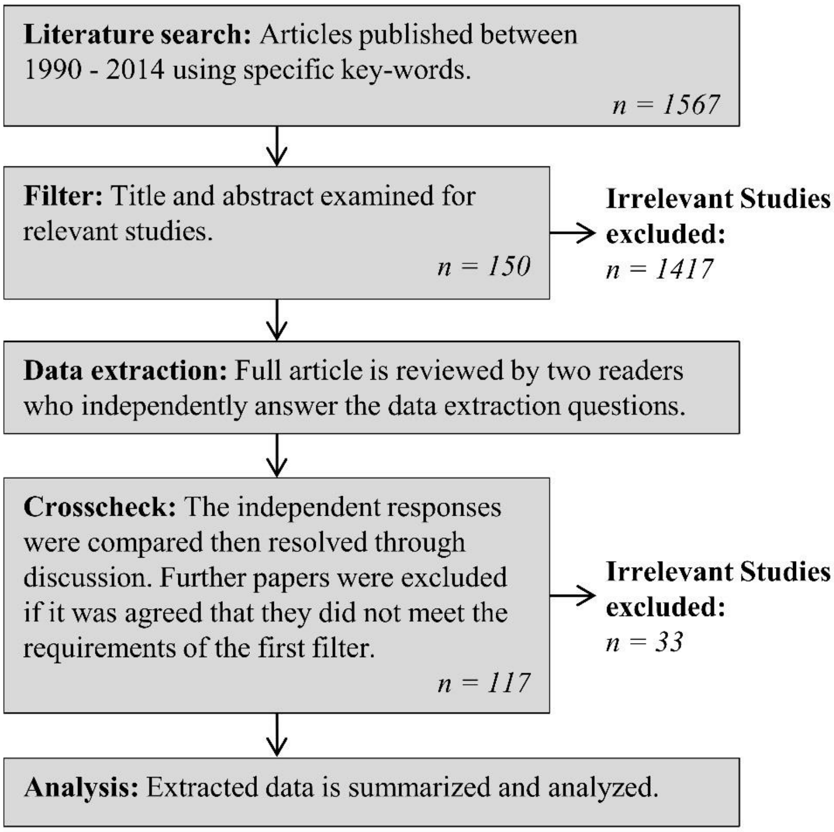


Figure 2.2 | Flow chart demonstrating the methods used in the systematic quantitative review. Articles published in 2014 only include those that appeared on Web of Science before November 2014.

We read the abstracts of these 1,567 papers to determine if they met the requirement for inclusion in this study (the filter, Figure 2.2). These criteria had three components. First, our criteria required papers to be an assessment of provisioning, regulating or cultural ecosystem services (in accordance with the TEEB (2010) framework). This excluded reviews or conceptual papers and articles that focused on biodiversity or supporting/habitat services, as these are better defined as ecosystem functions (de Groot *et al* 2002, 2010, Wallace 2007), and the impact of climate change on species and biodiversity has been reviewed elsewhere (Tylianakis *et al* 2008, Bellard *et al* 2012, Mantyka-Pringle *et al* 2012, Chapman *et al* 2014, Pacifici *et al* 2015). Second, we excluded studies that did not incorporate climate change impacts (e.g., studies focusing on carbon sequestration in the absence of climate change impacts but refer to its importance for mitigating climate change). Last, global-scale assessments of climate change impacts on ecosystem service provision were excluded because regional variations in climate drivers create unique challenges at sub-global scales (such downscaling global climate scenarios (van Vuuren *et al* 2007)), and adaptation to the impacts of climate change usually occurs at sub-global scales (Ford *et al* 2011).

The 150 papers that passed these criteria were read in detail to extract data using specific questions (Figure 2.2). These questions had fixed answer categories, along with an open-ended comment box to clarify responses and ensure consistency in data extraction (see Table 2.1 for a summary, and Table B.4 for details). In order to minimize errors and biases, each paper was read by two readers (co-authors of this review paper), who independently answered the data extraction questions. The two responses for each paper were then compared, and any discrepancies were noted qualitatively (the nature of the discrepancy) and quantitatively (0 for complete disagreement, and 0.5 for partial agreement [1 was given if there was no discrepancy]). These quantitative scores revealed a mean agreement of 22.3 (86%) answers (σ = 2.6 [10%]) of a maximum possible 26. Recording the differences qualitatively allowed any discrepancies to be resolved through a discussion between the readers, with a third opinion sought from an additional reader if needed. These final (i.e., resolved) responses were used for the subsequent analyses and form the basis of the results reported here. This process revealed that of the 150 studies that were not initially excluded (from reading the abstract), 33 studies did not fit the criteria described above, so they were excluded from further analysis leaving a total of 117 studies.

Table 2.1 | The structured questions used to extract data from the journal articles. The roman numerals indicate which component of the conceptual framework (Figure 2.1) the section relates to. Each question helps to address one of the aims: (a) identify gaps in the literature relating to the context of the assessments, (b) quantify the impacts of climate change and other drivers on ecosystem services, (c) determine how these impacts were measured or modelled, (d) determine how uncertainty was incorporated in these assessments, and (e) determine the extent to which decision making (actions, policies, or other interventions) was considered. The categories used to answer these questions are given in Table B**.4**.

Category	***No.***	***Aim***	***Question***
Filter	1	-	Is the paper an assessment of ecosystem services?
	2	-	Does the paper incorporate the impacts of climate change?
(i) Study area	3	(a)	Spatial scale of assessment
	4	(a)	Location of assessment
	5	(a)	Type of ecosystem(s)?
(ii) Ecosystem services	6	(a)	Which ecosystem service(s) were considered? State the indicator used.
	7	(a)	What aspect of each ecosystem service is considered?
	8	(c)	If monetary value was considered, what valuation method was used?
(iii) Drivers: Climate	9	(b)	What aspect(s) of climate change are considered?
	10	(b)	Were these attributes of climate change assessed cumulatively, in isolation from each other, or both?
	11	(b)	What was the impact of climate change on the ecosystem services studied?
	12	(b)	Are interactions between services considered (i.e., trade-offs)?
	13	(c)	What method was used to incorporate climate change and ecosystem services?
	14	(c)	Was the method static, or did it consider changes over time?
(iv) Drivers: other	15	(b)	Are other drivers considered?
	16	(b)	If other (non-climate) drivers were incorporated, list the drivers.
	17	(b)	What was the impact of the non-climate driver on the ecosystem service studied?
	18	(c)	How was the impact of the driver(s) assessed?
	19	(b)	How did each driver interact with climate change?
(v) Decision making	20	(e)	Is decision making considered (i.e., actions, policies, or other interventions)?
	21	(e)	How many objectives are considered (list all)?
	22	(e)	What method is used to model or assess the action, policy, or interventions?
	23	(e)	What category do these actions, policies or other interventions fall into?
(vi) Uncertainty	24	(d)	Was uncertainty considered?
	25	(d)	What was the source of the uncertainty, and what methods were used to incorporate it in the assessment?
	26	(d, e)	If decision-making is considered, are the decisions robust to uncertainty?

A range of questions were used to quantify the impacts of climate change and other drivers on ecosystem services (b) and the methods used to assess them (c). We collected information on which aspects of climate change (Q9) and which non-climate drivers of change (if any) (Q15, Q16) were considered. Options for which climate change attributes were included were adapted from IPCC (2014). The response categories for which non-climate drivers were assessed (Q15) were not pre-defined, so any driver could be included. To quantify the (directional) impact of drivers on ecosystem services, the impact of climate change (Q11) and non-climate drivers (Q17) was recorded as positive, negative, neutral, or mixed. We did not specify quantitative measures of the magnitude of change, as this would be problematic to compare across different services using different methods (particularly qualitative methods), baselines, and indicators. We also recorded if any interactions between services were assessed (Q12), and if the attributes of climate change were assessed cumulatively, in isolation from each other, or using both of these approaches (Q10). If the study considered both the cumulative and individual impacts of climate change and other drivers (Q18), we allowed an option to record the interaction between climate and non-climate drivers, specifically, whether their impacts are synergistic, antagonistic, additive or unclear (Q19) (based on definitions in Brown et al. (2013)). The methods used to assess the impact of climate change could be identified as empirical (i.e. a laboratory or field based study), a statistical or process-based model (with or without the use of local field based data), expert elicitation, or other methods (Q13). These methods were further classified as static (assessing only one future or past time point in addition to the baseline) or dynamic (assessing more than one future or past time points), and the interval between time points was also recorded (Q14). If monetary valuation was undertaken, the valuation method was specified (e.g., market value, avoidance cost, contingent valuation) (Q8), based on definitions from Christie et al. (2012).

To determine how uncertainty was incorporated in these assessments (d), we first recorded whether uncertainty was mentioned, explicitly incorporated in the assessment, or ignored (Q24). We then identified the methods used to incorporate uncertainty (i.e., scenario analysis, sensitivity analysis, multiple models, probabilistic approaches, or other methods), which were adapted from Polasky et al. (2011), Yousefpour et al. (2011), and Refsgaard et al. (2007) (Q25). For each method, we also identified which source(s) of uncertainty it addressed (e.g. the magnitude of climate change, or how ecosystem services are supplied) (Q25). This information was also used to identify gaps in the sources of uncertainty that were accounted for.

To get an understanding of the extent to which decision making was incorporated (e), we recorded if solutions were explicitly measured or modelled, just mentioned, or ignored (Q20). Where decision making was included, we identified the methods used (e.g., cost/benefit analysis, adaptive management) (Q21, Q22), the solutions proposed (Q23), and if these solutions were robust to the uncertainties included (Q26). Here we focused on decision making that occurred at a similar scale to the study area (Figure 2.1). Of course, decision making can also occur at much larger scales (e.g., global policies), but these decisions were usually bundled with other external drivers (and were treated as such in this review). A full list of questions and response categories are given in Table B.4.

We then conducted a meta-analysis to determine if there was statistically significant variation in climate change impacts on ecosystem services across service categories, climate change attributes, methods used, biomes and spatial scales. Given the categorical nature of our data, we used cumulative logit models with the ordinal categorical impacts of climate change on ecosystem services as the response variable, and the spatial scale of the study, type of ecosystem (i.e., terrestrial, freshwater or marine), climate change attributes (e.g., temperature increase, CO_2 fertilization or sea level rise), ecosystem service categories, and methods used (i.e. empirical, expert elicitation, process-based or statistical modelling) as predictor variables. Broad ecosystem service categories (i.e., provisioning, regulating, and cultural) were used instead of the 15 individual TEEB ecosystem service types to ensure a sufficiently large number of records in each category (see Appendix B for details).

2.4 RESULTS

2.4.1 Contextual information

Our review revealed clear patterns in the contextual information of the reviewed papers and the characteristics of the ecosystem services studied (Figure 2.1). All studies that passed the first filter were published since 2003, with 78% of these published since 2011 (Figure 2.1c). This trend suggests a growing interest in climate change impacts on ecosystem services. We found that the studies considered a diversity of spatial scales (Figure 2.1d), but there was a clear dominance of terrestrial ecosystems (91 studies) over freshwater (40 studies) and marine (17 studies) ecosystems (Figure 2.1e). Although a large number of countries were covered by at least one study (131 countries), there was a focus on the USA and Europe, with 30 studies (26%) in the USA and 49 studies (42%) in Europe (Figure 2.3g).

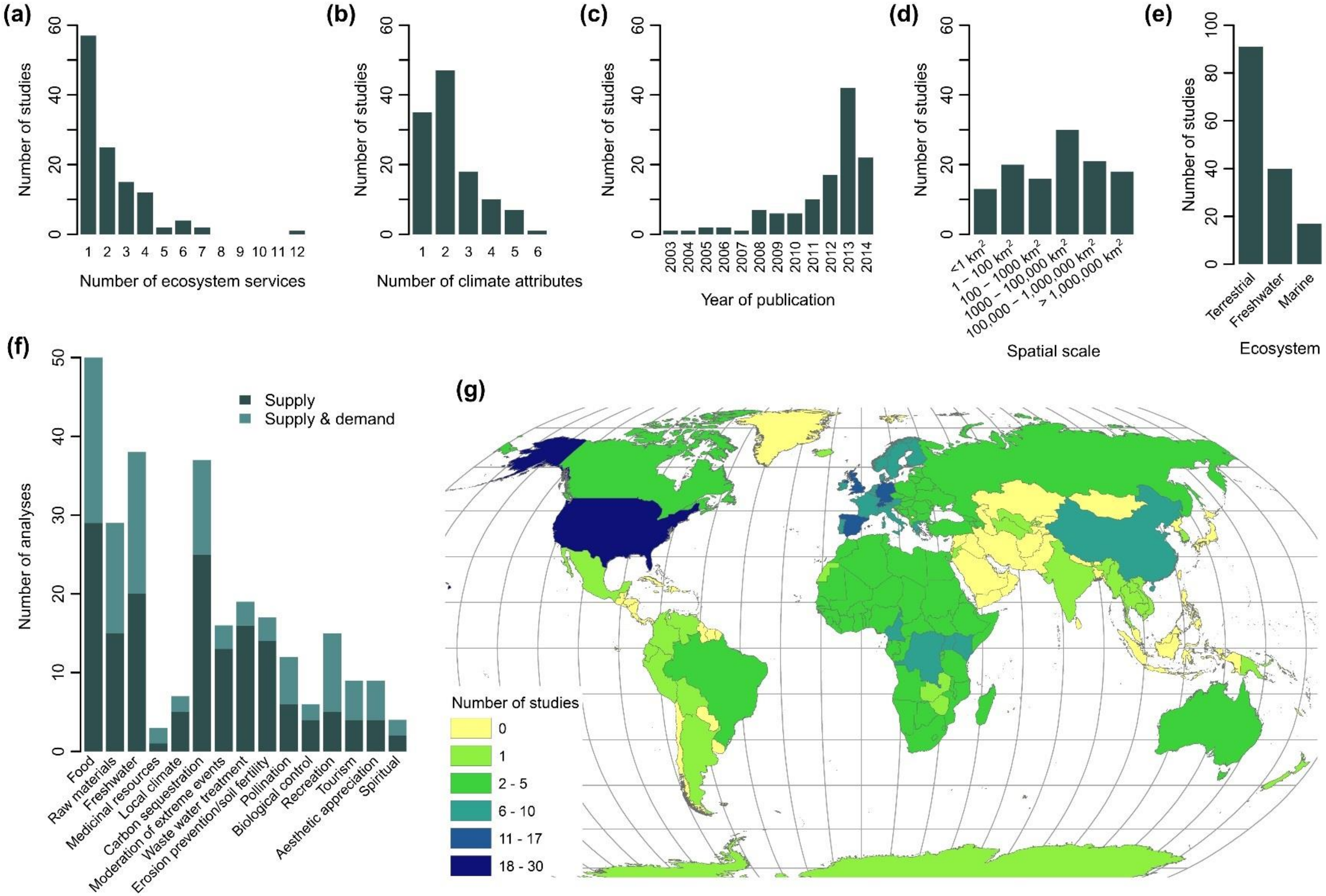


Figure 2.3 | Key attributes of the 117 ecosystem service assessments: (a) the number of ecosystem services included in each paper with a unique indicator (i.e. if the same indicator was used for multiple services, it was only counted once), (b) the number of attributes of climate change included in each paper, (c) the frequency of each year of publication (2014 only includes papers that appeared on Web of Science before November 2014), (d) the frequency of each spatial scale, (e) the frequency of each type of ecosystem, (f) the frequency of each ecosystem service and whether supply and/or demand was considered, and (g) the number of studies by nation. In panel (f), the ecosystem services are ordered in accordance with the TEEB (2010) framework, so that they are grouped by provisioning (i.e., food, raw materials, freshwater, and medicinal resources), regulating (from local climate to biological control) and cultural (i.e., recreation, tourism, aesthetic appreciation, and spiritual benefits) services. Panels (e), (f), and (g) sum to more than the total number of papers, as each paper could span more than one nation, and could cover more than one ecosystem and service.

There were also biases in the characteristics of the ecosystem services studied. Provisioning services (particularly food, raw materials and freshwater) and carbon sequestration dominated the literature, with cultural services receiving the least attention (Figure 2.1f). Whilst the focus of most studies was on the supply side of ecosystem service provision, the link to beneficiaries (demand) was also included in almost 40% of cases (Figure 2.1f). Finally, nearly half of the studies focused on a single ecosystem service (48%, Figure 2.1a), which provided the opportunity for in-depth analysis but meant that interactions between services (e.g., trade-offs) in the context of climate change were rarely considered (only 17% of studies).

2.4.2 The impact of climate change and other drivers

We found that a diversity of climate change attributes were included, with most studies considering more than one attribute (70%, Figure 2.1b). The most common attributes were temperature (81% of papers), often coupled with precipitation change (an increase, decrease or increasing variability; 63%), but other combinations of climate change attributes were also explored. Of those studies that considered two or more climate change attributes, 77% assessed these impacts cumulatively (all together), 9.8% assessed the attributes individually, and 13% assessed the impacts both individually and cumulatively. We found that the impact of climate change on ecosystem services was predominantly negative (59% of analyses were negative, 24% mixed, 13% positive, 4% neutral); however, this pattern was not consistent across services or attributes of climate change (Figure 2.2a). The category of ecosystem service (i.e., provisioning, regulating or cultural) influenced the results, with regulating and cultural services being impacted more negatively by climate change than provisioning services (regression coefficients are -0.38 [regulating] and -1.9 [cultural], relative to provisioning services, Table B.2). However, this effect was only significant for cultural services (p = 0.00155, Table B.2).

Based on the four impact categories, carbon sequestration had the most variable response to climate change (41% of analyses were mixed, 35% negative, 20.5% positive, 3.5% neutral), but other services had a more negative response (e.g., 92% of analyses of the impact on biological control were negative, with only 8% mixed) (Figure 2.4a). Similarly, CO_2 fertilization had the most positive impact on services (i.e., 36% of analyses were positive, 36% negative, 14% mixed, and 14% neutral), whereas other climate change attributes produced a stronger negative response (e.g., 96% of studies on the impact of sea level rise were negative) (Figure 2.2a).

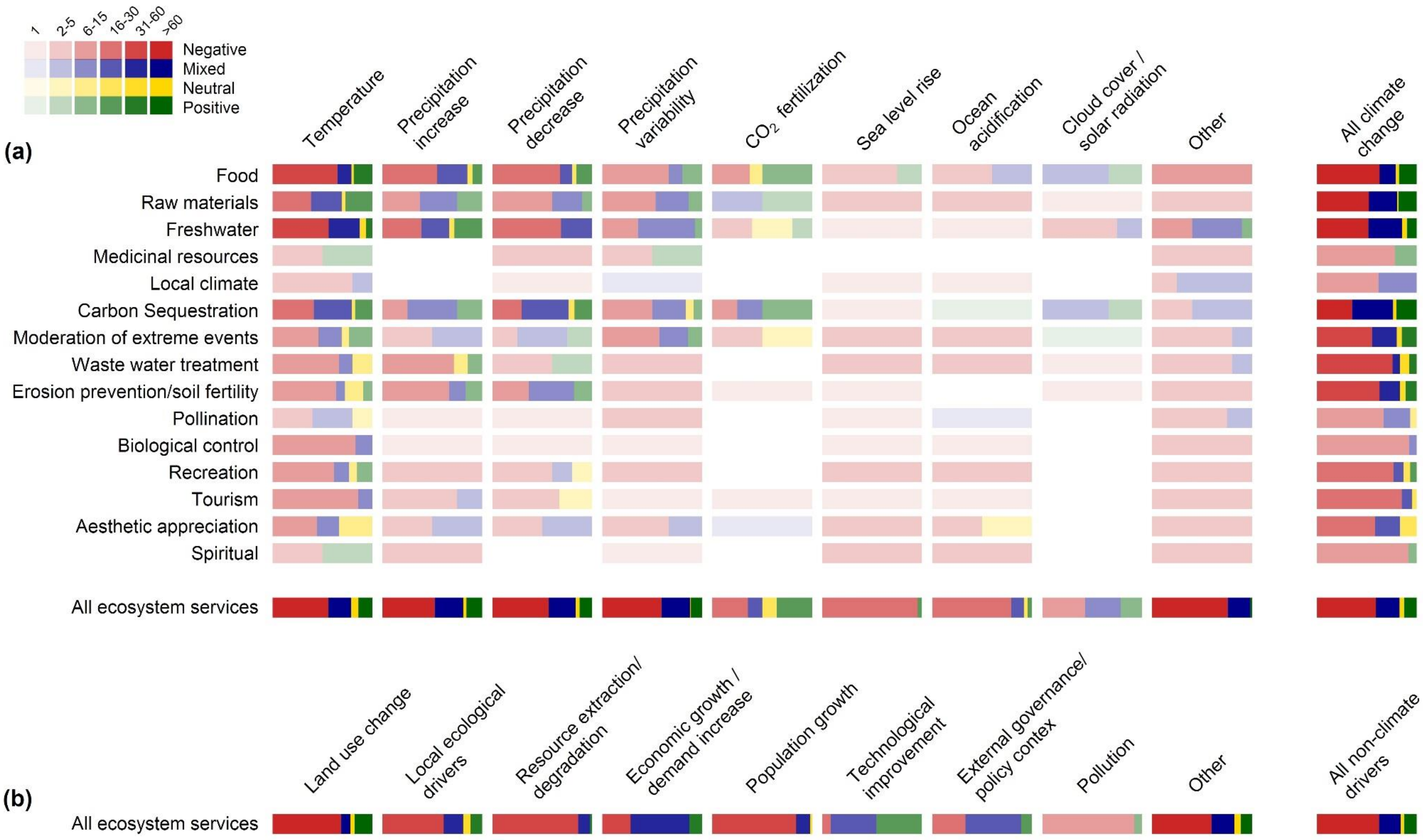


Figure 2.4 | The impact of climate change and other drivers on ecosystem services. Panel (a) shows the impact of individual attributes of climate change on individual ecosystem services. The bottom row of this panel shows the impact of each climate change attribute across all services, and the far right column shows the total climate change impact for each service. The bottom right bar of this panel gives the total impact for all services and attributes of climate change. Panel (b) shows the individual and total impact of other drivers on all ecosystem services. For both panels, the bar indicates the *proportion* of analyses giving a negative, mixed, neutral or positive response for each ecosystem service and driver combination (i.e., this does not take into account effect sizes). The strength of the colour represents the *total* number of analyses for that driver and ecosystem service (i.e. solid colours indicate many analyses, whereas faded colours indicate few analyses, and blank space indicates zero studies). The number of analyses for each level of colour strength is shown in the legend.

We found that more than half of the papers in our review (56%) incorporated drivers other than climate change, and 31% either mentioned in passing or discussed these drivers in depth (without incorporating them). Whilst the impact of all non-climate drivers varied, they had a predominantly negative impact (62% of analyses were negative, 33% neutral, 22% mixed, 13% positive), with the exception of technological improvement, which had a largely positive impact (46% of analyses were positive, 46% mixed, 8% negative) (Figure 2.2b). Land use (or land use management) change was the non-climate driver that was most often included (28% of analyses that included non-climate drivers), with largely negative impacts (69% of analyses were negative, 18% positive, 9% mixed, 4% neutral). Of studies that considered non-climate drivers, 61% assessed the cumulative impact with climate change, 5.8% assessed other drivers and climate change separately, and 33% considered both cumulative and individual impacts.

2.4.3 Methods used to assess impacts

A variety of methods were employed to determine the impact of climate change on ecosystem services. Process-based modelling (e.g., hydrological models, deterministic ecosystem service models) was the most frequently used method (51% of analyses), and most of these process-based analyses were parameterized with some local field data (85%). However, empirical field-based or laboratory studies were less frequently used (10% of analyses) (Figure 2.5a and c). Almost half of studies (48%) conducted a dynamic assessment (i.e., considered more than one future time point), and of these studies, the time interval between future time points varied between 0.2 days (for some hydrological models) and 100 years. Similarly, of the 19 papers (16%) that included monetary valuation of ecosystem services, a variety of valuation methods were used (including market methods, production approaches and avoidance cost), but benefit transfer was relied upon the most often (in 29% of analyses) (Figure 2.3e).

We also found that the method used may impact the outcome of the assessment. Specifically, relying on expert opinion to determine the impact of climate change (in 21% of analyses, Figure 2.3c) gave primarily negative results (94% of these analyses were negative), which was in contrast to other (empirical, quantitative modelling) methods that showed more variation in the impacts of climate change (where 47% of analyses were negative) (Figure 2.3d). The more frequently negative impacts of expert elicitation were reflected in a relatively large regression coefficient (-5.2, relative to process-based models) which was found to be statistically significant ($p = 0.003$) (Table B.2).

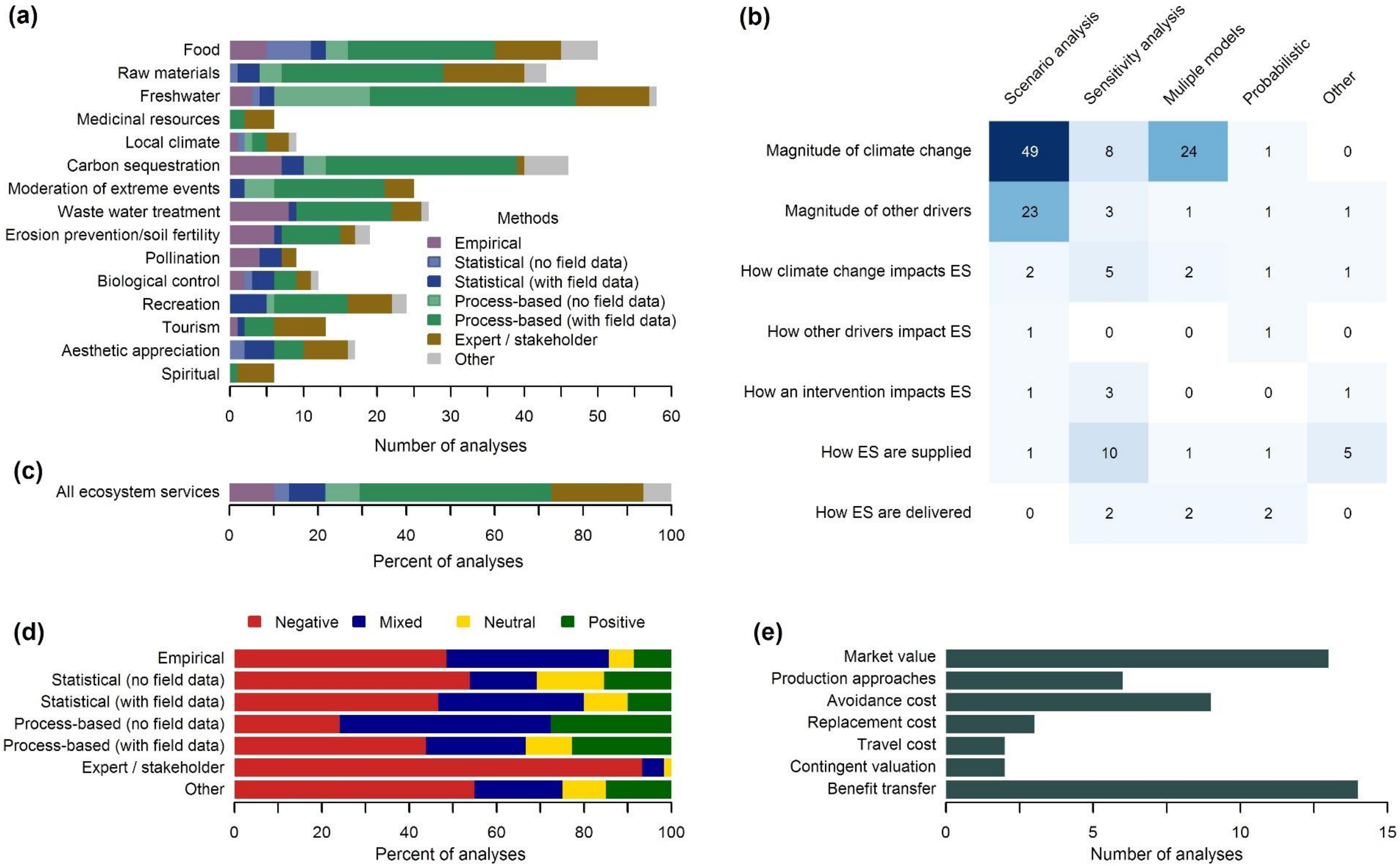


Figure 2.5 | Methods used to assess the impact of climate change on ecosystem services. Panel (a) shows the frequency each method was used to assess the impact of climate change on each ecosystem service. Panel (b) shows the frequency of methods used to incorporate uncertainty into the ecosystem service (ES) assessments by the frequency of the type of uncertainty that was addressed. Panel (c) shows the percent of analyses that used each method to assess the impact of climate change across all services, and panel (d) shows the proportion of analyses that had a negative, mixed, neutral or positive impact of climate change on ecosystem services by each of these methods. Panel (e) illustrates the frequency of different methods used when monetary valuation was included in the assessment. Each paper potentially assessed more than one ecosystem service and potentially used more than one method, so the number of analyses can sum to more than the total number of papers, and differ from those in Figure 2.3.

2.4.4 Uncertainty

We found that there were gaps in the sources of uncertainties considered in the analyses, along with the methods used to incorporate them (Figure 2.3b and Table B.4 for definitions of methods). At least one source of uncertainty was explicitly incorporated in 71% of studies and was mentioned or discussed by another 17%. Uncertainty in the magnitude of climate change was the main uncertainty addressed (Figure 2.3b), and the dominant method for addressing this, as for most sources of uncertainty, was scenario analysis, followed by using multiple models (Figure 2.3b). This was usually achieved through the use of multiple IPCC emissions scenarios to inform multiple global circulation models, which formed the basis of the analyses (e.g., Müller *et al.* (2014) and Matthews *et al.* (2013)).

2.4.5 Decision making

Whilst various types of decision making were often mentioned (83% of papers), decision making was less frequently included in analyses (29% of papers). A number of different solutions were proposed, and these were assessed using a variety of methods across the studies that incorporated decision making (Figure 2.4). Only five studies included decision making outcomes (i.e. policies or management strategies) that assessed robustness to at least one type of uncertainty, and three of these focused on a single ecosystem service (i.e., a single objective). These decision making strategies included: planting a climate-resilient species mix for silviculture (Seidl *et al* 2011, Steenberg *et al* 2011), protecting wetlands (Grossmann and Dietrich 2012), setting maximum stocking rates for livestock (Schaldach *et al* 2013), and managing a buffer stock of timber (Raulier *et al* 2014).

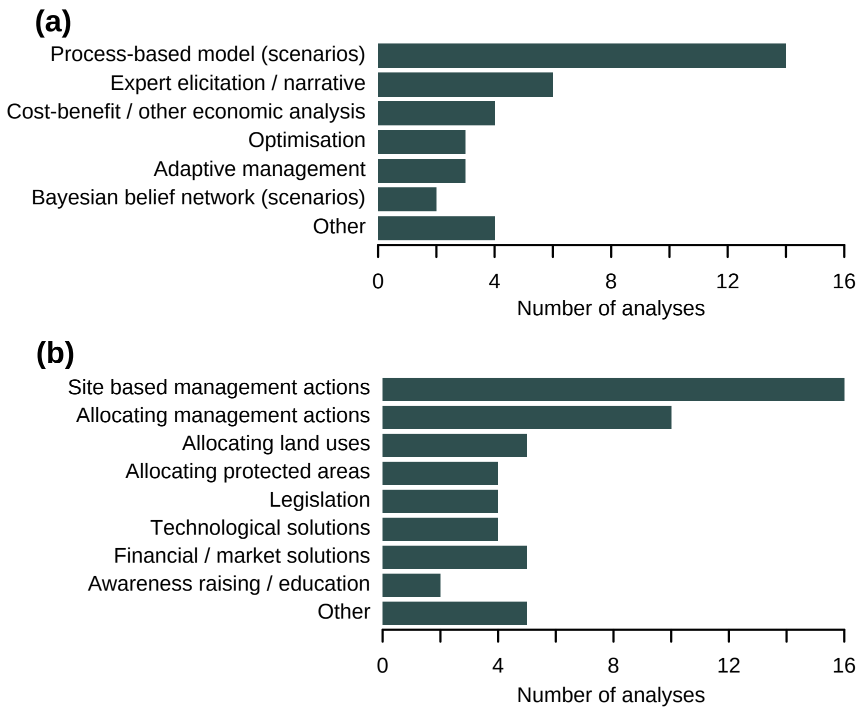


Figure 2.6 | Decision making for ecosystem services under climate change. Panel (a) shows the frequency of each method used to model decisions. Panel (b) shows the frequency of different classes of solutions (actions) that were assessed. The number of analyses sum to more than the total number of studies that incorporated decision making (n=34) as more than one method could be employed and solution could span multiple categories.

2.5 DISCUSSION

Our review revealed that the majority of studies found a negative impact of climate change on ecosystem services, yet the effects varied across services, climate change attributes, and assessment methods, and in some cases were positive. There is strong evidence that climate change is having a negative (but variable) impact on biodiversity (Bellard *et al* 2012, Pacifici *et al* 2015) so it is unsurprising that the services that flow from species and ecosystems are similarly impacted. Our finding of predominantly negative impacts is also in line with qualitative syntheses of climate change impacts on ecosystem services (Mooney *et al* 2009, Scholes 2016), which highlight the need for climate change adaptation strategies to ameliorate these impacts. The complex temporal and spatial patterns across multiple climate change attributes (Dobrowski *et al* 2013, IPCC 2013) suggests that the variability seen in our results is an accurate representation of climate change impacts.

We found that carbon sequestration had the most variable response to climate change (Figure 2.2a), and the context of each study appeared to affect the direction of climate change impacts. For instance, a freshwater mesocosm experiment showed that temperature increases reduced carbon sequestration by 13 percent by shifting the metabolic balance of the ecosystem (Yvon-Durocher *et al* 2010). In contrast, climate change had a positive impact on carbon sequestration in the Swiss Alps, as increasing temperatures enabled forest expansion into higher altitudes (Grêt-Regamey *et al* 2013). This variability is supported by other meta-analyses on the response of carbon sequestration to temperature increases or elevated atmospheric carbon dioxide. Luo et al (2006) found that elevated atmospheric carbon dioxide increased total carbon accumulation in terrestrial ecosystems, but these results were highly variable across studies and carbon pools. Similarly, the analyses by Lu et al (2013) revealed that carbon sequestration response to temperature increase varied by ecosystem type (i.e., forest, grassland, shrubland, tundra, and wetlands).

Although the impacts on other ecosystem services were more consistently negative (Figure 2.2a), contextual factors (e.g., climatic zone and type of ecosystem) still appeared to influence the results. For example, the impact of drought on the persistence and production of perennial grasses used for forage varied between temperate and Mediterranean climate types in France (Poirier *et al* 2012). This variability in food provision is supported by a global meta-analysis, which showed that whilst increases in temperature generally decreased crop yield, there was significant yield variability across crop types and temperate/tropical regions (Challinor *et al* 2014). Similar variability in food provision in response to temperature increases can be seen in the marine environment, with maximum fisheries catch potential increasing in offshore regions but decreasing in the coastal zone (Cheung *et al* 2010). The lack of generalities and statistical significance across services and climate change attributes indicates the importance of local and regional assessments of ecosystem services, by service type, rather than relying on averages, aggregates, or trends seen at broader spatial scales.

Our systematic review also revealed gaps in the context and characteristics of the ecosystem services studies. The literature was dominated by studies from the USA and Europe (Figure 2.1g), indicating a need for further studies beyond these regions. This is particularly important as the impacts of climate change on ecosystem services are likely to disproportionately affect developing countries, who also have a lower capacity to adapt to these impacts (Srinivasan 2011). Another major gap was the study of cultural services (Figure 2.1f), which is unsurprising given they are often omitted from assessments of ecosystem services due to the difficulties in characterizing these services (Chan *et al* 2012). Similarly, most studies focused on the biophysical supply (or 'supply side') of ecosystem services, which is consistent with the findings of other ecosystem services reviews (e.g., Martinez-Harms et al. (2015)). However, this focus on supply misses an opportunity to provide a complete assessment of ecosystem services by demonstrating benefit to people ('demand side') (Tallis *et al* 2012). This link is particularly important, as there is often a spatial mismatch between the supply and demand of ecosystem services (Bagstad *et al* 2012). It may be the case that only part of the area supplying the service may be necessary to meet demand, or, conversely, a greater area of supply may be required (Bagstad *et al* 2012). In addition, clearly demonstrating the benefits to humans is essential for meaningful integration with planning and policy decisions (Daily *et al* 2009).

Assessing both the relative and cumulative impacts of multiple attributes of climate change was often overlooked. We found that most studies considered the cumulative impacts of climate change, which is promising as this has previously been highlighted as an important area for future research (Tylianakis *et al* 2008, Staudt *et al* 2013). On the other hand, studies that isolate the impacts of individual attributes of climate change are still vital for determining the relative impact of each attribute. We found that the relatively few studies that considered both the cumulative and individual impacts of climate change allowed for further insights that would not have been possible with other study designs. This was illustrated by Lindeskog et al. (2013), who revealed that CO_2 fertilization would only partially offset the negative impacts of other climate change attributes (including temperature increase, precipitation change, and solar radiation) on carbon sequestration. Although these types of studies are often time and resource intensive, they are vital for determining the relative importance of each driver. Knowing which drivers are the most important may be valuable for future assessments where the inclusion of all climate change attributes (and other key drivers) is not possible due to resource constraints.

Integrating other global or local drivers with climate change is critical for understanding the complexities of the impacts on ecosystem services (Carpenter *et al* 2009, Bryan 2013). We found that land use change was the driver that was most often included, which is likely due to the well-established importance of this driver, the existence of land use change models, and the largely negative impacts of land use change (Foley *et al* 2005). For example, the conversion of forest to agriculture in the Brazilian Amazon not only reduces carbon stocks but could also reduce agricultural output in the long run, as deforestation exacerbates the negative impacts of climate change through regional land-climate feedbacks (Oliveira *et al* 2013). Where both cumulative and individual impacts of climate change and other drivers were considered, the interactions between these drivers was often ambiguous (i.e., it was unclear whether their interaction was antagonistic, synergistic or additive), which was largely because the nature of the interactions were not the focus of these studies. Additionally, the dominance of scenario analyses meant that in many cases, it would be problematic to completely isolate all the scenario components without violating the assumption of internal consistency (Amer *et al* 2013). Consequently, the impact on ecosystem services that results from interactions between climate change and other drivers remains an important area for future research.

Whilst some studies employed sophisticated dynamic models or conducted well-designed empirical research to determine the impact of climate change on ecosystem services, other studies utilized simpler methods, which may be prone to errors and biases. For example, when assessing the monetary value of ecosystem services, there was a reliance on benefit transfer (i.e., applying values quantified in other studies, conducted elsewhere) for many value estimates (Figure 2.3e). This method is considered to be unreliable as it is prone to errors resulting from a lack of transferability between locations (although these errors can be reduced if the two sites are very similar) (Plummer 2009, Eigenbrod *et al* 2010a). A variety of other methods for monetary valuation exist (e.g., market price, avoidance cost, damage reduction (Christie *et al* 2012)), which should ideally be utilized instead of a value transfer where possible.

We also found that relying solely on expert elicitation to determine the impact of climate change on ecosystem services may overestimate the negative impacts of climate change. Studies that used expert elicitation gave more frequent negative results than studies employing empirical or quantitative modelling methods, and this effect was statistically significant. This difference could be due to motivational or accessibility bias among experts (Martin *et al* 2012). Specifically, the knowledge that the impacts of climate change are generally negative may exert a disproportionate influence on the experts' judgement, even in cases where the actual impact of climate change may be positive or mixed. A variety of methods exist to minimize bias and verify the accuracy of elicited information (such as eliciting information from a high number and wide variety of experts, eliciting uncertainties alongside best estimates, and providing feedback to experts (Martin *et al* 2012)), but it was not clear if these methods were followed in the studies included in this review. Whilst involving stakeholders is important to facilitate implementation (Reed 2008), when assessing the impact of climate change, expert elicitation should follow formal procedures and ideally be accompanied by other methods where available.

In some assessments, a biological indicator (such as the presence, abundance, biomass, or percentage cover of a particular species or ecosystem) was used as a proxy to measure provision of an ecosystem service, and in some cases the same indicator was used for multiple services. This can be seen in Saulnier-Talbot et al. (2014), where the same set of indicators of lake health were used to measure tourism, freshwater, and food provision. This is particularly concerning, as the way an ecosystem service is measured has been shown to have a substantial bearing on the outcome of the assessment (Eigenbrod *et al* 2010b, Liss *et al* 2013). The importance of this is highlighted by Doherty et al. (2014) who found that biomass (a commonly used indicator) was negatively correlated with four regulating services (flow attenuation, stormwater retention, erosion resistance, and water quality) in some contexts. Consequently, future studies should avoid the use of proxies and measure or model service provision directly where possible.

Incorporating the uncertainty associated with climate change is vital given the current range of climate projections (IPCC 2014), and we found that the magnitude of climate change was the main source of uncertainty addressed. However, other potential uncertainties within the analyses received relatively little attention. For example, uncertainties relating to *how* climate change impacts ecosystem services were rarely incorporated (Figure 2.3b), as this can involve varying which model is used, or the model structure, which requires further time and expertise. Despite these challenges, Jung et al. (2013) included multiple uncertainties in their modeling of freshwater yield in South Korea by using two emissions scenarios, 13 global circulation models, and three different hydrological models. Other methods exist for incorporating multiple sources of uncertainty throughout the modelling process, such as Monte Carlo simulation or uncertainty matrices (Hamel and Bryant In Review; Refsgaard *et al.* 2007), but these were usually overlooked. Therefore, building on climate change scenarios to incorporate multiple sources of uncertainty into ecosystem service assessments remains an important area for future research.

Making decisions in the context of climate change and other drivers is difficult due to the long time frames and uncertainties involved. The main objective of most of the reviewed studies was to investigate the impact of climate change, rather than determine the outcomes of decisions (i.e., policy and management). As assessing the impact of climate change on ecosystem services is a substantial undertaking in itself, it is understandable that these papers also did not address decision making in any great detail. Studies that included decision making usually employed a limited assessment (i.e., only one ecosystem service or attribute of climate change), or had methods and results spanning multiple papers. This is illustrated by Bateman et al. (2013), who explored policy options for multiple ecosystem services in the context of multiple drivers, had a team of 15 authors, and some aspects of the study were published in separate papers (specifically Abson et al. (2014) and Fezzi et al. (2014)). Similarly, Bryan et al. (2015) explored policy options to preserve carbon and biodiversity services under a range of global change drivers using a complex, integrated environmental-economic model, which was developed over several papers (specifically Bryan et al. (2014) and Connor et al. (2015)). Therefore, it is unlikely to be feasible to include multiple drivers and decisions in every analysis, especially for empirical studies that seek to isolate climate impacts.

However, the results of these ecosystem services assessments could be useful for future studies that aim to develop or apply decision making methods under climate change, provided that the data underpinning the results of these ecosystem service assessments are shared by the authors.

A major gap exists in developing and applying decision making methods for ecosystem services under climate change that are robust to uncertainty. In our review, only one study (Raulier *et al* 2014) explicitly incorporated robustness to uncertainty into their decision making objectives. Many methods exist for making good decisions under uncertainty (Polasky *et al* 2011) and have been applied in other fields. For example, Lempert et al. (2012) combined a stochastic cost-benefit analysis with robust optimization to advise the Port of Los Angeles on which facilities (if any) it should upgrade to protect against extreme, but unlikely, sea level rise. Similarly, Bertsimas and Pachamanova (2008) applied robust optimization approaches to multi-period portfolio selection to develop an optimal, time-dynamic financial investment strategy under uncertainty in future returns. Alternatively, Regan et al. (2005) used information-gap theory to determine the optimal management strategies to minimize the extinction risk of the Sumatran rhino (*Dicerorhinus sumatrensis*) under severe uncertainty relating to population models, causes of decline, and the effectiveness of management strategies. Applying methods such as these to managing ecosystem services under global change will bring unique challenges that may require substantial methodological innovation, which should be the focus of further research.

We recommend incorporating complexity into ecosystem service assessments and decisions under climate change, which can involve using sophisticated methods and including multiple services, drivers of change, and sources of uncertainty. Yet acquiring the data (and expertise) to accurately assess and incorporate these complexities is likely to be costly and/or time consuming. However, this investment could lead to substantial improvement in outcomes (or cost savings) in cases where the inclusion of this additional information substantially changes the management strategy or policy (e.g., Runting et al. (2013)). Alternatively, unnecessary time and resources may be spent on incorporating multiple drivers, quantifying uncertainty and improving data quality for outputs that ultimately do not change the decision (e.g., Grantham et al. (2008) and Pannell (2006)). Consequently, an important area for future research is quantifying the value of including multiple drivers and sources of uncertainty into complex models for ecosystem service assessments and decisions. Similarly, assessing the individual *and* cumulative impacts of multiple uncertain drivers of change could be useful in revealing which drivers (or combination of drivers) have the greatest bearing on results and should therefore be prioritized for inclusion in future ecosystem service assessments.

2.5.1 Conclusions

Our systematic review revealed multiple gaps in the body of literature assessing the impacts of climate change on ecosystem services. Cultural services were under-represented, and studies on the USA and Europe dominated the literature. Overall, climate change and other drivers negatively impacted ecosystem services, but this varied across drivers, the services assessed, the context of the study and the method used. This highlights the importance of conducting local and regional ecosystem service assessments, rather than relying on averages or aggregates from other contexts. Although uncertainty was usually incorporated, there were substantial gaps in the sources of uncertainty included, along with the methods used to incorporate them. We found that relatively few studies integrated decision making, and even fewer studies aimed to identify solutions that were robust to uncertainty.

Climate change can have a significant impact on the effectiveness of management decisions targeted at sustaining ecosystem service provision (Poiani *et al* 2010). For management and policy to ensure the delivery of ecosystem services, an integrated approach that incorporates multiple drivers of change and accounts for multiple sources of uncertainty is needed. Explicitly incorporating the range of uncertainties into assessment methods is vital for meaningful integration with decision making (Gregr and Chan 2014). It is concerning that the relatively few studies that incorporated decision making did not assess how well their proposed solutions performed under the range of uncertainties. Making good decisions with limited information and substantial uncertainty will require innovative methods, such as the use of robust optimization (Hallegatte 2009). Whilst this is undoubtedly a challenging task, ignoring this uncertainty could result in misleading assessments of the impacts of climate change, sub-optimal management outcomes, and the inefficient allocation of resources.

3 Costs and opportunities for preserving coastal wetlands under sea level rise

This chapter is reproduced from the following paper, with some alterations to formatting and structure:

Runting, RK, Lovelock, CE, Beyer, HL, & Rhodes, JR. 2017. Costs and opportunities for preserving coastal wetlands under sea level rise. *Conservation Letters*. 10(1):49–57. dx.doi.org/10.1111/conl.12239

3.1 ABSTRACT

Rises in sea level can alter the distribution of coastal wetlands through migration landward and loss due to inundation. The expansion of coastal developments can prevent potential wetland migration, exacerbating loss as sea levels rise. Pre-emptive planning to set aside key coastal areas for wetland migration is therefore critical for the long term preservation of species habitat and ecosystem services, yet we have little understanding of the economic costs and benefits of doing so. Using data and simulations from Queensland, Australia, we show that the opportunity cost of preserving wetlands is likely to be much higher under sea level rise than under current sea levels. However, we found that payments for ecosystem services can alleviate these costs, and in many cases may make expanding the reserve network profitable in the long run. This highlights the need to develop markets and payment mechanisms for ecosystem services to support climate change adaptation policies for coastal wetlands.

3.2 INTRODUCTION

Coastal ecosystems have important biodiversity values, with ~2,700 threatened species globally using these habitats for at least part of their life cycle (IUCN 2013). Additionally, coastal wetlands provide substantial benefits to humans through the provision of ecosystem services, such as the maintenance of fisheries, coastal protection, and carbon sequestration (Barbier *et al* 2011). However, under sea level rise, coastal wetlands can be lost through inundation (Lovelock *et al* 2015), but they can also migrate landward in the absence of steep gradients in topography or anthropogenic barriers, such as built structures (Kirwan and Megonigal 2013). The establishment of anthropogenic barriers to wetland migration could be prevented by pre-emptively expanding the coastal reserve network (i.e. adding to the set of protected areas) to accommodate wetland response to sea level rise. However, we know little about the likely costs and benefits of such an approach. Global sea level rise is one impact of climate change that has seen recent upward revisions as further information becomes available (IPCC 2007, Church *et al* 2013). These revisions, combined with the accelerated subsidence of deltas from anthropogenic activity (such as fossil fuel and water extraction and the trapping of sediment in reservoirs) (Syvitski *et al* 2009), warrants urgent attention and the development of sound pre-emptive adaptation strategies. Despite this imperative, current spending on climate change adaptation remains low relative to the anticipated future costs (Parry *et al* 2009). However, emerging markets for ecosystem services, such as the carbon market (voluntary or otherwise), may have the potential to relieve the financial burden of preserving coastal wetlands under sea level rise.

Previous studies have estimated the impact of sea level rise on coastal ecosystems (FitzGerald and Fenster 2008, Craft *et al* 2009) and the species that depend on them (Traill *et al* 2011, Iwamura *et al* 2013), but none have quantified the costs of preserving wetlands under increasing rates of sea level rise and the potential of payments for ecosystem services to mitigate this cost. There has been a focus on the costs arising from human displacement or damage to private property and infrastructure (Dasgupta *et al* 2009, Bin *et al* 2011, Arkema *et al* 2013, Hinkel *et al* 2014), but there has been little consideration of the costs of preserving wetlands to facilitate their migration. Setting aside land for wetland migration has an opportunity cost, as this land might have otherwise been developed (e.g. for urban use) (Mills *et al* 2014). Whilst the human element is undoubtedly important, it is vital that strategies to preserve wetlands under climate change are considered alongside anthropocentric impacts in order to conserve species and ecosystem services.

The aims of this research were to (i) determine if the opportunity costs of preserving coastal wetlands is higher under sea level rise compared to current sea levels, and (ii) determine the extent to which potential payments for ecosystem services can alleviate these costs. Here we show that, because coastal land value increases with elevation, coastal wetlands are likely to migrate into more expensive land with sea level rise, thus increasing the costs of pre-emptively preserving those wetlands. We also demonstrate that, even when the area of coastal wetlands is projected to expand under sea level rise, the cost of preserving these wetlands is still likely to be greater with sea level rise than without it. Despite the higher costs of preserving wetlands under sea level rise, we show that payments for ecosystem services have the potential to offset the opportunity cost of the reserve network.

3.3 METHODS

To establish why preserving coastal wetlands might cost more under sea level rise we quantified the relationship between coastal land values and elevation for the state of Queensland, Australia. We then undertook a local scale case study to compare the cost of expanding the reserve system with and without sea level rise and payments for ecosystem services, to determine the change in costs and potential of ecosystem services (Figure 3.1).

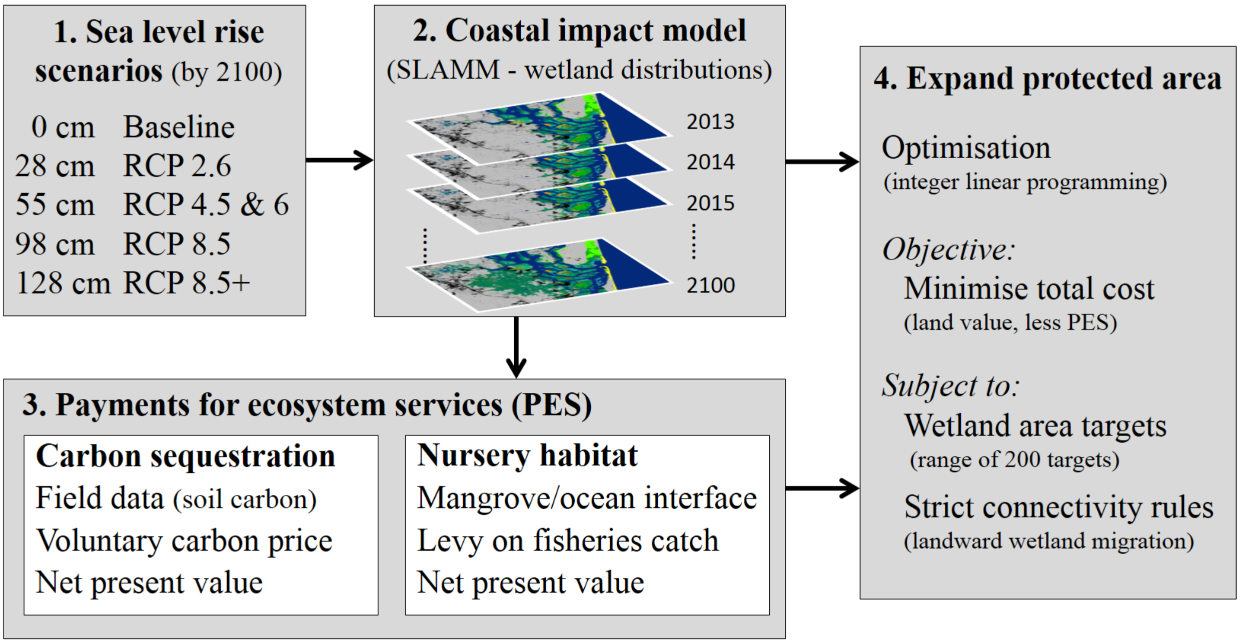


Figure 3.1 | Diagram of the methodology used to expand the reserve network under a range of sea level rise scenarios and potential payments for ecosystem services. The Sea Level Affecting Marshes Model (SLAMM) was used to simulate coastal wetland change under a range of sea level rise projections. This produced a map of coastal wetlands for each year to 2100 for as section of Moreton Bay, Queensland, Australia. Based on these wetland distributions, we modelled the provision of ecosystem services (carbon sequestration and nursery habitat for commercially important species) at each time step, and calculated the net present value of potential payments for these services. Using integer linear programming, we then optimised the selection of additional wetland sites under the range of sea level rise projections and compared the resulting opportunity cost under different combinations of payments for ecosystem services. This allowed us to determine the potential of payments for ecosystem services to compensate the cost of reserve expansion under sea level rise.

3.3.1 Coastal land value and elevation

To understand how land values vary with elevation we quantified the relationship between coastal land values and elevation for the entire 6,973 km coastline of Queensland. This coastline traverses 5 global ecoregions (WWF 2000) and 4 climatic zones (equatorial, tropical, subtropical and grasslands) (Stern *et al* 2000), with human settlement patterns varying from urban to remote (Pink 2011). As extensive elevation data were required, we used a 1 second (~ 30 m) Digital Elevation

Model (DEM) (Gallant 2010). We obtained unimproved land values for 2012 from the Queensland Valuation and Sales database (DERM 2013) and converted these into a value per hectare at a resolution of ~30 m (to match the elevation data). We then categorised the DEM into 100 classes based on 10 cm elevation increments up to 10 m above sea level. These categories were used to derive the mean land value for each 10 cm interval of elevation. To determine the effect of urban, regional or remote areas on this pattern, we separated the results based on the remoteness classes from the Australian Statistical Geography Standard Remoteness Structure (Pink 2011).

3.3.2 Wetland transition model

The Sea Level Affecting Marshes Model (SLAMM, (Clough *et al* 2012)) was used to predict wetland transitions under sea level rise for a 600 km^2 section of Moreton Bay, Australia (Figure 3.3a). SLAMM simulates the main processes driving coastal wetland conversions and shoreline modifications under sea level rise, including salt water intrusion, erosion and sedimentation, wetland transition dynamics, and anthropogenic barriers to these dynamics (Craft *et al* 2009, Clough *et al* 2012). When executed, SLAMM calculates the relative change in elevation and associated wetland transitions for each cell in each year through to 2100. The inclusion of these processes at a fine spatial and temporal resolution enables SLAMM to give an accurate assessment of sea level rise, particularly when combined with LiDAR-derived elevation data (McLeod *et al* 2010, Geselbracht *et al* 2011). Moreton Bay was chosen because it is located near two urban centres (Brisbane to the north and the Gold Coast to the south) and contains a variety of ecosystem types, along with agricultural land.

We parameterised SLAMM for Moreton Bay with a combination of field based and remotely sensed data for the area. Elevation data were derived from Light Detection and Ranging (LiDAR) data based on Airborne Laser Scanning data from 2009 (provided by the Queensland Department of Environment and Resource Management). This dataset was scaled up to a spatial resolution of five metres for incorporation with SLAMM. The absolute elevation accuracy (relative to the Australian Height Datum 71) has a root mean square error (RMSE) of 0.06 m at the 95% confidence level (Traill *et al* 2011). We used averaged data across the region (from Lovelock et al (2011)) for the net surface elevation change, which was set at 1.21 mm yr^{-1} for salt marsh (samphire/claypan) communities. For lower elevation mangrove communities, the rate of surface elevation change was set at -1.95 mm yr^{-1} (i.e. subsiding at mean sea level), increasing linearly to 1.03 mm yr^{-1} at 0.7 m above AHD, which aligns with the upper edge of mangroves. Data were used from Traill et al.

(2011) for overwash events (1 in 25 years), mean tide level (-0.01 m relative to AHD), tidal range (1.53 m) and the salt boundary (1.26 m above the mean tide level) and the current distribution of vegetation, wetlands, and land use.

As the future rise in sea level is uncertain, we used a range of projections to 2100 (28 cm, 55 cm, 98 cm and 128 cm) from the IPCC's fifth assessment report (Church *et al* 2013) to account for this variation. The lower projection of 28 cm is the minimum (5^{th} percentile) value from the representative concentration pathway (RCP) 2.6. This scenario assumes that global annual GHG emissions peak around 2010-2020, and decline substantially thereafter. The mid-range estimate of 55 cm is the median value from RCP 6. We did not model RCP 4.5 separately, as the median value was very similar to RCP 6 (53 cm). The first upper estimate of 98 cm is the maximum (95^{th} percentile) value from RCP 8.5 which assumes business as usual, and emissions continue to rise throughout the century. However, there are potential additional contributions from the collapse of the marine-based sectors of the Antarctic ice sheet (Church *et al* 2013). If initiated, this could cause global mean sea level to rise substantially above the likely range (Hansen 2007, Vermeer and Rahmstorf 2009, Joughin *et al* 2014). Whilst this additional contribution cannot yet be precisely quantified, the IPCC report estimates that its contribution would not exceed several tenths of a meter (Church *et al* 2013), so we included an additional upper estimate of 128 cm. We did not adjust these global estimates to account for regional variation in sea level rise as regional projections of sea level rise for the study region are similar to the global means (Church *et al* 2013). When combined with SLAMM, these projections produced fine resolution (~5 m) simulations of changes in the distributions of wetlands for each year (2013-2100) for each sea level rise scenario.

3.3.3 Ecosystem services

Whilst there are a range of ecosystem services provided by coastal wetlands, we focused on quantifying and valuing soil carbon sequestration and nursery habitat value for commercially important species. To quantify soil carbon sequestration, we used local field measurements for the different wetland types, and applied a range of carbon prices from the voluntary carbon market and estimates of the social value of carbon. For mangrove communities we extracted the mean soil carbon sequestration value (76 g C m^{-2} year $^{-1}$) from a field based study carried out in Moreton Bay (Lovelock *et al* 2014). We focused on soil carbon as this represents the vast majority of carbon storage in these ecosystems (Donato *et al* 2011). For saltmarsh communities, as there is substantial variation in the amount of carbon sequestered across Moreton Bay, we separated these communities into 'high' and 'low' carbon sequestration categories and applied the mean from the high (304 g C m^{-2} y^{-1}) and low (9.6 g C m^{-2} y^{-1}) values from Lovelock et al (2014). The high and low carbon sequestration saltmarsh communities were categorized in accordance with their South East Queensland Wetland class (Dowling & Stephens 1998), based on the dominant vegetation reported in Lovelock et al (2014) and field observations. This resulted in sedgelands (class 6A-D), grasslands (class 4B-D) and casuarina (class 5A-C) being defined as high carbon sequestration communities, with claypan (class 2) and samphire (class 3A) being defined as low carbon.

To determine the value of the carbon sequestered, we applied a range of values from the 2012 voluntary carbon market to these measurements of annual sequestration. We used the mean across all standards (US$5.9 converted to AUD$6.11 MgC^{-1} using the mean exchange rate from 2012 (OzForex 2013)) as the base estimate. The lower bound was represented by the mean of the Chicago Climate Exchange (CCX) (US$0.12, AUD$0.124 MgC^{-1}), and the upper bound was the mean of the Gold Standard (US$9.3, AUD$9.63 MgC^{-1}). To incorporate more comprehensive carbon accounting, we also applied values for the total economic damages from emitting an additional MgC^{-1} (i.e. the social value of carbon). These estimates range from USD $9.55 (Nordhaus 2007) to $84.55 (Stern 2007) MgC^{-1}, which converts to $10.94 and $96.94 2012 AUD respectively.

To determine the area of mangroves which were of nursery habitat value, we first identified three species which were both commercially important and entirely dependent on mangroves for at least part of their life cycle in Moreton Bay. These species were the banana prawn (*Penaeus Merguiensis*), mud crab (*Scylla serrata*), and barramundi (*Lates calcarifer*) (Manson *et al* 2005). However, these species do not utilise all areas of the mangrove forests equally. The mangrove-water interface has repeatedly been shown to be of much greater importance than other mangrove areas as nurseries for commercially important species (Vance *et al* 1996, Loneragan *et al* 2005, Manson *et al* 2005, Meynecke *et al* 2007, Aburto-Oropeza *et al* 2008, Blaber 2013, Zavalloni *et al* 2014). However, there is some uncertainty about what constitutes the mangrove fringe ranging from the linear edge of the mangroves to the first 10 m from the water's edge. To account for this uncertainty, we calculated the spatial component of nursery habitat in three different ways: (i) the length of the interface between mangroves and water as a linear feature, (ii) the area of a 5 m landward strip from the mangrove-water interface, and (iii) the area of a 10 m landward strip from the mangrove-water interface. We used the 5 m strip for the main analyses, but have included the results from using the linear feature and 10 m strip in the variation shown in Figure 3.7, Figure 3.8 and Table C.2.

To determine the value of the mangrove fringe, we used local catch data from 1988 to 2005 for the three commercially important species (DAFF 2006). We took the mean annual Gross Value of Production (GVP, in AUD, which was adjusted to 2012 values (RBA 2014)) over the time period and assumed a linear relationship with each spatial component (Table 3.1). Whilst the GVP is likely to overestimate the contribution of mangroves to producing the catch of a given species (as the contribution of fishing effort to the GVP is not accounted for), the total value may be an underestimate as we did not consider the value of other associated coastal wetlands (e.g. salt marsh (Saintilan *et al* 2007)), or the catch of other commercially important species that benefit from mangroves, but are not dependent of them. However, in practice, a payment for nursery habitat services would be unlikely to reflect the total value of production. To address this, we calculated a 4% levy on the GVP, which is in line with similar levies in the region (Fisheries and Other Legislation Amendment Regulation (No. 1) 2006 (*QLD*)), and more accurately reflects other nursery habitat payment schemes (Lau 2013). Payments flowing from this levy were included in the main analyses, however we also included potential payments for the total value of production as part of the sensitivity analysis.

Table 3.1 | The mean nursery habitat value and total site value based on the linear feature, 5 m strip and 10 m strip. The total site value is based on the current wetland extent. The mean total value represents the total value per unit area. The mean levy value represents the potential payment per unit area based on a 4% levy on the gross value of production.

	Linear feature	*5 m strip*	*10 m strip*
Total for site	522.6 km	256.6 ha	504.0 ha
Total site value	\$847,930.6 yr^{-1}	\$761,601.2 yr^{-1}	\$761,798.1 yr^{-1}
Mean total value	\$1,622.7 $km^{-1}\ yr^{-1}$	\$2,967.6 $ha^{-1}\ yr^{-1}$	\$1,511.5 $ha^{-1}\ yr^{-1}$
Mean levy value	\$64.9 $km^{-1}\ yr^{-1}$	\$118.7 $ha^{-1}\ yr^{-1}$	\$60.5 $ha^{-1}\ yr^{-1}$

The potential annual payments for carbon sequestration and nursery habitat are not comparable to the upfront cost of setting aside land. Therefore, we transformed these potential annual payments into a net present value in 2012 (to match with the year of land valuation), including annual payments up to 2100 (the final year of sea level rise projections) based on the annual simulations of wetland change. The net present value was calculated using the standard equation:

$$NPV(i,N)=\sum_{t=0}^{N}\frac{R_t}{(1+i)^t}$$

where t represents the time of the cash flow (i.e. year 0 [2012] to year 88 [2100]), R_t equates to the potential annual revenue from payments for ecosystem services at time t, N represents the total number of periods (89), and i equates to a conservative discount rate of 10% (varied from 5-15%). This process was repeated for the range of carbon prices and nursery habitat values (68 combinations, Table C.4). This produced a range of values that were appropriate to compare with the opportunity cost of reservation.

3.3.4 Finding the optimal reserve network

We used integer linear programing to find the optimal pre-emptive reserve network (i.e. a group of protected sites) (Beyer *et al* 2016) for a range of wetland area targets for the least cost. Property boundaries were used as the spatial unit for analysis, as this is the level at which land would be set aside for inclusion in a reserve system (Naidoo and Adamowicz 2006). The spatial extent of all wetland types in every year up to 2100 were used to clip the property boundaries for each sea level rise scenario (i.e. if an area did not contain any wetlands in any year up to 2100, it was excluded from the analysis). This resulted in 4192, 5713, 6083, 6850, and 7224, property parcels for the 0 cm, 28 cm, 55 cm, 98 cm and 128 cm SLR scenarios respectively. Data on unimproved land values (DERM 2013), plus a $20,000 AUD transaction cost per property (Adams *et al* 2011), were used as the opportunity cost of setting aside areas for wetland migration. A land value of $0 was applied if property parcels were absent (which occurred in some areas with very low elevation), or if the property was contained within the current reserve network. Each property parcel was either set aside for wetlands (i.e. protected), or assumed to be lost to future development. The general form of the optimization is:

minimise: $\sum_{i=1}^{N} c_i x_i$

subject to: $\sum_{i=1}^{N} r_i x_i \geq T$

$$\sum_{j\in M_i} x_j - m x_i \geq 0, \quad i \in N$$

$$x_i \in \{0,1\}, \quad i \in N$$

where x_i is a binary variable determining whether property i is selected (1) or not (0). The cost variable, c_i, was adjusted to represent different scenarios of payments for ecosystem services. In the case of no payments for ecosystem services, c_i represents the opportunity cost (here unimproved land value and transaction cost) of setting aside property i. When considering scenarios of payments for ecosystem services, c_i represents the opportunity cost of the property less the capitalised value of payments for ecosystem services for that property. The first constraint ensures targets are met. Here, r_i is the area of wetlands contained in property i, and T is the minimum wetland area to be preserved. We used 200 different targets at equal intervals ranging from zero to 80% of the total wetland area in each sea level rise scenario.

The second constraint enforces spatial dependencies among planning units to ensure that neighbouring seaward parcels are also set aside, to allow for the process of wetland migration. Here, M_i is the set of all neighbours adjacent to property i that had wetlands present in any previous year and the constant m determines which of two rules were evaluated: planning unit i can be selected if all adjacent seaward neighbours are also selected (wherein m is the count of these neighbours), or planning unit i is selected if at least one of the neighbours is selected (m=1). The first, stricter connectivity requirement is likely to slightly overestimate the property parcels required, whereas the second, more flexible constraint may result in an underestimate (Table C.3). As such the true connectivity requirement would likely fall between these two estimates.

We implemented the integer linear programming problem using the R programming language (R Core Team 2012), and solved it using the software Gurobi (Gurobi Optimization Inc. 2014). All models were solved to completion, resulting in exact solutions. These solutions (i.e. reserve networks) were compared based on the total cost of the solution, the area of wetlands and nursery habitat preserved, along with the amount of carbon sequestered within the reserve network.

3.4 RESULTS

3.4.1 Land value and elevation

Our analysis of coastal land values and elevation for the coastline of Queensland, Australia showed a generally positive association between land value and elevation in the narrow coastal strip (up to 10 m above sea level, Figure 3.2). The positive relationship was most apparent in major cities and regional settlements, but values were consistently low in remote areas (Figure 3.2). This rise in land values for cities and regional settlements is likely due to the declining flood risk with elevation. The shapes of the curves differ as the confounding drivers of land value (such as slope, accessibly, and amenity) are regionally variable.

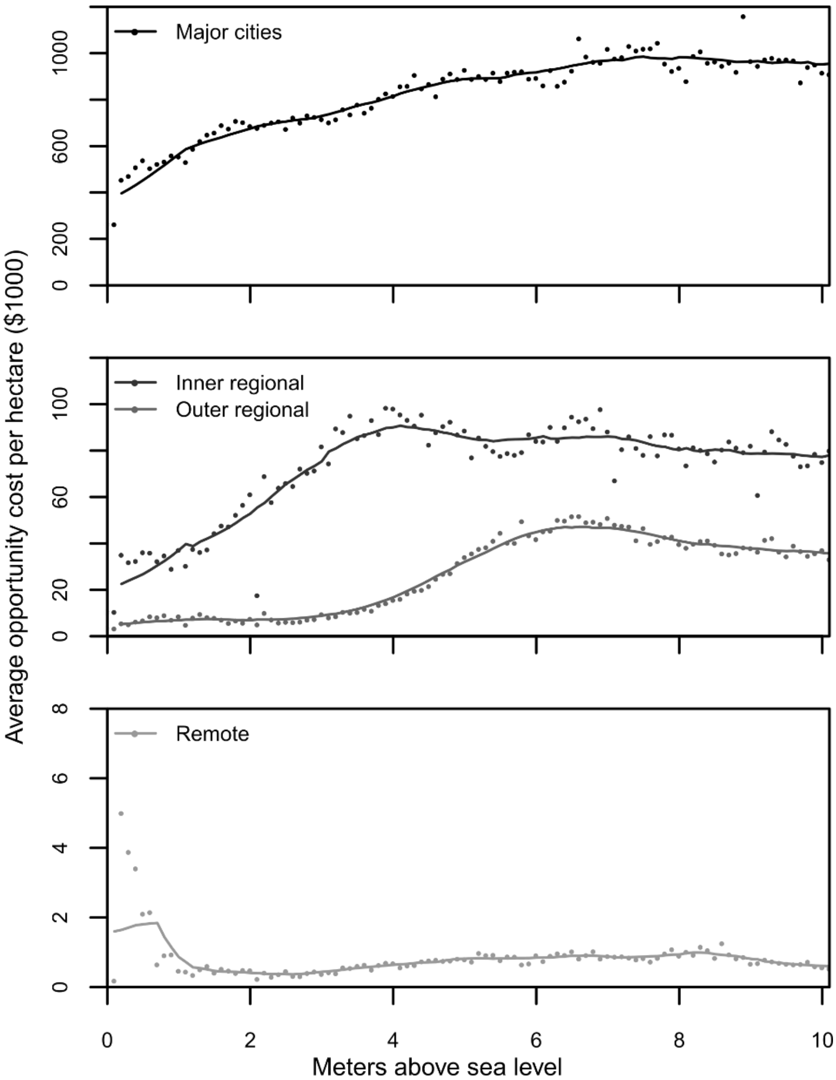


Figure 3.2 | The average (mean) value of coastal land at increasing elevation in Queensland, Australia, separated by remoteness class. The remoteness classes are categorised based on the level of accessibility to remoteness to various service centres via the road network (Pink 2011). Trend lines indicate the moving average.

3.4.2 Cost of reserve network

We predicted a substantial change in the distribution and extent of wetlands under sea level rise for the case study in Moreton Bay, Australia (Figure 3.3). Under the current reserve network, the landward movement of wetlands resulted in fewer wetlands protected under sea level rise. We estimated a loss of 4-31% of the current area of protected wetlands, with higher sea level rise scenarios resulting in lower levels of protection, despite an overall increase in wetland extent (Figure 3.4).

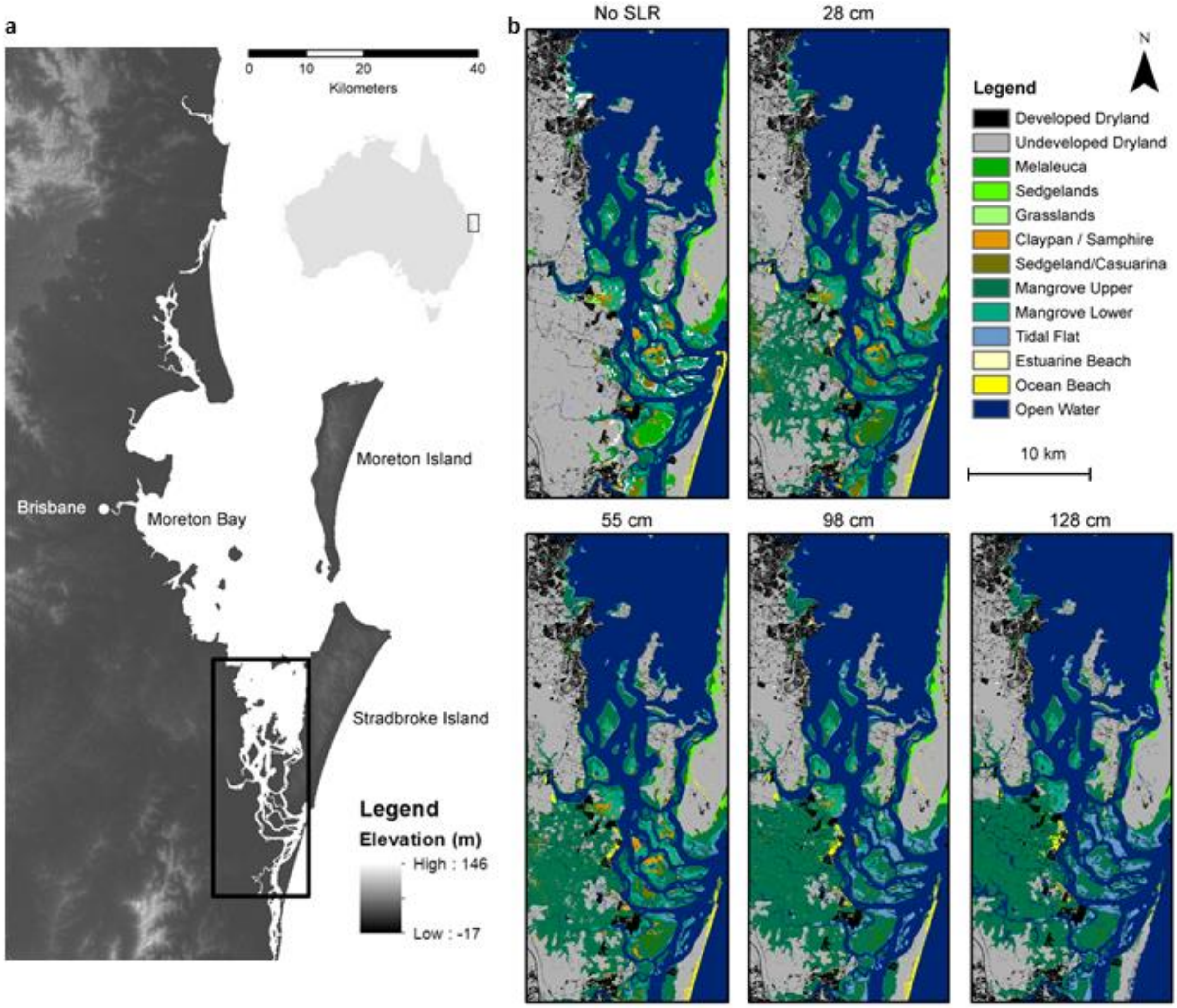


Figure 3.3 | The distribution of coastal vegetation in the south of Moreton Bay, Australia. Panel (a) shows the location of the case study (specifically latitude 27.3°S to 27.5°S and longitude 153.15°E to 153.25°E), and panel (b) shows the distribution of coastal vegetation in 2100 based on no sea level rise (SLR), a rise of 28 cm, 55 cm, 98 cm and 128 cm.

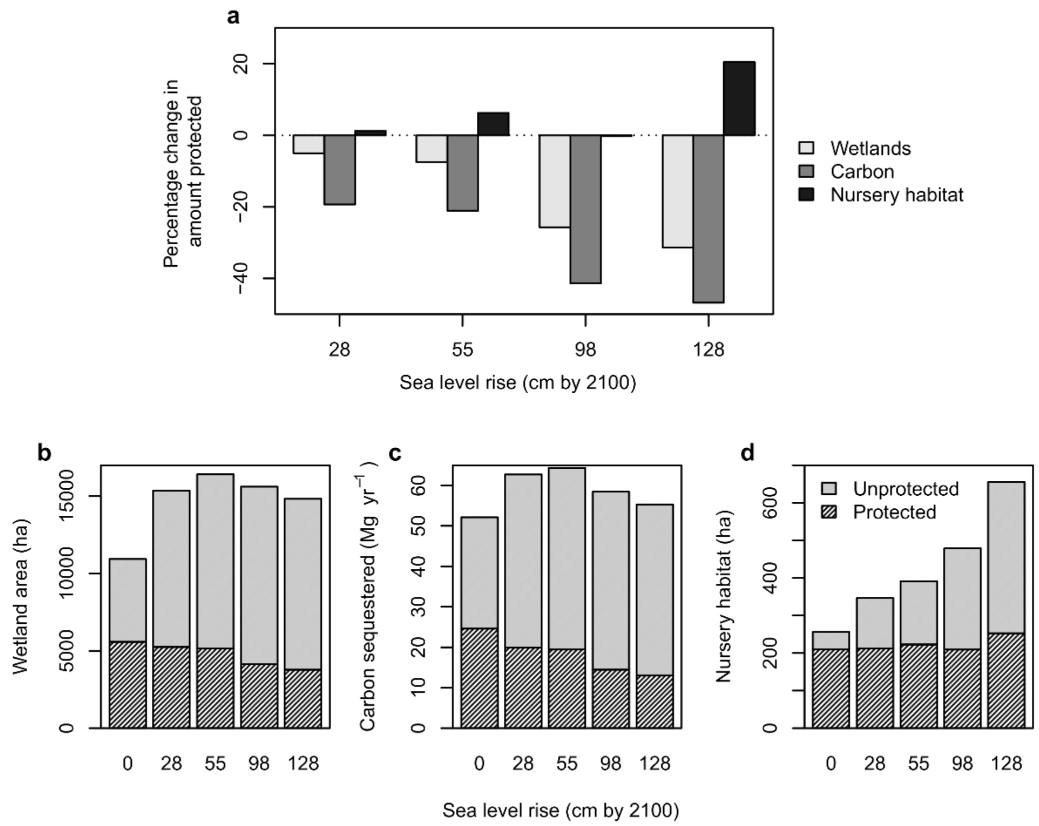


Figure 3.4 | The change in the provision of wetlands and ecosystem services under sea level rise. Panel (a) shows the percentage change in the area of wetlands (wetlands), amount of carbon sequestration (carbon), and area of nursery habitat for commercially important species (nursery habitat) under sea level rise based on the current reserve network. The remaining panels show the area of wetlands (b), amount of carbon sequestration (c), and area of nursery habitat for commercially important species (d) that would be protected and unprotected in 2100 based on the current reserve network in Moreton Bay. 'Protected' refers to areas that are currently contained within the reserve network, and 'unprotected' refers to all other areas. Exact values are given in Appendix C (Table C.1).

Therefore, to maintain the area of wetlands protected under future sea level rise, additional resources are required to expand the reserve network to allow for wetland migration. Under the lower rates of sea level rise (28 and 55 cm), matching the current level of protection would only require a modest additional investment (up to $40,000 AUD), yet a much larger investment is required under the higher rates of sea level rise (98 and 128 cm, a 377% [$151,000 AUD] and 677% [$271,000 AUD] increase respectively over lower rates of sea level rise) (Figure 3.5, Figure 3.6). Further, increasing the level of protection beyond current levels exacerbates the increase in cost even further. For example, under current sea levels, a 20% increase in the area of wetlands protected would cost $105,000 AUD, with much of this target being met on public lands. However, as coastal wetlands move landward onto private land under the higher sea level rise scenarios, the required investment to match this target could be up to $1.3 million AUD (a 1,138% increase over current sea levels, Figure 3.5).

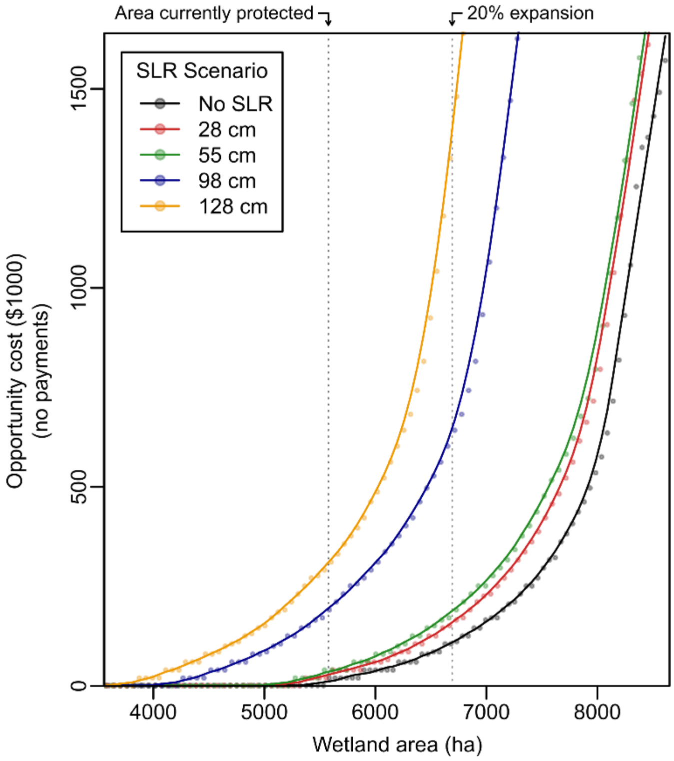


Figure 3.5 | The total cost of preserving increasing wetlands under different rates of sea level rise (SLR) in the absence of payments for ecosystem services. Dotted lines indicate the area of wetlands that are currently contained within the reserve network (5577 ha), and a 20% expansion of the area of wetlands protected (6692 ha).

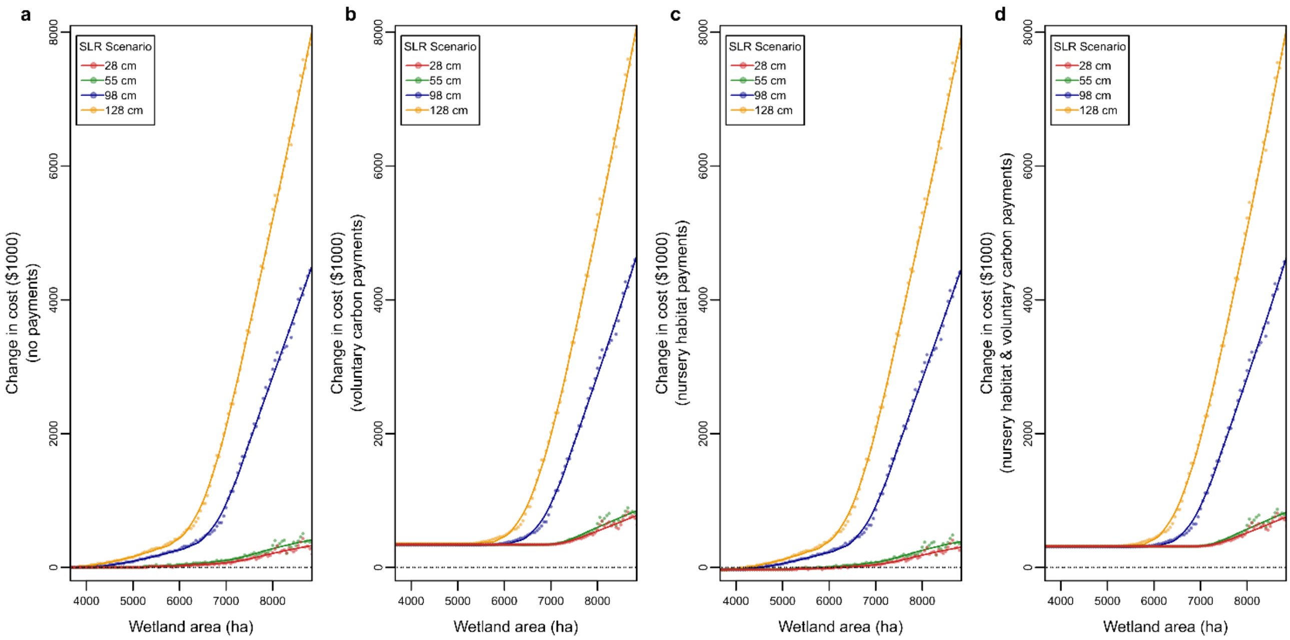


Figure 3.6 | The change in cost of preserving wetlands under increasing rates of sea level rise (SLR) and different market conditions when compared to the baseline (no sea level rise). Panel (a) shows the increase in cost with sea level rise in the absence of any payments for ecosystem services ('no market'). Panel (b) shows a greater increase in cost due to sea level rise (relative to the baseline) in the case of an active voluntary carbon market. Panel (c) shows the change in cost in the presence of nursery habitat payments, but in this case the cost could be slightly reduced (or profit increased) with lower sea level rise projections and wetland targets. Panel (d) shows the change in cost from stacking voluntary carbon payments and nursery habitat payments. Whilst payments for ecosystem services generally increased the *change* in cost (relative to the baseline), the *overall* cost was reduced for all sea level rise scenarios, and in many cases resulted in a profit.

3.4.3 Payments for Ecosystem Services

Payments for ecosystem services have the potential to attenuate the opportunity costs of protection. We found that a carbon payment alone (at \$6.11 MgC^{-1} AUD) completely compensated for the cost of protecting an additional 32-33 km^2 of wetlands (a ~60% increase over the current reserve network) under the baseline (0 cm) and lower sea level rise scenarios (28 and 55cm, Figure 3.5). However, under higher rates of sea level rise (98 cm and 128 cm), including a carbon payment only compensated for the cost of protecting an additional 20 km^2 and 15 km^2 (a 37% and 27% increase from the current reserve network) respectively (Figure 3.7). Stacking carbon payments with a potential nursery habitat payment provided only a modest additional expansion over carbon payments alone (up to an additional 1.3 km^2 [~2% increase]), as the most cost-efficient areas for nursery habitat were already selected by a payment for carbon (Figure 3.7). Protecting a smaller area of wetlands (than given by the above values) would be more than compensated for by ecosystem service payments, as the capitalised value of the ecosystem services exceeded the opportunity cost of the reserve network (Figure 3.8).

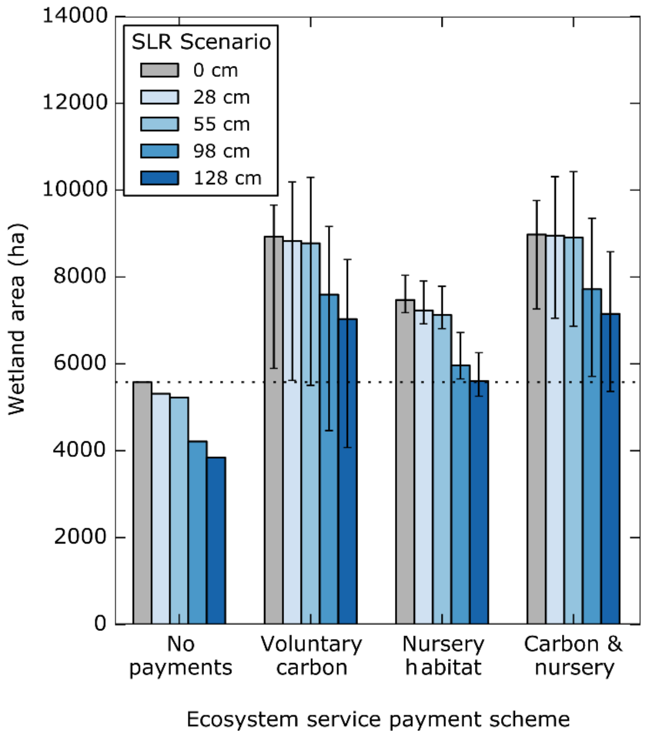


Figure 3.7 | The maximum area of wetlands that can be preserved and still ‘break-even’ (\$0 cost) under different sea level rise (SLR) scenarios and payments for ecosystem services. The ‘break even’ point is where the capitalised revenue from ecosystem service payments exceeds the opportunity cost of expanding the reserve network. ‘No payments’ refers to the baseline case where there are no payments for any ecosystem services. ‘Voluntary carbon’ is the result of an active voluntary carbon market with recent (2012) carbon prices. ‘Nursery habitat’ refers to payments that could flow from a levy on the gross value of production for commercially important and mangrove dependent species. ‘Carbon & nursery’ is the result from stacking payments for carbon and nursery habitat. Error bars represent the minimum and maximum wetland area based on variations in discount rates, voluntary carbon payments, and the method used to calculate the amount of nursery habitat. The dotted line indicates the wetland area that is currently contained within the reserve system (5577 ha).

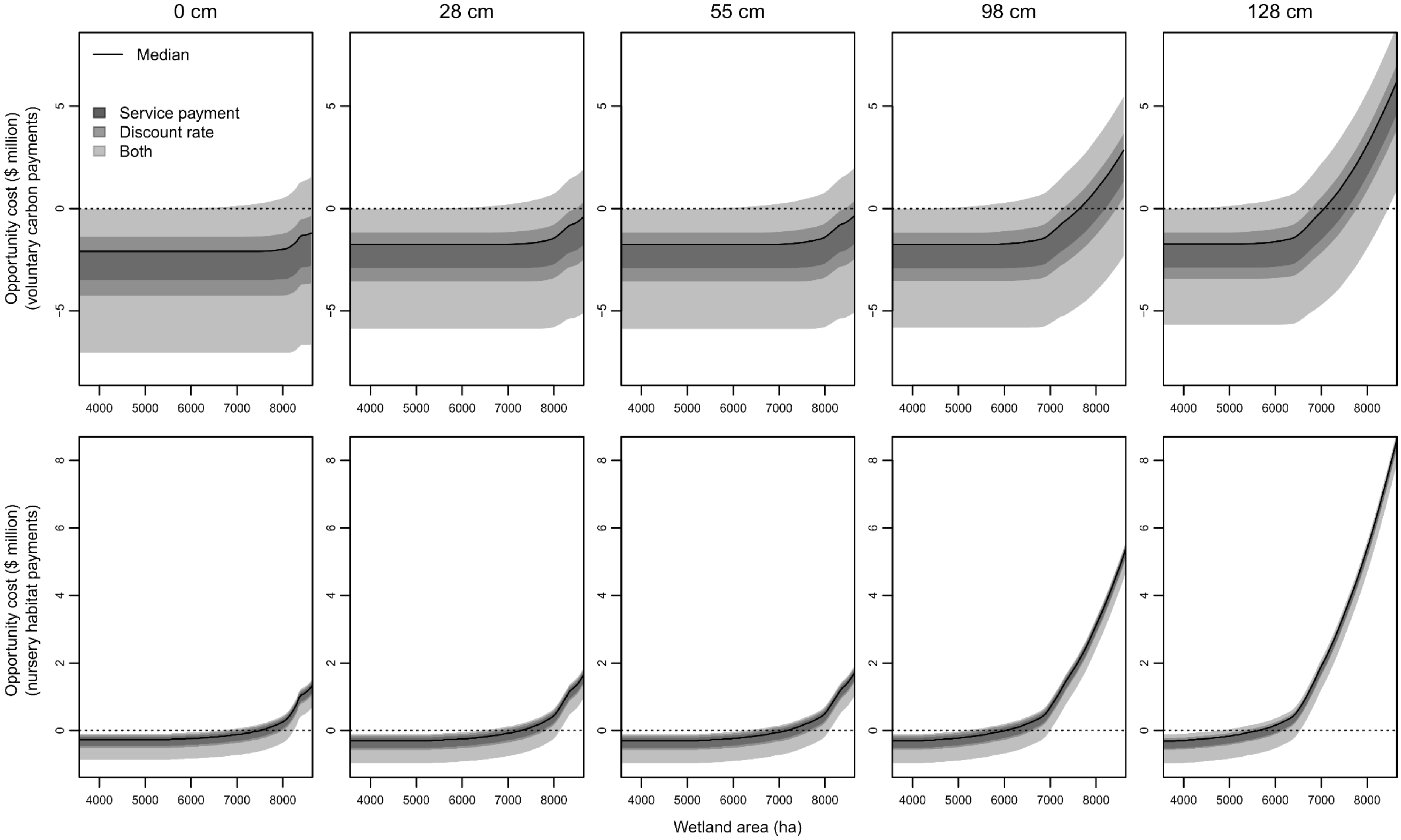


Figure 3.8 | The variation in the potential for ecosystem services to attenuate the costs of preserving wetlands under sea level rise. The shaded areas for carbon and nursery habitat payments represent the uncertainty from varying the discount rate, the method for calculating nursery habitat, and the carbon price. Negative costs indicate a net gain (profit).

3.5 DISCUSSION

We have shown that substantial changes in the distribution of coastal wetlands under seal level rise are likely to lead to increases in the costs of protecting them. Consistent with other studies, we predicted a landward movement of wetlands (particularly mangroves) under sea level rise (Traill *et al* 2011, Di Nitto *et al* 2014, Saintilan *et al* 2014) (Figure 3.3b). This landward movement, combined with the positive association between land values and elevation (Figure 3.2) drives the increase in cost of pre-emptively protecting wetlands to facilitate landward wetland migration under sea level rise. In fact we show that the higher the sea level rise projection, the higher the opportunity cost of expanding the protected area network (Figure 3.6). This higher cost of preserving coastal wetlands is likely to be a general consequence of sea level rise, particularly in regions where the potential for urban development places further upward pressure on coastal land values.

Despite these higher costs, payments for ecosystem services have the potential to substantially reduce the net cost of expanding the reserve network under sea level rise. It is possible that the benefits from payments for ecosystem services could be further increased under different market conditions. For example, even more wetlands could be preserved if the carbon price reflected the social value of carbon (i.e. the total economic damages from emitting an additional 1 MgC^{-1}), or if these higher carbon payments were combined with those for the total value of nursery habitat. In both of these cases, the capitalised values of the services exceed the opportunity cost for all modelled wetland targets (up to 80% of the total wetland area in each scenario) (Table C.2). Furthermore, including payments for additional ecosystem services not quantified here, such as storm protection or nutrient retention, would likely increase the economic benefits of coastal wetland protection.

Whilst receiving payments for ecosystem services reduces the costs of coastal wetland protection for local planning authorities, this cost is shifted to the beneficiaries of the services. Carbon sequestration has potential buyers in both the public and private sectors, and transactions can be facilitated through the relatively well-established voluntary carbon market (Hamrick *et al* 2015). In this case, shifting the cost burden to the buyer is unlikely to be problematic, as the buyers' participation is voluntary (such as individuals who purchase voluntary carbon offsets for air travel (Mair 2011)). In contrast, a nursery habitat payment shifts the costs to local fisheries via a compulsory levy. This may face opposition from commercial fishers if the additional cost is perceived to threaten the economic viability of their enterprise (Marshall 2007). Given that stacking nursery habitat payments with carbon payments facilitated only a modest additional expansion of the reserve network over carbon payments alone (~2%, Figure 3.7), the additional administrative burden and potential controversy of a nursery habitat levy might not be justified in this case.

It is imperative that local planning authorities pre-emptively limit development in dryland areas that are likely to transition to wetlands under climate change. The primary difficulty in implementing this strategy is that the opportunity costs of purchasing properties or re-zoning land are borne immediately, whereas the benefits take much longer to materialise and often flow to beneficiaries external to the local area (Friess *et al* 2015). Even when the capitalised value of payments for ecosystem services exceed the opportunity cost of expanding the reserve network, the revenue from ecosystem service markets would not start flowing until the wetlands had migrated sufficiently landward. This delay in receiving benefit could explain why this strategy is not adopted in many vulnerable areas, despite the long term advantages. For example, local and state governments along the USA Atlantic coast plan to develop 60% of land below 1m elevation (Titus *et al* 2009), and Australian state governments across the eastern sea board have removed sea level rise from state planning policies (Bell and Baker-Jones 2014). However, climate change adaptation policies are emerging in other areas, such as the Thames Estuary 2100 plan (for London and the tidal reaches of the Thames river) which incorporates a projected sea level rise of up to 1.9 m and includes provisions for intertidal habitat creation (Environment Agency 2012).

Given the dynamic nature of land markets under sea level rise, coastal land may be cheaper in the future as flood risk increases (Bin *et al* 2011). However, this does not necessarily justify local planning authorities delaying the purchase or re-zoning these areas. If new dwellings or other hard structures are permitted in the potential future locations of wetlands or their migration pathways, this will not only impact biodiversity through arresting wetland migration, but will also have socio-economic impacts. For example, the costs may be shifted to the coastal property owner who may face reduced property prices, periodic flooding, or relocation in a worst-case scenario. Furthermore, it may not always be the case that the cost of coastal land will decline. Despite increasing risks, coastal populations are large and growing (Martínez *et al* 2007), which is likely to create upward pressure on land prices in future (Glaeser *et al* 2005). Furthermore, future risks may not be given appropriate consideration (Newell *et al* 2015), particularly if insurance companies are able to compensate damages (Bagstad *et al* 2007) or the impacts of sea level rise are predicted to occur outside of the investors’ outlook.

3.5.1 Conclusions

We have shown here that payments for ecosystem services can alleviate some of the costs of expanding the coastal reserve network under climate change, and in many cases may result in a profit in the long run. These cost reductions are possible because the costs are shifted from planning authorities to the beneficiaries of the services, which may not always be well received. Higher rates of sea level rise can reduce the effect of payments for ecosystem services, which highlights the importance of ambitious climate change mitigation efforts alongside adaptation plans. Although profits are possible in the long run, planning authorities may be strained in the short term, as some of the revenue from ecosystem service payments would not be received until wetland migration occurred. Alternatively, delaying the implementation of climate change adaptation policy may risk losing key areas of coastal wetlands, the species they support, and services they provide.

4 Risk-sensitive conservation planning under climate change: A case study of coastal ecosystem services under sea level rise

4.1 ABSTRACT

Climate change is expected to impact many species and ecosystem services, though it is difficult to predict when and how these impacts may arise. It is challenging to account for this uncertainty when planning management actions intended to mitigate these impacts, such as designating new protected areas. The danger of ignoring uncertainty is that resulting plans may eventually fail to achieve conservation objectives, yet this is not usually incorporated in conservation planning. We adapt an approach for risk-sensitive resource allocation from finance, Modern Portfolio Theory, to conservation planning. The key advantage of this approach is that it accounts for correlations in projected outcomes among sites, in order to identify plans that are likely to achieve multiple conservation objectives across a wide range of climate scenarios, whilst still including typical features of conservation planning, such as connectivity requirements. We exemplify the approach using a case study of conservation planning for coastal wetlands and associated ecosystem services under uncertain rates of sea level rise in Moreton Bay, Australia. This case study is pertinent as sea level rise projections are highly variable and can alter the distribution of coastal wetlands through loss due to inundation and landward migration. We compared our risk-sensitive approach to climate adaptation plans that ignored uncertainty. We found that ignoring uncertainty was a high-risk strategy, even when planning for the worst-case scenario. In contrast, explicitly accounting for uncertainty resulted in solutions that ensured the supply of ecosystem services with relatively low risk of failure across all climate scenarios. This method is likely to be of use in other conservation contexts where the impacts of climate change on species, ecosystems, and their services vary spatially over different climate change scenarios.

4.2 INTRODUCTION

Conservation planning in the context of a changing climate is inherently uncertain (Hoegh-Guldberg and Bruno 2010, Pacifici *et al* 2015). Changes in climate can alter the range of species and the distribution of ecosystems, but the precise extent and direction of these changes are subject to interacting factors, such as invasive species, topology, and ecosystem processes (Pearson and Dawson 2003). These uncertain changes have implications for the services that flow from species and ecosystems, which face similarly uncertain impacts (Scholes 2016, Runting *et al* 2017a). Compounding these uncertainties, future impacts on species, ecosystems, and their services depend on the global greenhouse gas emissions trajectory, which in turn depends on unpredictable national and global efforts to reduce emissions (IPCC 2013). Additionally, these climatic changes do not occur in isolation from other risks to natural capital, such as fire, land-use change, and over-exploitation. Consequently, planning long-term conservation actions, such as the designation of protected areas, are subject to substantial risks that need to be addressed in planning.

Identifying spatial conservation priorities based on different deterministic scenarios of climate change is a common approach to understanding the implications of this uncertainty (for examples see (Bush *et al* 2014, Adams-Hosking *et al* 2015)). In this context, scenario analysis can play an important role in participatory planning (Tress and Tress 2003) by stimulating dialogue and revealing the possible consequences of alternative futures (Deshler 1987, Peterson *et al* 2003). However, selecting an individual climate change scenario on which to base decisions essentially assumes that the future emissions scenario (and potentially also impacts) are known with certainty. Implementing a conservation plan based on a deterministic scenario (or expected mean) could fail to account for potential losses from more extreme changes, or alternatively, potential windfalls from less severe impacts.

Explicitly incorporating the uncertainty surrounding climate change projections into spatial conservation plans requires innovative methods. Previous approaches include methods to minimize or reduce the risk in missing conservation targets due to the impacts of climate change (Game *et al* 2008, Carvalho *et al* 2011b, Maina *et al* 2015), or to improve the robustness of the solution by incorporating info-gap decision theory into spatial prioritization (Moilanen *et al* 2006, Kujala *et al* 2013). Significantly, these approaches assess the risk posed by climate change for each planning unit (or site) *individually* within the optimization or prioritization. However, climate change often produces spatially variable impacts within and across different emissions scenarios (Hijmans *et al* 2005, IPCC 2014), so any pair of planning units could have a similar individual risk (or variance) but different responses to alternative climate change scenarios (covariance). Therefore, assessing risk for individual planning units misses the opportunity to further reduce the overall risk of the final solution by considering the covariances among planning units, and adjusting their selection accordingly (Ando and Mallory 2012a).

Modern Portfolio Theory, is a mathematical framework that allows covariances to be incorporated explicitly. It was originally developed to select a financial investment portfolio (a collection of *assets*) that maximizes expected returns for a given level of risk (or minimises risk for a given level of expected returns) (Markowitz 1952). The overall risk can be reduced by investing in complementary combinations of assets that have negative correlations in returns (or at least a low positive correlation). Ultimately, this method reveals what fraction of the investor's budget to invest in each financial asset to achieve the desired level of returns (or risk) (Markowitz 1952).

Modern Portfolio Theory has previously been applied to conservation problems, with financial assets being substituted for species (Koellner and Schmitz 2006), populations (Moore *et al* 2010), genetic diversity (Crowe and Parker 2008), or ecosystem services (Halpern *et al* 2011). These applications are limited in that they do not use Modern Portfolio Theory to inform the spatial allocation of investments (although Halpern (2011) evaluated the overall impact of random spatial configurations of marine reserves post hoc). However, recent advances have considered spatial planning units as assets, allowing risk to be reduced by allocating conservation investment across space (Ando and Mallory 2012a, Mallory and Ando 2014, Shah and Ando 2015, Shah *et al* 2016). The main drawback of the approach used in these spatial applications is that it does not address the discrete nature of reserve design problems — it is not usually possible to purchase arbitrary portions of land parcels or regions. These approaches also fail to incorporate multiple conservation objectives or how planning regions may be biologically or functionally connected in space.

We extended the approach used by Ando and Mallory (2012a) to overcome these limitations by adapting the problem formulation in several ways to better suit typical conservation planning problems. Firstly, as conservation planning problems often consider a large number of planning units (typically thousands) in which an action (e.g., protection) can either take place or not (Ball *et al* 2009), we adopted a binary decision variable rather than a continuous one. Secondly, our formulation incorporates multiple objectives with relative weightings that the decision-maker can adjust. Additionally, some degree of connectivity between selected sites is usually required to ensure that the final solution is ecologically functional (Beger *et al* 2010a), so we included a connectivity constraint. Finally, we also allowed a budget to be set to ensure the final solution could be feasibly implemented by the decision-maker. Ultimately, this formulation selects a complementary set of connected planning units, for a given budget, that meet a set of conservation objectives while hedging the risk posed by different climate change scenarios. This formulation more closely resembles the types of problems conservation planners typically solve (i.e. with tools such as Marxan (Ball *et al* 2009)), and manages the risk posed by climate change (or other threats).

We illustrate this approach using a case study of conservation planning for coastal wetlands and associated ecosystem services under one aspect of climate change uncertainty, sea level rise and associated wetland response, for a section of Moreton Bay, Queensland, Australia. Planning for coastal wetland migration under sea level rise is challenging due to uncertain changes in wetlands in response to sea level rise (Craft *et al* 2009) along with imperfect elevation data (Gesch 2009) and sea level rise projections (IPCC 2014). Coastal land also faces significant development pressure, which can result in a high opportunity cost in setting aside land to allow for wetland migration (Mills *et al* 2014, 2015, Runting *et al* 2017b). Within this case study we aim to: (i) determine the risk-return trade-offs by adapting Modern Portfolio Theory to conservation planning; (ii) compare scenario-based planning strategies to this approach; and (iii) determine the trade-offs among different conservation objectives, and how these are altered by risk.

4.3 METHODS

4.3.1 Modern Portfolio Theory

The traditional portfolio approach determines how to allocate investment among assets in a financial portfolio. It is generally formulated as either maximising risk-adjusted returns, or minimizing risk subject to achieving a given level of expected returns (Markowitz 1952, Bertsimas and Pachamanova 2008). To maximize risk-adjusted returns, the problem is:

$$\begin{aligned} &\text{maximise} && \mathbf{r}^T\mathbf{w} - \lambda\mathbf{w}^T\Sigma\mathbf{w} \\ &\text{subject to} && \sum_i w_i = 1 \end{aligned} \tag{4.1}$$

where $\mathbf{w}$ is a vector of weights for each investment asset i, $\mathbf{r}$ is a vector of expected (monetary) returns from each asset, Σ is the covariance matrix for the returns on the assets and λ is a term representing risk tolerance where larger values represent higher risk aversion and $\lambda \geq 0$. The expression $\mathbf{w}^T\Sigma\mathbf{w}$ represents the variance in the returns. Individual weights can be negative, which represents the short-selling of assets (i.e., ‘borrowing’ assets and selling them in the expectation the price will drop so that a profit can be made by buying the asset at a lower price at a future time (Arrow and Debreu 1954)). To minimise risk subject to a given level of expected returns, the problem is:

$$\begin{aligned} &\text{minimise} \quad && \mathbf{w}^T \Sigma \mathbf{w} \\ &\text{subject to} \quad && \mathbf{r}^T \mathbf{w} \geq \mu \\ & && \sum_i w_i = 1 \end{aligned} \tag{4.2}$$

where μ is the target level of expected returns, and other terms are as specified above. The initial application of Modern Portfolio Theory to a spatial conservation decision problem (Ando and Mallory 2012a, Mallory and Ando 2014) adopted the latter formulation, replacing financial assets i with planning units, and monetary returns with returns from a conservation index or returns from the conservation index divided by the cost (i.e., land purchase price). Ando and Mallory (2012a) included an additional constraint, to exclude negative weights ($w_i \geq 0$).

The application of modern portfolio theory to spatial conservation planning differs from financial market applications in five key ways. First, there is no analogy to “short-selling” in conservation planning (i.e. `borrowing ‘ assets and selling them in the expectation the price will drop so that a profit can be made by buying the asset back at a lower price at a future time). Thus, negative weights (w) are not permitted in conservation problems (as in Ando and Mallory (2012a)). Second, in finance the problem addressed is what proportion of the total capital should be invested in each asset (a continuous measure). Although this is also applicable to some conservation planning problems, it is instead more common for conservation problems to determine what discrete set of planning units to select in order to best achieve objectives. For example, if assets represent land ownership parcels, it may be necessary to purchase the entire parcel rather than a fraction of it. Third, in contrast to financial markets, in conservation planning there is usually an upper limit to the resources that can be invested in any one asset (planning unit) and this limit is often small relative to the total resources available.

Fourth, conservation problems often consider multiple objectives whether this be multiple species (Wilson *et al* 2011), ecosystems (Giakoumi *et al* 2013) or ecosystem services (Chan *et al* 2006). Although in some cases a single index or indicator is used, this is not possible or desirable in many cases (Lawler *et al* 2003, Fleishman *et al* 2006), particularly as conservation planning is moving towards including a wider array of stakeholder preferences and policy objectives (Runting *et al* 2015). Here we used a weighted sum approach with relative weightings for each conservation objective that can be adjusted by the decision-maker(s). Finally, some degree of connectivity between planning units is usually required in most reserve design problems. This connectivity can take the form of a simple clustering of protected areas to minimise the impacts of habitat fragmentation and edge effects (Klein *et al* 2009), asymmetric connectivity for freshwater systems (Hermoso *et al* 2011), or marine spatial planning for larval dispersal (Beger *et al* 2010b). Here we include a flexible connectivity constraint that can be adjusted based on the strength and direction required for a specific planning problem.

4.3.2 Integrating Modern Portfolio Theory and reserve selection

Here, we combine a portfolio approach (Markowitz 1952, Ando and Mallory 2012a) with a parcel-level reserve design problem for multiple conservation objectives, with budgetary and connectivity constraints. The general form of the model is:

$$\begin{aligned} \text{maximize} \quad & \sum_{k=1}^{K} w_k \sum_{i=1}^{N} r_{ik} x_i - \lambda \mathbf{x}^T \Sigma \mathbf{x} \\ \text{subject to} \quad & \sum_{i=1}^{N} c_i x_i \le B \\ & \sum_{j \in M_i} x_j - m x_i \ge 0, \quad i \in N \\ & x_i \in \{0,1\} \end{aligned} \tag{4.3}$$

where w_k specifies the weight given to conservation objective k ($w \geq 0$; $\sum_k w_k = 1$), N is the number of planning units, r_{ik} is the expected (mean) benefit of planning unit i for objective k, x is the vector of binary decision variables representing whether the planning unit is selected or not, and λ is a term representing the risk tolerance of the decision maker, were larger values represent a higher risk aversion and $\lambda \geq 0$. Σ is the combined covariance matrix for all conservation objectives, and is recalculated for each unique combination of weights, based on the weighted summation of each objective. Summing the conservation objectives prior to the calculating the covariance matrix ensures that potential interdependencies among conservation objectives are accounted for. Returns (and risks) can only be realised if the planning unit is selected.

The first constraint ensures that the sum of opportunity costs (c) among all selected planning units does not exceed the total budget (B). The second constraint enforces connectivity requirements among planning units. Specifically, M_i defines a set of planning units adjacent (or otherwise connected) to planning unit i. M_i can refer to all adjacent planning units, a subset of adjacent planning units (in the case of unidirectional connectivity requirements), or non-adjacent planning units that are functionally connected (Beger *et al* 2010a). The parameter m can take any value between 1 and $|M_i|$. If m is set to $|M_i|$, planning unit i can be selected only if the entire set of given neighbours are also selected; if m is set to 1, planning unit i can be selected only if at least 1 of the neighbours are selected. An even more flexible approach to connectivity can be formulated as a penalty for disconnected planning units in an additional term in the objective function (as described in Beyer et al. (2016)), but here we focus on the former formulation.

4.3.3 Moreton Bay Case Study

We demonstrate the application of our model (Eqn 4.3) to a 400 km^2 section of Moreton Bay and adjacent land in Queensland, Australia (Figure 4.1a). Coastal ecosystems can be lost with climate change losses due to continual inundation from sea level rise (Lovelock *et al* 2015), but they can also migrate landward under the right conditions (Kirwan and Megonigal 2013). The services provided by these coastal ecosystems are particularly vulnerable to climate change, so the application of novel climate adaptation strategies to these systems is valuable (Ruckelshaus *et al* 2013). Moreton Bay was chosen as it is an internationally important wetland site (Ramsar listed), and it is also threatened by further urban development within Australia's fastest developing region, South East Queensland (Department of Infrastructure and Planning 2009).

<u>Coastal impact model</u>

To test a range of planning strategies, we first simulated how the distribution of coastal wetlands could change under sea level rise to the year 2100 for our study site (Figure 4.1a). To simulate wetland change, we incorporated the uncertainties in future sea level rise, elevation data, and other biophysical parameters within the Sea Level Affecting Marshes Model 6.2 (SLAMM) (Clough *et al* 2012). SLAMM simulates the key processes driving coastal wetland conversions under sea level rise, including uplift and subsidence, salt water intrusion, tidal ranges, erosion and sedimentation, wetland transition dynamics, and physical barriers to these dynamics (Craft *et al* 2009, Clough *et al* 2012). SLAMM 6.2 allows a probability distribution to be specified for each input parameter (such as sea level rise and accretion), and the software then samples from these distributions for multiple iterations of wetland change (Monte Carlo simulations) (Clough and Propato 2012).

Parameterising SLAMM requires a range of input data and estimates of uncertainty. Elevation information was derived from Light Detection and Ranging (LiDAR) data from 2009 (provided by the Queensland Government Department of Environment and Resource Management (Traill *et al* 2011)). This dataset was scaled up (from 5 m) to a spatial resolution of 10 m for incorporation with SLAMM, and the uncertainty associated with this dataset was also included (specifically, the absolute elevation accuracy has a root mean square error (RMSE) of 0.06 m at the 95% confidence level (Traill *et al* 2011, Runting *et al* 2013). Data on the distribution of wetland types was sourced from the Queensland Herbarium (Dowling and Stephens 1998), and the extent of urban areas and hard surfaces was sourced from Lyons et al. (2012). Projections of sea level rise in 2100 were based on a the Representative Concentration Pathways (RCP) 2.6 (44 cm [10.3 s.d]), 4.5 (53 cm [10.9 s.d]), 6.0 (55 cm [10.9 s.d]) and 8.5 (74 cm [14.6 s.d]) from the IPCC's fifth assessment report (Church *et al* 2013). These means and standard deviations were used to characterise a normal distribution of sea level rise for each RCP. We did not adjust these global projections to account for regional variation as regional projections of sea level rise for our study region are similar to the global means (Church *et al* 2013). The remaining parameters associated with accretion, erosion, overwash, and tides and their probability distributions are detailed in Table D.1. When executed, SLAMM calculated the change in elevation (relative to sea level) and associated wetland transitions for each combination of parameter samples in each cell in 5 year intervals through to 2100. We ran 200 iterations for each of the 4 RCPs in addition to a deterministic run (i.e., based on the parameter means) for each RCP. This produced 804 simulations (200 iterations + 1 deterministic run for each RCP) of the distribution of wetlands for each 5 year interval. We did not apply any weighting to these scenarios.

Ecosystem services

We modelled two key ecosystem services in Moreton Bay: carbon sequestration and nursery habitat for fisheries. We focused on soil carbon as this represents the vast majority of carbon sequestered and stored in these coastal wetlands (Donato *et al* 2011). We mapped annual soil carbon sequestration for each potential distribution of wetlands in 2100, using field data on soil carbon sequestration rates for different wetland types (Table D.2) (Lovelock *et al* 2014). To account for the uncertainty in soil carbon sequestration rates, we sampled from a normal distribution of rates for mangroves and saltmarsh, and applied one sample to each of the 804 maps of wetlands in 2100 using the Python programming language (van Rossum and the Python Community 2012).

To map nursery habitat for fisheries, we had to determine which wetland areas are important for providing this service. Some commercially important species in Moreton Bay are dependent on mangroves for at least part of their life cycle, including the banana prawn (*Penaeus Merguiensis*), mud crab (*Scylla serrata*), and barramundi (*Lates calcarifer*) (Manson *et al* 2005). It has also been repeatedly demonstrated that the seaward fringe of mangroves is of much greater importance than other mangrove areas as nurseries for commercially important species, both in Moreton Bay and elsewhere (Manson *et al* 2005, Aburto-Oropeza *et al* 2008, Zavalloni *et al* 2014). However, there is some uncertainty about what constitutes the mangrove fringe, with that uncertainty ranging from the linear edge of the mangroves to the first 10 m from the water's edge. To account for this uncertainty, we created a landward strip from mangrove-water interface, and the width of this strip was sampled from a uniform distribution ranging from 1 to 10 m. The sampled widths were applied randomly to the 804 maps of wetlands in 2100 (one sample per map) using the Python programming language (van Rossum and the Python Community 2012).

Optimisation

We applied our model (Equation 4.3) to the Moreton Bay case study to find the optimal reserve configuration for multiple conservation features under risks associated with sea level rise. Property boundaries were used as the spatial unit for analysis (i.e., the units represented by the decision variable vector **x**), and each property parcel was either set aside for wetlands (i.e. protected), or assumed to be lost to future development ($x_i \in \{0,1\}$). The spatial extent of all wetland types in every scenario up to 2100 were used to identify all properties boundaries containing at least 0.25 ha of wetlands (n=1225). The cost of each property, c_i, was calculated as the unimproved land values (DERM 2013), plus a $20,000 AUD transaction cost per property (Adams *et al* 2011). Existing protected areas within the study site were given an opportunity cost of zero, but were not forced to be included in the final solution to allow for greater flexibility in site selection. The total budget, *B*, was set to AUD$50 million, which represents ~3% of the total land value in the study area and was considered to be a modest budget for addressing this problem.

We optimised for 3 conservation objectives in the year 2100: wetland area (ha), carbon sequestration (Mg CO_2 yr^{-1}), and nursery habitat (ha). Each of the 1225 planning units had 804 estimates of each of these three objectives in 2100, arising from the SLAMM scenarios. The values for each objective were standardised (Supplementary Information) to facilitate calculation of a single covariance matrix and simplify the selection of weights. Four separate targeting strategies were developed, including three single-objective problems where weights for the other two objectives are zero (wetlands only, carbon sequestration only, and nursery habitat only) and a problem in which all three objectives were equally weighted. In addition, relative weights among objectives were varied in order to determine the relationships among each of the three pairs of objectives. λ was iteratively increased to represent increasing risk aversion of decision-makers.

Specific connectivity requirements for coastal wetlands under sea level rise were also incorporated. In reserving a parcel, the connectivity constraint ensured that neighbouring seaward parcels were also preserved, to allow for the process of wetland migration. Specifically, M_i was used to define the set of neighbours adjacent to property *i* that had wetlands present in a previous year (based on mean year of first occurrence from the SLAMM modelling). The parameter *m* was specified $0.5*|M_i|$(half of the number of neighbours of planning unit *i*). This meant that planning unit *i* could be selected only if *at least half* of the neighbours are selected. $0.5*|M_i|$ was chosen to strike a balance between connectivity and flexibility in reserve selection, but *m* could be any value from 1 to $|M_i|$.

For comparison, we also developed conservation plans for each of the 4 primary targeting strategies based on the means, of each of the IPCC RCP projection of sea level rise (i.e., RCP 2.6, 4.5, 6.0 and 8.5), rather than the distributions. These scenarios were also based on the means for all other parameters in SLAMM given in Table S1. Here we sought to maximise the conservation objectives without consideration of risk (λ was set to 0). All data organisation and pre- and post-optimisation processing was performed in R (R Core Team 2012), while the optimisation was directly solved as an integer quadratic problem in the software Gurobi (Gurobi Optimization Inc. 2014) within a guaranteed 5% gap of optimality.

4.4 RESULTS

4.4.1 Wetland and ecosystem service change

We found that there was a notable change in the distribution of wetlands in 2100 under sea level rise, with mangroves migrating landward, replacing salt marsh, *Melaleuca* and dryland areas (Figure 4.1b and c, Figure D.1). However, there was also considerable uncertainty surrounding these future distributions (Figure D.1). Spatially, the highest uncertainties occurred at the lowest and highest elevations of the future wetland distribution due to potential losses (continual inundation) and gains (landward movement) in the coastal wetland extent (Figure 4.1d). This variation in the future extent and type of coastal wetlands also affected the ecosystem services that flow from these wetlands which exhibited even greater variation (Figure 4.2). Greater variation is to be expected as the calculation of carbon sequestration and nursery habitat required additional models (and propagation of uncertainty) based on the wetland distributions. Whilst the impact of climate change on ecosystem services is generally negative, these impacts can be variable, particularly for carbon sequestration (Runting *et al* 2017a). However, the local scale of this study means that broader trends are not captured, such as the poleward expansion of mangroves and the services they provide (Saintilan *et al* 2014).

4.4.2 Risk-return trade-offs

We found that reductions in the risk of the final solutions were possible, but this came at the expense of reduced returns (Figure 4.3 and 4.4). Reducing risk also changed the spatial configuration of the reserve network (Figure 4.3). Selecting combinations of properties that are negatively correlated or un-correlated to reduce risk drove these changes, and often resulted in more expensive properties being purchased at the expense of a larger area. While targeting all objectives simultaneously is ideal, targeting any of the objectives (wetlands, carbon or nursery) individually still achieved solutions that were relatively close to combined multi-objective solutions (Figure 4.3a). This is expected, given that the initial expected value of wetland area and carbon sequestration in 2100 are highly correlated (R^2 = 0.95). However, optimizing only for nursery habitat was further from the combined multi-objective solutions, as the locations that provided nursery habitat were more constrained (i.e., along the land-ocean interface) than the other two objectives. Importantly, the variation in returns resulting from risk aversion far exceed difference in returns resulting from alternate weighting of objectives. The optimizations based on deterministic modelling of sea level rise produced the high returns, but were also relatively high risk strategies, irrespective of which RCP scenario informed the optimization (Figure 4.3 and Figure 4.4).

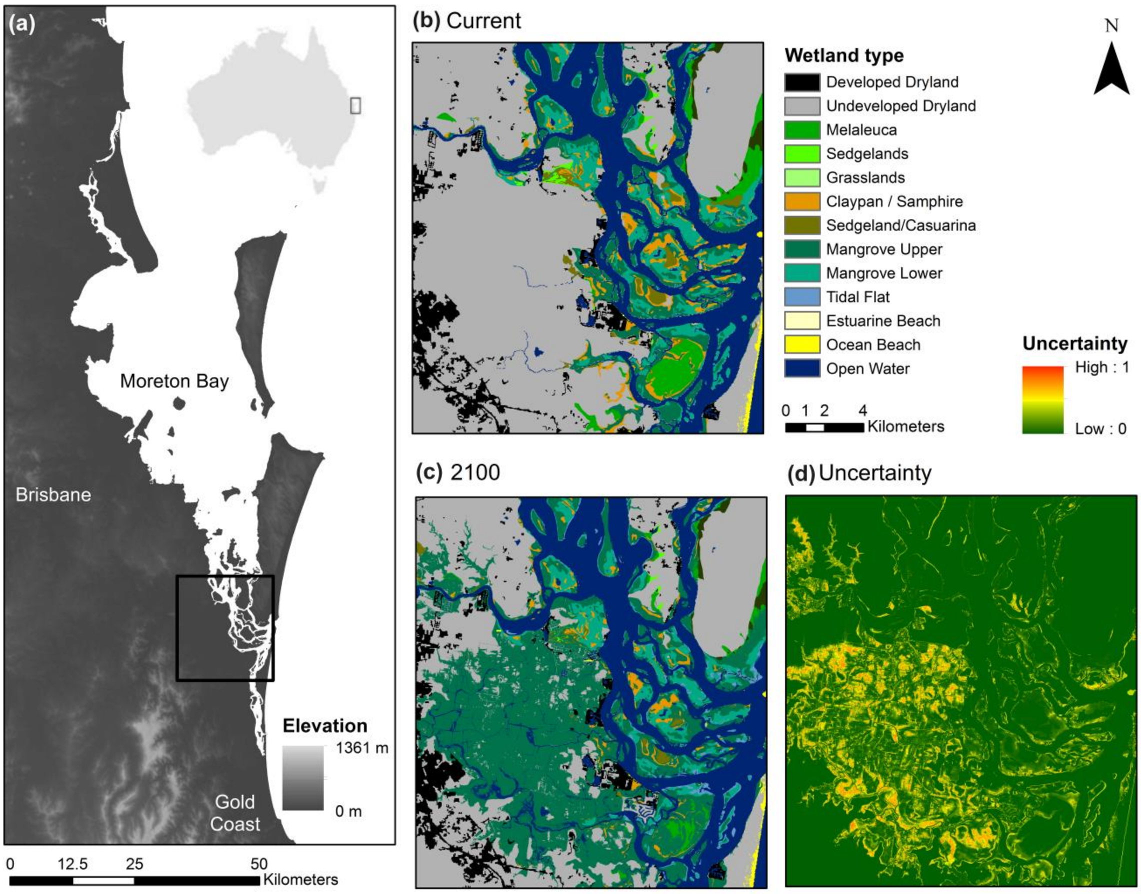


Figure 4.1 | Coastal wetland change under sea level rise for Moreton Bay, Queensland, Australia. Panel (a) shows the location of the study site from 153°14’49”E - 153°26’36”E to 27°38’59”S - 27°50’15”S. Panel (b) shows the current distribution of wetlands, and (c) shows the average (mode) wetland type projected to occur in 2100. The uncertainty in allocating each pixel to dryland, wetlands (any type), or water, is shown in panel (d) and described Appendix D.

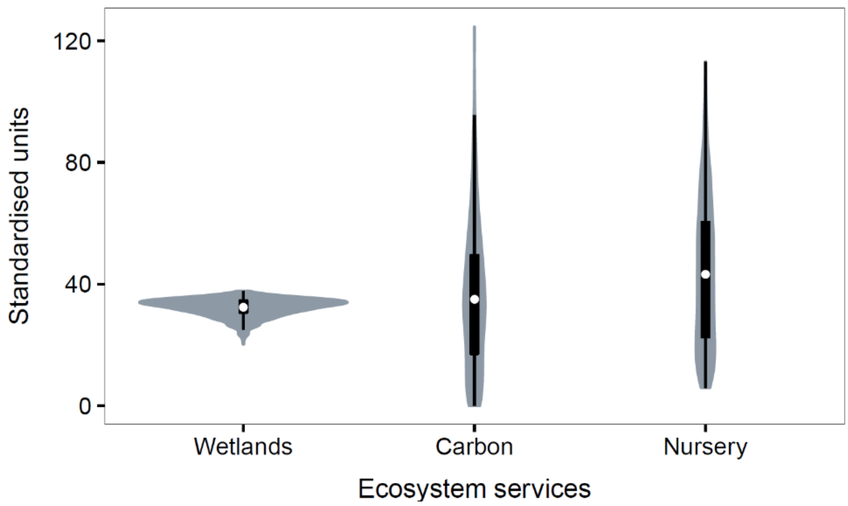


Figure 4.2 | The variation in the total amount of ecosystem services provided by the study site in 2100. The units for each ecosystem service were standardised by the range of the expected (mean) returns over the 804 scenarios. White circles indicate the mean, the black rectangle indicates the interquartile range, and the black line represents the range less outliers. The grey shading shows the distribution of values.

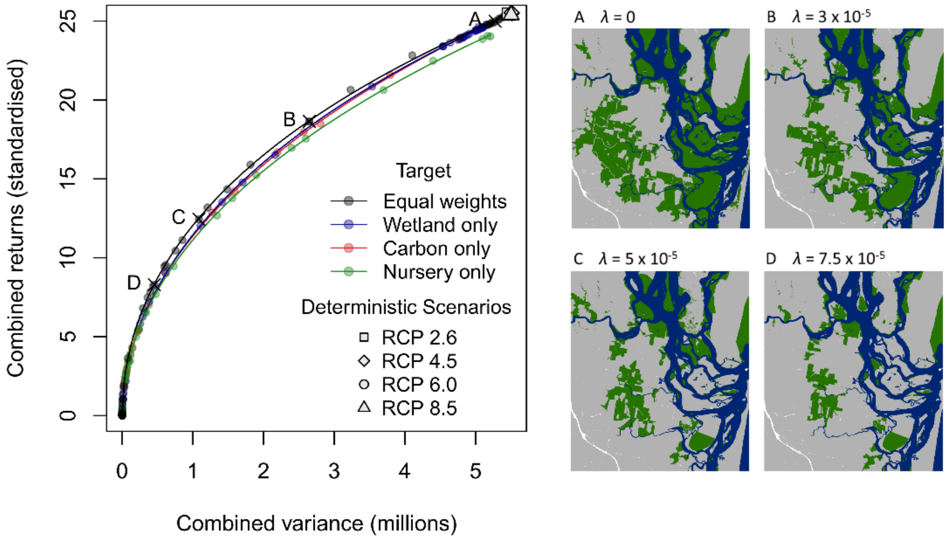
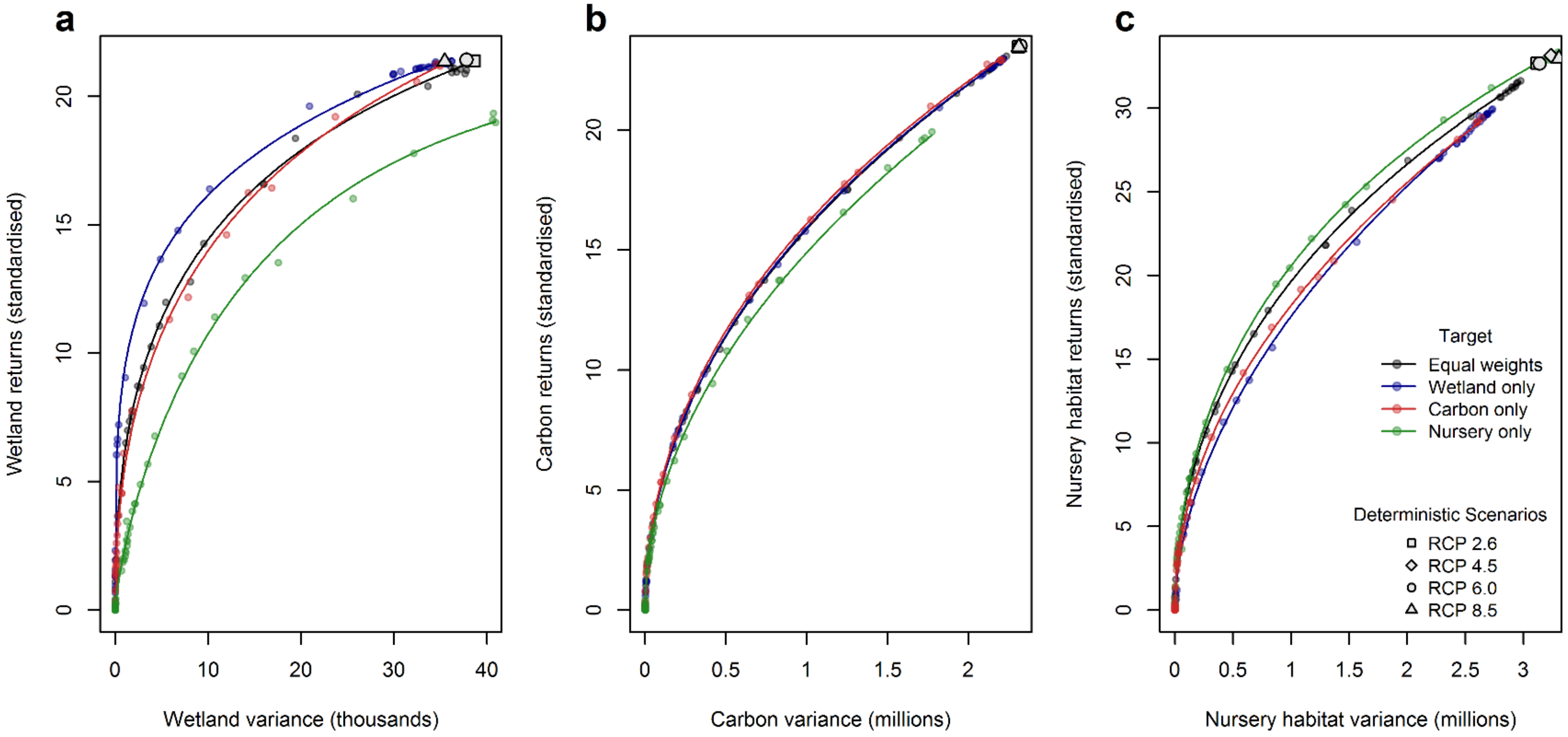


Figure 4.3 | Risk-return trade-off curves (or pareto frontiers) under different conservation targeting strategies (targeting wetlands only, carbon only, nursery habitat only, or an equally weighted combination of standardized values). Each point represents a potential reserve network, and moving left along a curve indicated the solution was optimized with increasingly risk aversion (λ). The curves approach, but do not reach, zero variance. The spatial distribution for 4 points along the curve are illustrated, with green representing selected properties, blue repressing (current) water, and grey showing unselected properties. All points along the pareto frontier are equally efficient, and the desired reserve configuration would depend on the risk preference of the decision maker. The risk and expected return of the scenario-based approaches targeting wetlands are also shown.

4.4.3 Relationships among services

We found that even though our three conservation objectives were largely synergistically provided in the landscape, there was still some divergence among objectives. Whilst carbon sequestration and wetland area exhibited negligible trade-offs at all levels of risk (Figure 4.5a), optimizing for nursery habitat area somewhat competes with both wetland area (Figure 4.5b) and carbon sequestration (Figure 4.5c). In both of these cases, intermediate levels of risk produced the greatest divergence among these conservation objectives (i.e., the centre of the graphs in Figure 4.5b and c). Reducing risk restricted the optimal combinations of planning units, narrowing the trade-off space, whilst increasing risk forced solutions towards the cheapest planning units with the highest expected returns, resulting in more similar combinations of planning units under different weighted combinations (Figure 4.5b and c).


a
b
c
Wetland returns (standardised)
Wetland variance (thousands)
Carbon returns (standardised)
Carbon variance (millions)
Nursery habitat returns (standardised)
Nursery habitat variance (millions)
Target
Equal weights
Wetland only
Carbon only
Nursery only
Deterministic Scenarios
RCP 2.6
RCP 4.5
RCP 6.0
RCP 8.5

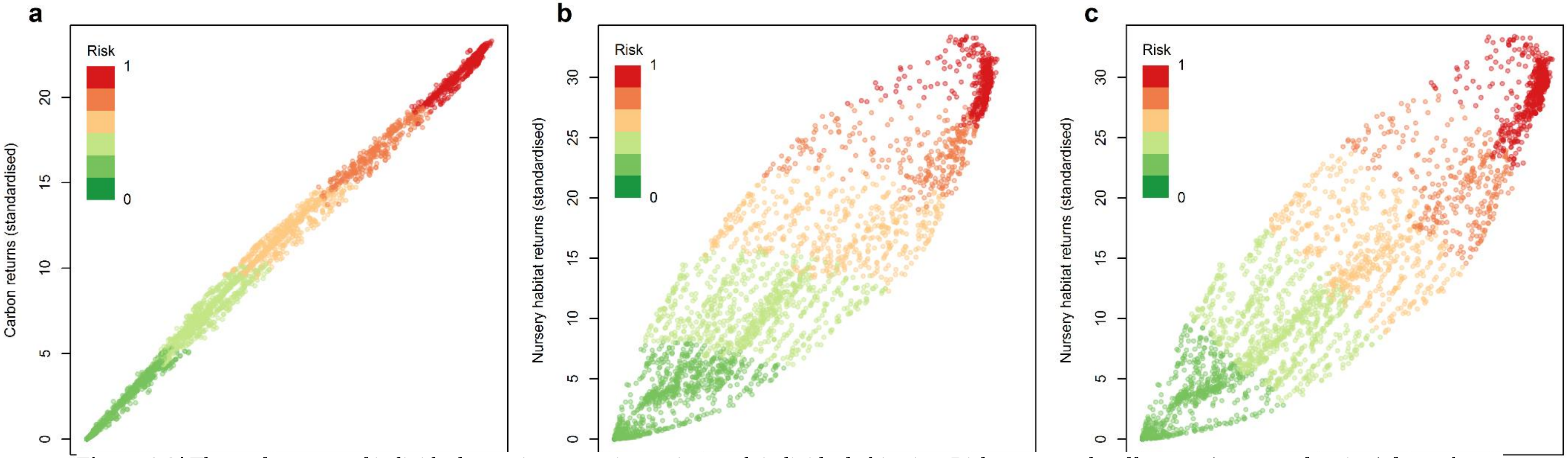


Figure 4.4 | The performance of individual targeting strategies against each individual objective. Risk-return trade-off curves (or pareto frontiers) for each targeting strategies (targeting wetland area only, carbon only, nursery habitat only, or an equally weighted combination of standardized values), against each individual objective: (a) wetlands, (b) carbon, and (c) nursery habitat value. Each point represents a potential reserve network, and moving left along a curve indicated the solution was optimized with increasing risk aversion (λ). The curves approach, but do not reach, zero variance. The risk and expected return of

Figure 4.5 | Relationships among ecosystem services when optimized for increasing risk preferences and varying weights among pairwise objectives. Each point represents the outcome of an optimization. Relationships are shown between (a) carbon sequestration and wetland area, (b) nursery habitat and wetland area, and (c) nursery habitat and carbon sequestration. Risk is calculated as a relative measure for each panel, based on the range of the standard deviation across all solutions.

4.5 DISCUSSION

Developing conservation plans that are resilient to uncertain patterns of ecosystem change and incorporate ecosystem services required an innovative planning approach. Here, we adapted Modern Portfolio Theory (Markowitz 1952) to a reserve design problem (Ball *et al* 2009, Beyer *et al* 2016). Rather than allocating a fraction of the project budget to spend in each planning unit (Ando and Mallory 2012a), we framed the problem such that each planning unit was either selected or not. We also incorporated connectivity requirements among planning units to ensure that important functional connectivity between planning units was maintained, and included multiple conservation objectives. This novel problem formulation allowed the selection of a complementary set of connected planning units that maximise a set of conservation objectives whilst hedging risk under climate change uncertainty.

For our case study application, we found that whilst planning based on only the most severe climate change scenarios (i.e. the highest rate of sea level rise) might appear to be a risk-averse strategy, the overall risk was still high compared to risk-averse optimization (Figures 4.3 and 4.4). This is because planning based on a deterministic scenario does not account for the covariance of benefits among planning units, and is therefore unable to select a complementary set of sites to minimise risk. Planning for the worst case climate scenario may reduce risk in some climate adaptation contexts (particularly engineering applications (Stewart and Deng 2015)), but this does not apply in conservation contexts where the impacts of climate change on species, ecosystems, and their services vary spatially over different climate change scenarios.

The key uncertainties we incorporated into our models and optimization were based on the best available information for our study region. However, it is important to note that all results are based on modelled future outcomes, rather than reality. Modelling natural capital and ecosystem services well into the future means there is no empirical ‘reality’ against which to compare our results (as no model is perfect (Dickey-Collas *et al* 2014)). In this case, uncertainty was incorporated in a coastal impact model (SLAMM (Clough and Propato 2012)) via a Monte Carlo simulation approach that included a probability distribution for all input parameters. The combination of this recent functionality in SLAMM and our novel problem formulation could be of major benefit to coastal conservation planning in our region and elsewhere. Yet the characterisation of these probability distributions is inexact and they may change as more information becomes available. Reductions in the uncertainty for key parameters, such as future rates of sea level rise, would be useful for projecting future wetland distributions and planning for them (Chu-Agor *et al* 2011, Runting *et al* 2013). Ideally, future work would also incorporate the uncertainty inherent in land acquisitions costs and owners' willingness to sell.

However, the absence of perfect information does not justify delaying the formulation and implementation of climate adaptation plans (Grantham *et al* 2009), particularly when known uncertainties have been accounted for when formulating the plan. Importantly, we note that our approach does not include unknown uncertainties, which may have catastrophic impacts (Makridakis and Taleb 2009), such as the impacts of severe storms or droughts which can influence the distribution of coastal wetlands (Gilman *et al* 2008). Info-gap decision theory attempts to deal with this issue (Moilanen *et al* 2006, Kujala *et al* 2013), however even this method has been criticised for starting from a best estimate and not considering all possibilities (Sniedovich 2007). Methods to effectively incorporate unknown uncertainties in a spatially explicit manner require further development.

We employed a mean-variance approach to account for the uncertainty in sea level rise projections and other model inputs. However, the mean-variance approach may be insensitive to highly skewed distributions and may not adequately reflect the risk preference of a decision maker in the cases where they wish to avoid returns below a specific benchmark (Ando and Mallory 2012b, Dunkel and Weber 2012). Accordingly, Shah and Ando (2015) developed an approach to optimize conservation investment among regions where the decision maker is particularly averse to returns below the amount given by the current climate in each region. However, choosing this threshold (or any other threshold) for downside risk aversion is dependent on the context of the analysis and preferences of the decision maker, and may not be appropriate in some cases. To illustrate, in our case we have many (landward) planning units that do not currently contain any wetlands, but are projected to gain wetlands in future climate scenarios (Figures 4.1b,c, and d). Here, setting a threshold for returns based on the current conditions would mean that these landward planning units would only exhibit "upside" risk, and would therefore be favoured over other planning units with similar mean returns, but with largely downside risk (such as those planning units at low elevations that currently contain wetlands, but are projected to lose some area with sea level rise). Whilst accounting for downside risk in this way would not be ideal for our case study, the potential of a downside risk approach should be considered when applying our method in other contexts.

It has been argued that the issue of "complete markets" has limitations for the spatial application of Modern Portfolio Theory (Mallory and Ando 2014, Shah *et al* 2016). Specifically Mallory and

Ando (2014) reason that in order to avoid producing a complete market (i.e., where any level of return can be guaranteed, thus unrealistically removing all uncertainty) the number of scenarios (*N*) must always exceed the number of planning units (or assets), such that there can never be more than *N-1* planning units (Mallory and Ando 2014, Shah *et al* 2016). However, our approach has several characteristics which enable us to avoid this limitation. Firstly, as the concept of a complete market was developed for financial markets, it assumes that the short-selling of securities is permitted (Arrow and Debreu 1954), which is not the case in land markets. Second, our problem formulation has a binary constraint on the selection of any planning unit, a fixed budget (such that all planning units cannot be selected), and a strict directional connectivity constraint. These types of strong constraints that are common in many conservation planning problems eliminate the possibility of a complete market in most cases. However, this should always be checked. If the standard deviation of the returns are greater than zero, then a complete market has not been achieved, and this is the case in our case study. Although we generated hundreds of scenarios for this analysis (804 potential distributions of wetlands under sea level rise), it is important to emphasise the characteristics of our problem formulation mean it could potentially be applied with far fewer scenarios (or more planning units) and not result in a complete market.

Here we focused on the supply side of ecosystem services, however ideally we would also incorporate the non-linear flows of these services to beneficiaries. Whilst the assumption of a linear accrual of benefits is reasonable for carbon sequestration, other coastal ecosystem services such as storm protection may face diminishing benefits as the area protected increases (Barbier *et al* 2008). Additionally, the spatial configuration of the ecosystem service supply can affect the flow of the service to beneficiaries, potentially leading to non-linear effects (Mitchell *et al* 2015). Including non-linear benefits can sometimes be achieved in a linear programming framework with piecewise linear approximations of non-linear functions, such as functions representing diminishing rates of return. In more complex circumstances, such as those in which there are feedbacks among objectives or time lags in responses, an approach that accounts for dynamics may be necessary (Golovin *et al* 2011). Furthermore, although dynamic problems are difficult to solve directly in a linear programming framework, dynamics can be approximated by solving a problem in increments of time, and updating state variables (e.g. the values of planning units for each objective) each time based on models of dynamics. For example, this approach was adopted by Bryan et al. (2016a, 2016b) when evaluating the supply of carbon and biodiversity services from agricultural lands under land use and climate change in Australia. Alternatively, approaches to decision making that are adaptive and participatory have the potential to find solutions to such ‘wicked’ problems that cannot yet be adequately modelled (Davies *et al* 2015, Head 2016).

4.5.1 Conclusions

The guiding principles for conservation planning under climate change include expanding reserve networks to accommodate future impacts, increasing connectivity, and ensuring a diversity of sites are included to ensure resilience (Lawler 2009, McLeod *et al* 2009). Here we have developed a novel problem formulation that adapts Modern Portfolio Theory to a conservation planning problem to simultaneously incorporate these principles for multiple conservation objectives. This approach addresses risks arising from climate change and uncertainties in modelling parameters, but these are not necessarily the only potential applications. Other threats to ecosystems and their services, such as fire (Westerling *et al* 2011) or land-use change (Metzger *et al* 2006) can have spatially variable impacts across scenarios and could benefit from the explicit consideration of risk. Additionally, this approach is not restricted to designing reserve networks, and could similarly be used to design plans for other conservation actions, such as restoration or the control of invasive species. Although reducing the risk of any conservation plan will inevitably trade-off with its expected returns, accounting for risk can improve the resilience of the solution through diversification and help ensure the continued supply of ecosystem services into the future.

5 Managing livestock production and greenhouse gas regulation under global change in northern Australia

5.1 ABSTRACT

Livestock grazing provides vital food supplies, but concerns have been raised of the industry's contribution to climate change, primarily through the emission of methane from cattle. Extensively grazed cattle generally have a relatively high methane output per animal due to poor quality pasture and limited options for intensification, but significant potential for emissions reductions exists. At the same time, the capacity of tropical savanna to maintain livestock production is likely to be impacted by climate change, primarily through the impact of changes in temperature, rainfall, and fire regimes on pasture. In addition, external economic drivers, such as changing livestock and carbon prices, could affect the viability of these production systems and abatement actions. The combined impact of climate change and global economic drivers has not previously been considered for livestock production and greenhouse gas regulation in tropical savannas. We used an integrated modelling approach to assess the impact of climate change, fire, and global economic drivers on the profitability and effectiveness of the livestock management action of safe stocking rates and the greenhouse gas emissions abatement actions of controlled burning and nitrate supplementation in northern Australia's rangelands. We found that the profitability of livestock production increased with growing demand, but rising farm input prices and new biophysical constraints posed by climate change counteracted these gains in some cases, and reduced the number of animals produced. Innovative strategies, such as changing fire management practices or nitrate supplementation were able to reduce greenhouse gas emissions, but they came with financial costs. Higher carbon prices under some global change scenarios were able to compensate for the costs of controlled burning, but costs remained a barrier for nitrate supplementation, even with a carbon price. Much of the grazing lands in northern Australia and elsewhere are already marginal for livestock production, so the opportunity to diversify income streams may prove vital in a changing climate.

5.2 INTRODUCTION

How to conserve natural capital whist meeting growing human needs is a problem of utmost importance (Rockström *et al* 2009). Ecosystem functions support human activities; for example, functioning ecosystems are vital in maintaining a stable and habitable climate (Foley *et al* 2003). On the other hand, human activities are substantially altering natural systems across the globe (Steffen *et al* 2015). These impacts are generated by a range of interacting drivers, including accelerating land use change, climate change, and the exploitation of natural resources (Foley *et al* 2005, Liu *et al* 2007). These drivers, and the interactions among them, can have a significant impact on the effectiveness of actions to manage natural capital and ecosystem services (Poiani *et al* 2010, Liu *et al* 2015a).

Policies for the conservation of natural capital have traditionally focused on areas of high species richness and biomass, such as tropical rainforests (Myers *et al* 2000, Naidoo *et al* 2008). However, there is growing interest in the potential of rangelands to provide ecosystem services due to the very large extent of these biomes (Steinfeld *et al* 2006, Thornton 2010, Witt *et al* 2011, Holechek 2013). Savanna is the world's largest terrestrial biome, covering 15% of the land area (19.31 M km^2) (Asner *et al* 2004), containing 17% of the globe's terrestrial aboveground carbon stores (Liu *et al* 2015b), and also has the largest area of land under managed grazing (9.48 M km^2) (Asner *et al* 2004). Although livestock production provides a vital food source, concerns have been raised about its contribution to climate change, primarily through the emission of methane (CH_4) which has a global warming potential 25 times that of CO_2 (Lassey 2007, Gill *et al* 2010). Extensively grazed cattle generally have a relatively high methane output, due to poor quality pasture and limited options for intensification (Rolfe 2010). Livestock currently contributes 14.5% of anthropogenic greenhouse gas emissions (Gerber *et al* 2013), but significant potential for emissions reductions remain (Thornton and Herrero 2010).

The capacity of tropical savanna to maintain livestock production is likely to be impacted by climate change (Lohmann *et al* 2012). Climate change, particularly increasing temperatures and changing rainfall patterns, has been highlighted as a key issue for rangelands (Brown and Thorpe 2008). Whist it has been established that temperatures are likely to increase, the regional impacts of climate change on rainfall are still uncertain (IPCC 2013). Rainfall in tropical savannas is already highly seasonal, and while this seasonality is likely to remain, climate change may lead to wetter conditions, drier conditions, or more inter-annual variability in rainfall (IPCC 2013). This may have similarly uncertain influence on wildfire, potentially leading to more intense and more frequent fires in a worst-case scenario (Bowman *et al* 2009). The combined effects of changes in rainfall, temperature and fire will have implications for livestock production, primarily via their impacts on pasture production (McKeon *et al* 2009).

In extensive grazing systems, management actions to mitigate greenhouse gas production can include reducing stocking rates, nitrogen supplementation, and fire management, amongst others (O'Reagain *et al* 2014, Walton *et al* 2014). Stocking at, or just below, the carrying capacity of the property does not only have environmental benefits, but can also be profitable for the landholder in the long run (O'Reagain *et al* 2011). This is because higher stocking rates can cause environmental degradation over the dry season and low rainfall years, resulting in animals in a poor condition for their age, which receive a lower price (O'Reagain and Scanlan 2013). The longer amount of time to gain weight can also increase greenhouse gas emissions intensity per animal (due to more methane emitted over time) (Charmley *et al* 2008). As nitrogen is often the limiting nutrient in the diet of cattle in extensive grazing systems, cattle are generally provided urea licks in low input systems to address this inadequacy (Bowman and Sowell 1997). Replacing this urea supplementation with nitrate supplementation has the potential reduce enteric methane production without impacting liveweight gain, but this comes with a much higher economic cost (Callaghan *et al* 2014). Additionally, controlled burning can also help to mitigate climate change. Burning tropical savanna early in the dry season may prevent more intense wildfire late in the dry season, thereby reducing the amount of greenhouse gasses emitted from fire (Williams *et al* 1999). Whilst these management actions appear promising, their performance under a changing climate has not been evaluated.

Further adding to the uncertainty surrounding the viability of these management actions is the potential impact of changing global economic conditions. Changes in the price for beef cattle and the cost of farm inputs can alter the profitability of livestock production (Thornton 2010). The growing demand for beef is likely to place upward pressure on livestock sale prices (McAlpine *et al* 2009), yet the costs of production are also likely to increase (Hatfield-Dodds *et al* 2015). These changes may create opportunities for emissions reduction (if livestock production becomes less profitable), or alternatively intensify the trade-off (if livestock production intensifies to meet global demand). Increases in productivity, through the development and adoption of new technologies (such improved livestock breeds or herd management practices), may also increase profits from livestock production (Nossal *et al* 2008). Alternatively, adequately pricing carbon is likely to make emissions abatement actions more profitable, but how this would play out with other economic and climate drivers is unknown.

Whilst previous studies have looked at the relationship between livestock production and greenhouse gas sequestration (for example; Lusiana et al. (2012) and Blandford et al. (2014)), the effects of global climate and economic change have not been considered. The combined impact of climate change and global economic drivers has rarely been considered for ecosystem services in any system (but see Connor *et al.*, (2015) and Bryan *et al.*, (2016) for exceptions), and never for livestock production and greenhouse gas regulation in tropical savannas. Here we used an integrated modelling approach to assess sustainable management actions for two ecosystem services (livestock production and greenhouse gas regulation) in northern Australia's rangelands under different climatic and economic conditions. We focused on four management actions: safe stocking rates, nitrate supplementation, prescribed burning, and destocking, whilst also considering combinations of these actions where plausible. We explored how these management actions (and combinations) perform in terms of livestock production, greenhouse gas emissions, and profitability under different scenarios and combinations of climate change and global economic drivers.

5.3 METHODS

5.3.1 Study Area

Northern Australia has a largely semi-arid tropical climate and highly seasonal rainfall, with 94% falling between November and April, and a steep rainfall gradient towards the coastal regions (CSIRO 2009) (Figure 5.1c). The region features large tracts of savanna vegetation, covering ∼2 million km^2. Dryland beef production dominates land use in the region (Figure 5.1), occupying ~60% of the land area and producing ~80% of the nation's live exports (Grice *et al* 2013). Grazing properties tend to be large (up to ~300,000 ha) with generally low productivity because of the rainfall and soil conditions (O'Reagain and Scanlan 2013). These soils are typically old, weathered, and nutrient poor, producing relatively sparse pasture (O'Reagain and Scanlan 2013). The forage base for cattle enterprises is predominantly unimproved native pasture, with very limited areas of exotic pastures or legumes (Brennan McKellar *et al* 2013). Grazing these lower-quality tropical (C4) pastures produces a relatively high amount of methane compared to other pastures (Callaghan *et al* 2014). Management strategies must be relatively low cost and easy to implement, which excludes more intensive management systems (e.g., cell grazing). Climate change is likely to bring higher temperatures and potentially more variable rainfall, making sustainable land management in northern Australia even more challenging (McKeon *et al* 2009).

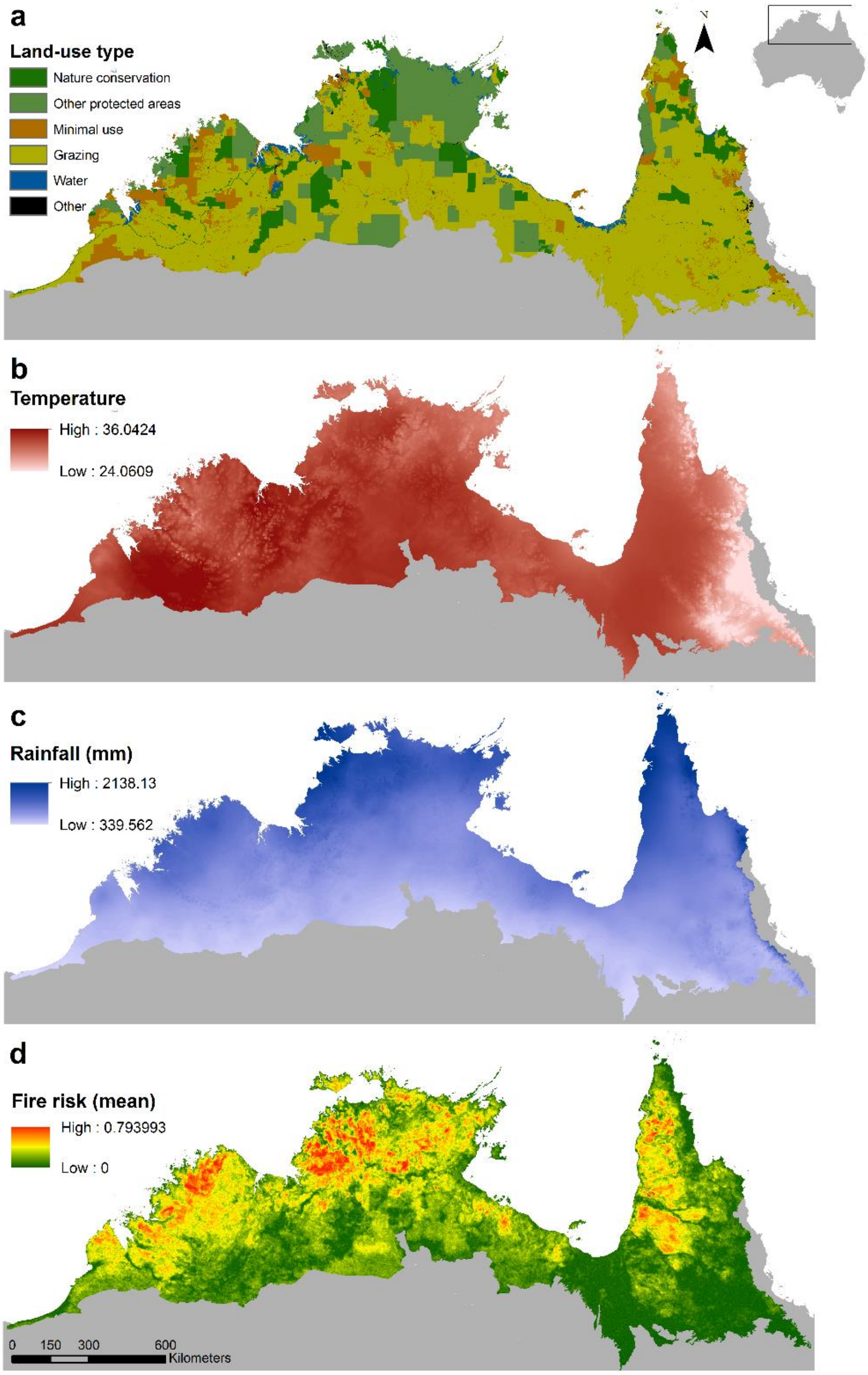


Figure 5.1 | The northern Australian study region. The area depicted was defined by the Interim Biogeographic Regionalisation for Australia (IBRA) (Australian Government 2012) at 0.01 decimal degrees (~1 km^2). Panel (a) shows the dominant land uses of the region from (ABARES 2016). Panels (b) and (c) show the average daily maximum temperature (^oC) and average annual rainfall (respectively) across 1987-2010 using data from Australian Government Bureau of Meteorology (Jeffrey *et al* 2001). Panel (d) shows the mean fire risk (proportion of vegetation burnt in a given year from 1988 – 2014) as described in Appendix E.

5.3.2 Global change scenarios

We used a combination of scenario analysis and sensitivity analysis to incorporate the uncertainty in global change and local management strategies from 2013-2050. The climate and economic scenarios were taken from the Australian National Outlook (Hatfield-Dodds *et al* 2015), which integrated Representative Concentration Pathways (RCP) from the IPCC (2013) (Table 5.1).

Table 5.1 | Key components of the global change scenarios used in this analysis (Hatfield-Dodds *et al* 2015).

Parameter	Units	Global Outlook			
		L1	M3	M2	H3
Representative Concentration Pathway		2.6	4.5	4.5	8.5
Temperature increase in 2100	°C	1.3 – 1.9	2.0 – 3.0	2.0 – 3.0	4.0 – 6.1
Population	billion people	8.1	10.6	9.3	10.6
Abatement effort		*Very strong*	*Strong*	*Moderate*	*None*
Cumulative emissions (2007 – 2050)	Gt CO_2^e	1437	2091	2091	2823
Emissions per capita	t CO_2^{-e} yr^{-1}	2.2	4.7	5.4	8.7
Size of the global economy (GDP)	US$ trillion	161.6	197.0	179.1	197.8
Carbon price	A$ tCO_2^{-1}	199.74	118.73	59.31	0
Livestock demand	% change 2007 – 2050	147	112	22	61
Oil price	% change 2007 – 2050	42	44	45	43

These scenarios are internally consistent (e.g., RCP 2.6 is not possible without strong greenhouse gas emissions abatement effort) and also provide projections of key economic parameters, including likely prices for livestock, oil, and carbon (Bryan *et al* 2016a). Projections of climate change parameters (e.g., temperature and rainfall change) were derived from 3 different general circulation (climate) models (GCM's) to encompass the range of climate outcomes (Hatfield-Dodds *et al* 2016). Specifically, the GCM's used were: the Canadian Earth System Model (CanESM) (Chylek *et al* 2011); Max Planck Institute – Earth System Model – Low Resolution (MPI-ESM-LR) (Giorgetta *et al* 2013); and the Model for Interdisciplinary Research on Climate version 5 (MIROC5) (Watanabe *et al* 2010). We also incorporated the variation in modelling parameters relevant to northern Australia. Three simple rates of increase of 0%, 0.57%, 0.114% p.a. in the total factor productivity of northern Australian beef cattle were applied, spanning the range of increases seen in the north Australian region between 1977/1978–2006/07 (Nossal *et al* 2008). Variation in other modelling parameters were also included as described below.

5.3.3 Simulation of sustainable management

Simulation modelling offers a useful approach to assess the impact of climate change, allowing the integration of economic and biophysical models (Campbell *et al* 2006, Tietjen and Jeltsch 2007). We focused on four actions (and combination of actions) in our simulation that are particularly relevant to the sustainable management of northern Australia's rangelands: (i) stocking at 'safe' levels, (ii) nitrate supplementation to reduce methane emissions, (iii) early dry season burning of savanna areas, and (iv) destocking. Where appropriate, we combined the different management actions to create combinations of actions were they were feasible (Table 5.2). These land management strategies affect the amount of greenhouse gasses emitted, and amount of food (beef) produced, and the economic returns to land. Climatic changes (i.e., changes in temperature, precipitation and fire) and external economic drivers (i.e., productivity growth, the costs of farm inputs, livestock price and carbon price projections) were also incorporated in the modelling framework, as they can impact the relative provision of greenhouse gas regulation and livestock production services and the potential of management actions (Figure 5.2). This allowed a comparison of livestock production, profit, and greenhouse gas emissions for each management strategy under global change.

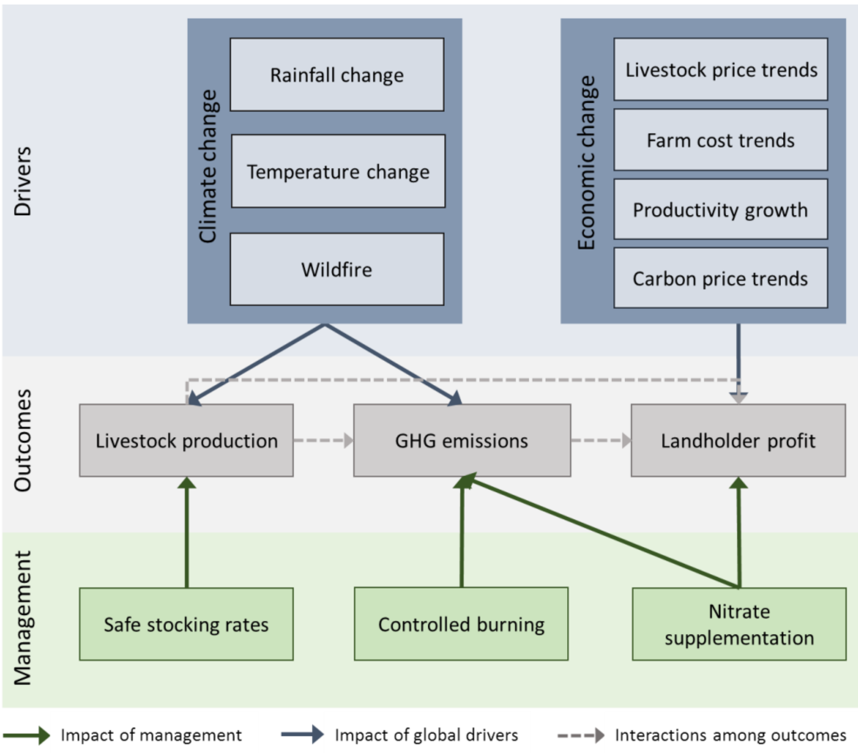


Figure 5.2 | A simplified conceptual model of the integrated assessment of sustainable management for livestock production and greenhouse gas (GHG) regulation under global change in northern Australia.

Table 5.2 | Different combinations of stocking, nitrate supplementation and controlled burning. Our assessment of the 'safe stocking', 'safe stocking + nitrate', and 'destocking' strategies include emissions from wildfire, as there was an absence of fire control.

	Stock	***Nitrate supplementation***	***Controlled burn***
Destocking	None	-	-
Destocking + Controlled burn	None	-	Yes
Safe stocking	Safe	-	-
Safe stocking + Nitrate	Safe	Yes	-
Safe stocking + Controlled burn	Safe	-	Yes
Safe stocking + Nitrate + Controlled burn	Safe	Yes	Yes

5.3.4 Fire modelling – controlled burning

Wildfire impacts greenhouse gas emissions through the combustion of vegetation, with hotter and more frequent fires generally having a greater impact (Hunt *et al* 2014). We calculated fire frequency and severity using recurrent-event regression analysis with shared frailty (i.e. for each cell in the study region) based on 27 years of burn scar data (1988 – 2014) and simulations based on Relative Difference Normalised Burn Ratio calculated from time-series satellite imagery (see Appendix E for details). The key output of from this modelling was the fire risk (occurrence and severity) in each pixel, which can be interpreted as the proportion of vegetation burned, for the historic baseline and the year 2050. High fire risk is characterised by warm temperatures, a lack of temperature seasonality, and high (but seasonal) rainfall, with much of the northern savanna having a high chance of experiencing fire (Figure 5.1d). This model found that climate change increased fire frequency and intensity, primarily through higher temperatures, although there was some variation across space and GCMs (Appendix E). Consequently, there was a slight reduction in fire risk across the area currently managed for grazing. To calculate the change in the proportion of vegetation burnt over time, we assumed a linear change in fire risk from the historic baseline to 2050. The central setting of the integrated simulation was based on the mean fire risk, with the 5th and 95th percentiles used as upper and lower bounds.

We calculated the greenhouse gas emissions from wildfire, and the emissions abated via controlled burning, using methods adapted from the official greenhouse gas accounting methodology of the Australian Government (Department of the Environment and Energy (DEE), 2015). Controlled burns are typically undertaken early in the dry season, with the aim of preventing the extent and severity of wildfires late in the dry season by reducing the fuel load (Russell-Smith *et al* 2013). The official methodology was designed to apply to the property scale, so modifications were necessary to be suitable for a broad scale assessment (akin to Heckbert et al. (2012) and Adams and Setterfield (2013)). Burnable fuel was calculated by reclassifying vegetation data from the National Vegetation Information System (NVIS 2016) and applying the corresponding value for burnable fuel given in Heckbert et al. (2012). The mass of fuel burnt (in Gg) in each year from 2013 – 2050 was calculated by:

$$M_i = BF_i \times FR_i \times (1 - ER) \quad (5.1)$$

Where M_i is the mass of fuel burnt in each cell, BF_i is the burnable fuel in each cell, FR_i is the simulated fire risk (occurrence and severity) for each cell, and *ER* is the reduction in fire risk from management (i.e. controlled burns). *ER* was set to either 0 (to represent no management), 0.34 (the most likely amount of emissions reduced by management (Russell-Smith *et al* 2009b, 2013)), 0.25 (a conservative estimate of management effectiveness (Heckbert *et al* 2010)), or 0.48 (the upper potential of management (Russell-Smith *et al* 2009a)). This equation was applied in every year from 2013 – 2050 as fire risk changed in each year.

Only methane and nitrous oxide emissions are accounted for in the official methodology, as it is assumed that any CO_2 released is eventually re-absorbed as the vegetation regrows (DEE, 2015). Therefore, to convert the mass of fuel burnt into greenhouse gas emissions, the following equations were applied:

$$EM_i = M_i \times CC \times EF_{CH_4} \times G_{CH_4} \quad (5.2)$$

$$EN_i = M_i \times CC \times EF_{N_sO} \times G_{N_sO} \times NC \quad (5.3)$$

$$GHG_i = MP_{CH_4} EM_i + MP_{N_2O} EN_i \quad (5.4)$$

Where EM_i and EN_i are the annual emissions of methane and nitrous oxide respectively for each cell *i*, *CC* is the carbon content of fuels (0.46 (DEE, 2015; Heckbert et al., 2012)), EF_{CH_4} and EF_{N_sO} are the emission factors for methane (0.00455) and nitrous oxide (0.00784) (DEE, 2015), G_{CH_4} and G_{N_sO} are the elemental to molecular mass fractions for methane (1.33) and nitrous oxide (1.57) (DEE, 2015; Heckbert et al., 2012), *NC* is the nitrogen to carbon ratio (0.00857) (DEE, 2015), MP_{CH_4} and MP_{N_sO} are the multipliers to convert methane (25) and nitrous oxide (298) to CO_2 equivalents (CO_2e) (DEE, 2016), and GHG_i is the Mg of CO_2e in each cell *i*. The cost of undertaking controlled burning was set at \$0.4685 ha^{-1}, based on data from Heckbert et al. (2012). This methodology allowed us to assess the greenhouse gas emissions from wildfire, and the potential emissions abatement from controlled burning in each year from 2013 to 2050.

5.3.5 Livestock production – safe stocking rates

Livestock production, in terms of 'safe' number of animal equivalents per year were modelled from a combination of pasture growth, safe pasture utilisation rates, and pasture intake per animal. We first built a statistical model of pasture growth based on rainfall and temperature for each of the 65 IBRA subregions in northern Australia (defined by the Interim Biogeographic Regionalisation for Australia (IBRA) (Australian Government 2012)). Data on past rainfall, temperature, and pasture growth was sourced from AussieGRASS (an Australia-wide implementation of the point-based GRASP (Grass Production) model (Carter *et al* 2000)). An ordinary least squares regression was used to predict pasture growth with annual rainfall and average maximum daily temperature as the explanatory variables using the 'ols' function from the 'statsmodels' module in the Python Programming Language (van Rossum and the Python Community 2012). The regression model was then used to project pasture growth from 2013 to 2050 under the 4 global change scenarios and 3 GCMs (Appendix E). A baseline of annual rainfall and maximum temperature was created by taking the mean from 1987 to 2010 from using data from Australian Government Bureau of Meteorology (Jeffrey *et al* 2001). We also created upper and lower bounds based on the 25th and 75th percentiles. These baselines were used to project the change in maximum temperature, rainfall, and subsequently pasture growth based on the projections for each global outlook and GCM.

However, the pasture available to livestock is also impacted by fire (McKeon *et al* 2009, Hunt *et al* 2014). The proportion of pasture burnt in a fire is generally greater than the proportion burnt of burnable fuel classes (i.e., as woody vegetation), as fine fuels are more flammable (Russell-Smith *et al* 2009b). To incorporate this effect, we assumed a certain percent of pasture in each cell was burnt (i.e., not available for cattle consumption in that year) based on the severity of the simulated fire. We classified the severity of the simulated fire (described above) into 3 classes of severity: low (≤0.33), moderate (0.33-0.66), and high (>0.66), and applied a percent of pasture burnt to each class based on values given in Russell-Smith et al. (2009) (low = 69%, moderate = 85%, high = 97%). The pasture available to livestock in a given year was reduced by these amounts in the cells where a fire was simulated to occur. For cells where no fire was simulated to occur, then 100% of the pasture was available for livestock. Whilst controlled burning could potentially reduce the amount of pasture burnt, to be eligible for emissions reduction funding, the number of livestock cannot be increased from the baseline (previous 10-15 years) (DEE, 2015). Therefore, we did not increase the pasture available to livestock as a result of controlled burning to ensure this condition was met.

We then calculated the number of livestock that could be supported by the amount of simulated pasture growth in each year without adversely impacting land condition (i.e., the ‘safe’ stocking rate (Scanlan *et al* 1994)). We assumed that the number of livestock could be varied from year to year in response to changing conditions. While this is a valid stocking strategy, there are constraints to its application in practice, as it can be challenging to rapidly increase of decrease stock numbers when managing a breeding herd in northern Australia (O’Reagain *et al* 2014). However, research results recommend applying flexible stocking rates to manage for climate variability (O’Reagain and Scanlan 2013). The safe stocking rate (adult animal equivalents per km^2) in each year was calculated using the following equation:

$$AE = \frac{P \times U}{C} \qquad (5.5)$$

Were AE is the number of animal equivalents (~450 kg), *P* is the annual amount of pasture growth (in kilograms), *U* is the safe pasture utilisation rate, and *C* is the amount of pasture consumed by an animal equivalent in a year (in kilograms). The safe pasture utilisation rate was set to 25% (and varied ±5% in the sensitivity analysis) for all pasture types in northern Australia based on data from (Hunt 2008b, Scanlan *et al* 2011, Walsh and Cowley 2011, O’Reagain and Scanlan 2013, Hunt *et al* 2014). The pasture consumption per animal equivalent was set at 9 kg per day (± 1 kg per day) based on a range of studies (Bernado, 1989; Holechek, 1988; Pieper, 1988; DAFF, 2013; Scanlan et al., 1994; Walsh and Cowley, 2011), and multiplied by 365 to give an annual value. We constrained the model to the broad area currently grazed by livestock (61% of the study area, Figure 5.1a) to avoid unsuitable vegetation types, soils, or topographies, and ensure appropriate land tenure.

We also calculated the potential profit from the simulated safe stocking rates. First, we created a baseline of the potential profit from safe stocking rates using recent (1997-2013) time series data for each Australian broadacre region in our study area (Navarro *et al* 2016). Time series data (including revenue, costs, cattle heads and herd structures) was compiled from ABARES Farm Survey data on specialist beef farms (ABARES 2015), and values with high relative standard error (> 0.9) were discarded. We calculated the mean (± the standard deviation) of revenue and costs per head of cattle for each region (Table 5.3), and converted these to a value per animal equivalent. Each region had a different typical herd structure, so the conversion to animal equivalents were specific to each region based on modelling using *Breedcow* software (Navarro et al., 2016; DAFF, 2013).

Table 5.3 | The baseline revenue, costs and greenhouse gas emissions per head from beef cattle for each broadacre region in northern Australia.

Broadacre Region*	**Price head^{-1}**	**Costs head^{-1}**	**Mg CO_2e head^{-1}**	**AE head^{-1}**
QLD: Cape York and the QLD Gulf	\$58.93 (±19.90)	\$26 .86 (± 8.69)	0.231 (± 0.061)	0.60
QLD: West and South West	\$152.92 (± 39.84)	\$68.61 (± 25.90)	0.274 (± 0.103)	0.68
QLD: Central North	\$116.98 (± 44.60)	\$49.11 (± 18.13)	0.258 (± 0.082)	0.74
WA: The Kimberly	\$81.70 (± 42.86)	\$35.25 (± 17.58)	0.214 (± 0.070)	0.63
NT: Barkly Tablelands	\$90.48 (± 38.42)	\$53.93 (± 26.71)	0.155 (± 0.040)	0.73
NT: Victoria River District - Katherine	\$82.79 (± 42.56)	\$40.21 (± 14.75)	0.171 (± .0.063)	0.66
NT: Top End Darwin and the Gulf of NT	\$107.12 (± 37.35)	\$63.63 (± 16.30)	0.163 (± 0.052)	0.64

*QLD = Queensland, WA = Western Australia, NT = Northern Territory. AE = Animal equivalents.

The economic outlook for livestock production could improve in the future due to technological innovation and an improvement in the price for livestock. To calculate the potential change in profit, the projected changes in livestock price for each global outlook (from Hatfield-Dodds et al. (2015)) were applied to the baseline revenues. We used the projected changes in oil price as a proxy for trends in the cost of farm inputs, and applied these to the baseline costs. We also increased yields by the total factor productivity (0.57%) in each year to 2050. This was calculated for each global outlook and GCM combination (with upper and lower extrema) using the equation:

$$PF_{iy} = AE_{iy}P_{iy}\Delta P_{y}TFP_{y} - AE_{iy}C_{iy}\Delta C_{y} \tag{5.6}$$

Where PF_{iy} is the profit (or loss) for cell i in year y, AE_{iy} is the number of animal equivalents simulated for cell i in year y, P_{iy} and C_{iy} represent the price and costs for an animal equivalent for cell i and year y respectively, ΔP_y and ΔC_y are the changes in livestock price and oil price, and TFP_y is the total factor productivity increase.

Livestock also produce greenhouse gas emissions, primarily from enteric fermentation (microbial action in the digestive system) (Cottle *et al* 2011). Greenhouse gas emissions per head were calculated in a similar way to profitability: the mean (± the standard deviation) biogenic greenhouse gas emissions per head of beef cattle were taken from times series data (1997-2013) for each Australian broadacre region (Navarro *et al* 2016), and converted to emissions per animal equivalents. These beef cattle biogenic emissions were calculated by applying the data on total head and herd structure into the Greenhouse Gas Accounting Framework (Eckard *et al* 2008, Navarro *et al* 2016). Whilst this analysis does not capture greenhouse gas emissions from farm operations, these additional sources are considered to be relatively minor in extensive grazing systems relative to biogenic emissions (Steinfeld and Wassenaar 2007).

5.3.6 Livestock production – nitrate supplementation

There is potential to reduce these biogenic emissions without impacting livestock production, but this comes at a higher financial cost (Grainger and Beauchemin 2011). Nitrogen is typically the limiting nutrient in extensive livestock systems, so cattle are typically provided with nutritional supplementation in the form of urea lick blocks to increase liveweight gain (Bowman and Sowell 1997). Replacing urea supplementation with nitrate supplementation has the potential reduce enteric methane production without impacting liveweight gain by reducing enteric methanogenesis (the formation of methane by microbes) (Nolan *et al* 2010). However, the nitrogen proved by calcium nitrate molasses blocks is lower than urea blocks, resulting in a higher number of blocks required (2.5 times) and a subsequently higher cost (+$0.17 per animal per day) (Callaghan *et al* 2014). Nitrate supplementation reduces methane emissions of 15 g per animal per day (Callaghan *et al* 2014), which we multiplied by 25 to convert to CO_2e (DEE, 2016). To model the impact of nitrate supplementation, we applied these emissions reductions to the greenhouse gas emissions simulated from the safe stocking strategy, and subtracted the additional cost from the profit per animal (equation 5.6).

5.3.7 Carbon price

We created an additional set of scenarios that captured the effect of the carbon prices associated with the global emissions abatement effort assumed within each global outlook. This meant that emissions abatement, in addition to stocking, could contribute to profits. The calculation of profit remained the same for ‘safe stocking’ (equation 5.6) as there was no emissions abatement. However the equations for other management actions changed. For nitrate supplementation the equation was:

$$NPF_{iy} = AE_{iy}\Delta P_y TFP_y P_{iy} - AE_{iy}\Delta C_y \left(C_{iy} + NC\right) + AE_{iy} CP_y ER \qquad (5.7)$$

Where NPF_{iy} is the profit from safe stocking with nitrate supplementation for cell i in year y, NC is the additional cost of nitrate supplementation compared to urea per animal, ER is the emissions reduction from nitrate supplementation per animal, and CP_y is the carbon price in year y. All other parameters are as per equation 5.6. The potential profit from destocking was calculated as:

$$DPF_{iy} = AE_{iy} CP_y E_i \qquad (5.8)$$

Where DPF_{iy} is the profit from destocking for cell i in year y, E_i is the biogenic greenhouse gas emissions per animal from safe stocking in cell i, and the remaining parameters are as above. The profit from controlled burning was calculated as:

$$BPF_{iy} = ER_{iy}CP_y - \Delta C_y BC \quad (5.9)$$

Where BPF_{iy} is the profit from controlled burning for cell i in year y, ER_{iy} is the emission reductions (in Mg of CO_2e) from controlled burning in cell i in year y, and BC is the cost of conducting a controlled burn. The change in oil price ΔC is also used here as a proxy for the trends in farm costs. Where multiple actions were undertaken simultaneously, these costs and emissions reductions were summed. Together, this allowed a comparison of greenhouse gas emissions and profits for each of the management combinations under a range of carbon prices.

5.4 RESULTS

Our integrated modelling approach revealed the profitability and effectiveness of different strategies to manage livestock production and greenhouse gas regulation under global change. Under destocking, emissions were entirely driven by wildfire with a baseline of 2.59 million Mg CO_2e yr^{-1} in total across the northern Australia (Figure 5.3a). Grazing livestock without any emissions abatement actions ('safe stocking') has the highest baseline emissions (3.54 million Mg CO_2e yr^{-1}) (Figure 5.3a). Supplementing livestock with nitrate instead of urea, and undertaking controlled early dry season burning could substantially reduce these emissions (by up to 1.28 million Mg CO_2e yr^{-1}). However, nitrate supplementation did not have as large a reduction as removing livestock altogether and managing fire ('controlled burning'), which had a reduction of 1.83 million Mg CO_2e yr^{-1} from 'safe stocking' (Figure 5.3a). In terms of livestock production, the most severe climatic change scenarios had the largest reduction in stocking rates, leaving fewer animal equivalents produced from the same land area in each year (and therefore lower total GHG emissions from livestock) (Figure 5.3b).

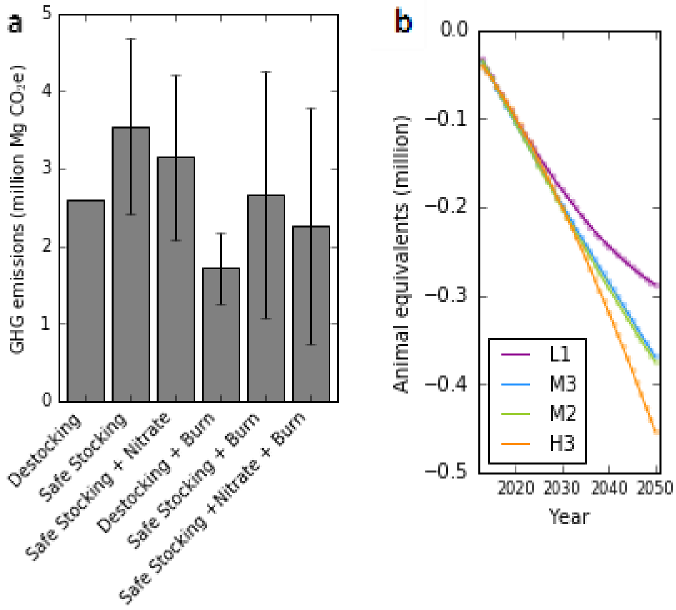


Figure 5.3 | Baseline GHG emissions and change in livestock production under global outlooks. Panel (a) shows the baseline GHG emissions for each management action (and combination of actions). The baselines were calculated as the mean from 1987 to 2010, including the mean historic baseline for fire risk, and the error bars show the upper and lower bounds over this period based on abatement potential from controlled burning, pasture utilisation rates, and consumption per head. The destocking action does not have error bars as the emissions from this action were from historic wildfire. Panel (b) shows the change in animal equivalents from the baseline safe stocking (of 2.9 million AE across the region) with each global outlook under all management actions that include safe stocking (i.e., 'safe stocking', 'safe stocking + nitrate', 'safe stocking + burn', 'safe stocking + nitrate + burn'). M3 and M2 are indistinguishable here as they are based on the same RCP (4.5).

In the case of both greenhouse gas emissions and livestock production, we found that there was substantial spatial variation in outcomes (Figure 5.4, column 1 and 2). Livestock production was generally higher in the east (in the state of Queensland), and particularly the south-east, due to better conditions for grazing (e.g. less extreme temperatures). However the declines in livestock production brought about by climate change were also focused in this area (Figure 5.4, column 1). Greenhouse gas emissions were higher in the north (Figure 5.4, column 2), and these were primarily due to unmanaged wildfire. The future change in greenhouse gas emissions saw a trend of emissions increasing in the north and declining in the south under all global outlooks (Figure 5.4, column 2). We also found substantial uncertainty in the magnitude of these results. Although the trends remained similar, the range of the upper and lower bounds for all outcomes was considerable (Figures E.18 and E.19). This variation arises from the projections of different GCMs, the extrema of fire risk (5^{th} and 95^{th} percentile), and varying the range of parameters to assess the management potential (i.e., pasture growth, utilisation rates, consumption, and emissions per head) and profitability (i.e., upper and lower bounds for revenue and costs).

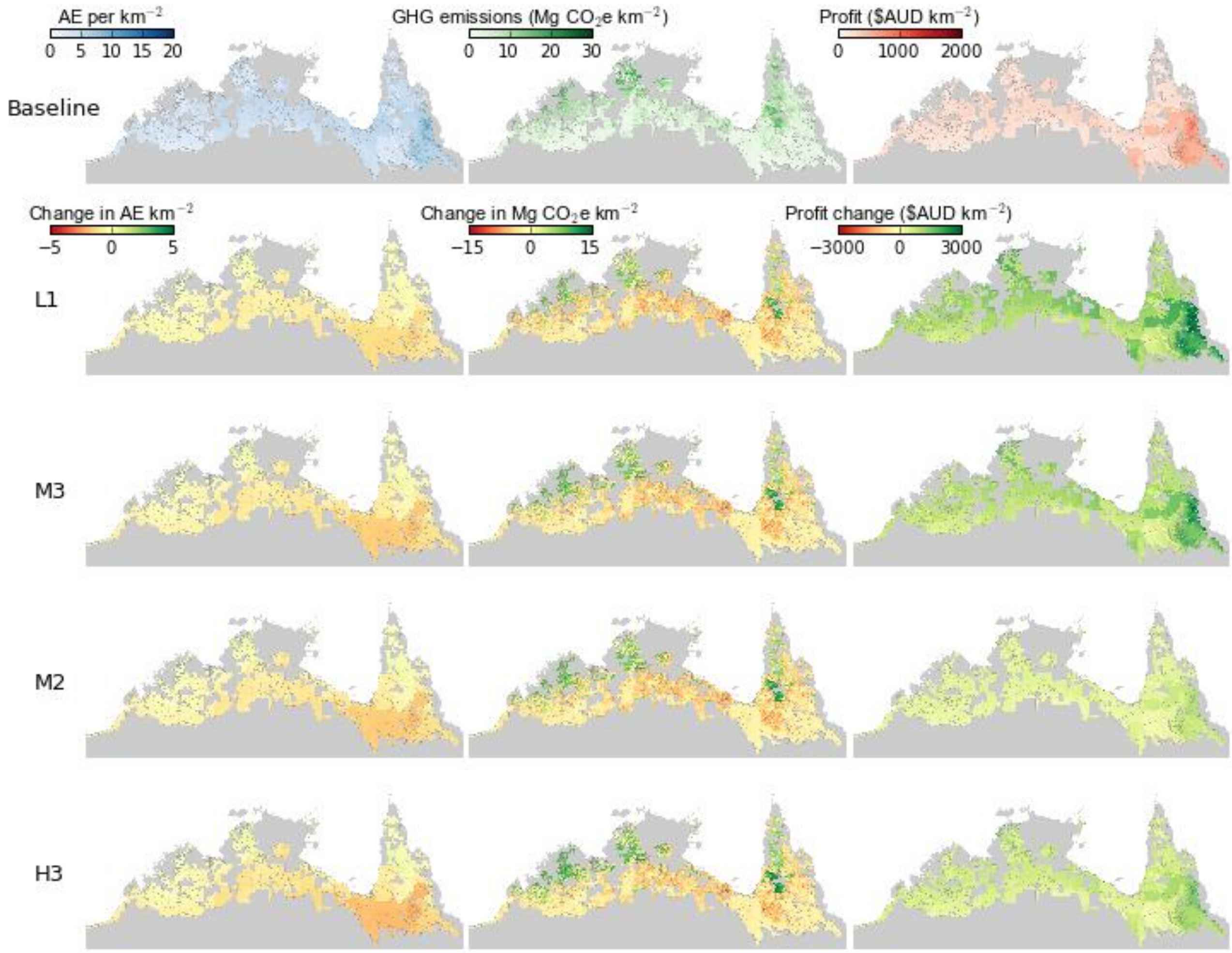


Figure 5.4 | Mean outcomes for safe stocking rates under global change scenarios to 2050. The baselines for livestock, greenhouse gas emissions, and profit are shown in the top row for the 'safe stocking' management action (safe stocking rates without controlled burning). The remaining rows show the mean change by 2050 in each outcome under the global outlooks. GHG emissions include emissions from both wildfire and livestock as there was no action to control fire in the 'safe stocking' management action. The upper and lower bounds for these baselines and changes over time is given in Supplementary Figures E.18 and E.19.

Despite the general decline in safe stocking rates with climate change, an increase in the profitability of safe stocking strategies occurred under most global change outlooks (Figure 5.4, column 3, and Figure 5.5). This increase is due to the increasing profit margins for livestock under most global outlooks (Table 5.1). Climate change alone (i.e., without concurrent changes in global prices) had a limited impact on the profitability of all strategies (Figure 5.5). The cost of supplementing livestock with nitrate was substantial, and these costs were not recovered even with relatively high carbon prices (i.e., 'nitrate' does not reach the level of 'safe stocking' in row 3, Figure 5.5). The economic outlook for controlled burning (only) and destocking (only) improved with carbon pricing (Figure 5.5). Therefore the most profitable management combination is combining controlled burning with safe stocking rates (i.e. adding columns 1 and 3 in Figure 5.5).

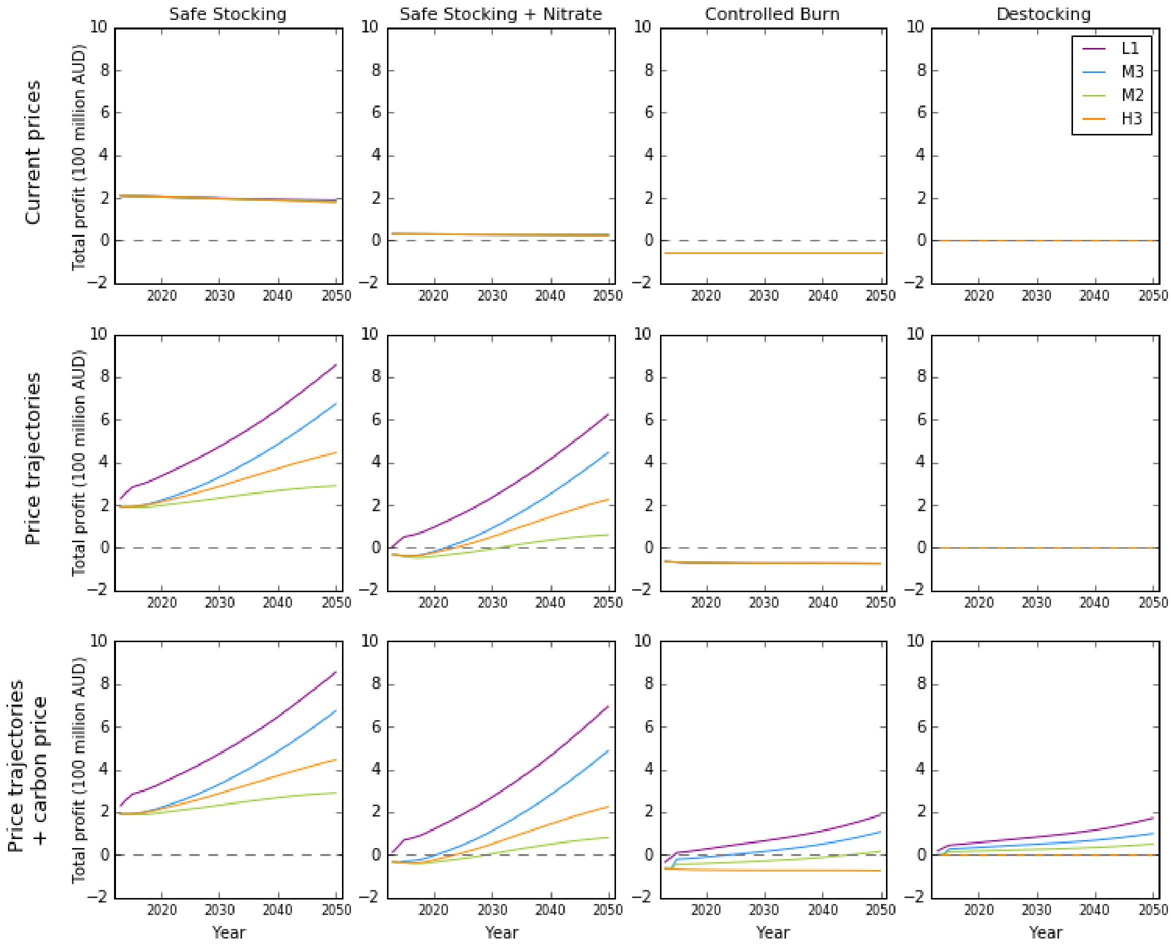


Figure 5.5 | The total profitability of management actions under global change across northern Australia. The first row, 'current prices', shows profitability over times resulting solely from changes in climate (i.e. no economic change). The second row, 'price trajectories', shows the changes due to climate change in conjunction with trajectories for livestock prices and the cost of farm inputs. The third row, 'price trajectories + carbon price', shows the total impact of each global outlook (i.e., the effect of climate change and all associated price trajectories, include carbon pricing).

The profitability of each management action was also spatially variable, so we mapped the most profitable strategy for each pixel and global outlook in 2030 and 2050 (Figure 5.6). Given the lack of carbon pricing in global outlook H3, none of the abatement actions could compete with the profitability of safe stocking (in any year). In the global outlooks that contained a price for carbon (L1, M3, and M2), controlled burning (combined with safe stocking) was generally more profitable in the north, but safe stocking (without any abatement) remained the most profitable in the south (Figure 5.6). None of the other abatement actions were profitable in any area under any global outlook. There was some variation in the specific areas that were profitable for controlled burning, driven by the spatial variability in abatement potential, along with interplay between the trends in carbon prices, livestock prices, and farm costs over time under different global outlooks.

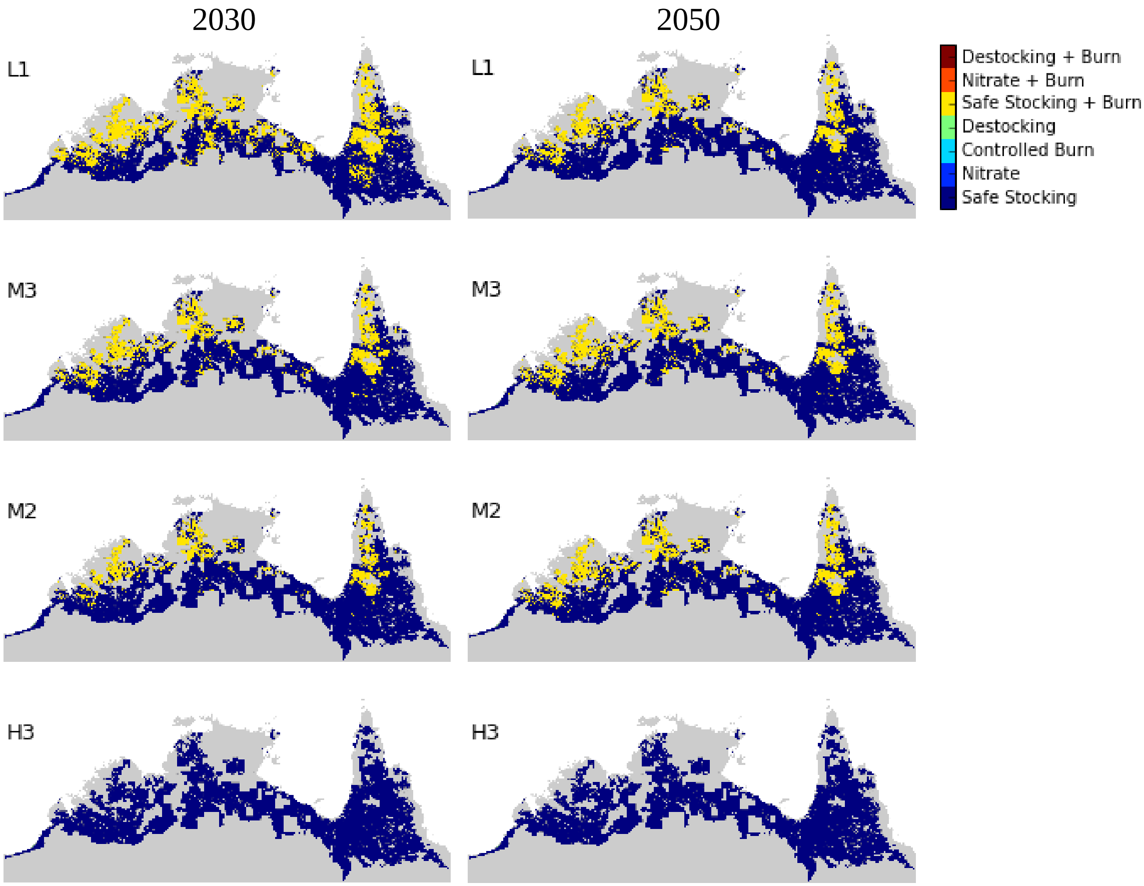


Figure 5.6 | The most profitable land management in 2030 and 2050 under global change with carbon price trajectories. Where all land management actions resulted in a loss, we considered no management (i.e., no cattle or fire management) to be the most profitable.

5.5 DISCUSSION

We employed an integrated systems modelling approach to account for the cumulative impacts of climate change, external economic drivers, and management actions on livestock production and greenhouse gas regulation. Climate change reduced the capacity of northern Australia to support livestock, with the number of cattle that could be safely stocked declining over time and under more severe projections of climate change (Figure 5.3b). This finding is supported by numerous other studies, with a review by McKeon et al. (2009) finding that safe stocking rates were strongly dependent on climate. Fewer cattle resulted in lower total greenhouse gas emissions from livestock, but these results varied spatially and in response to global drivers. These findings are consistent with a global review, which found that the impacts of climate change on food production were generally negative, and carbon sequestration had the most variable response to climate change of all ecosystem services (Runting *et al* 2017a).

We found these greenhouse gas emissions could be further reduced by supplementing the cattle with nitrates (to reduce enteric methane emissions). However, nitrate supplementation remained economically unprofitable, even with future trajectories for carbon payments. Replacing urea with nitrates is a relatively new option for reducing greenhouse gas emissions in northern Australia, and financial considerations were hypothesised to be one of the primary limiting factors for adoption (Callaghan *et al* 2014). In contrast, we found planned early dry season burning resulted in substantial emissions reductions, and became marginally economically profitable under global change scenarios that included a carbon price. This is in line with other studies that have found significant emissions abatement potential from managed fire across the region (Heckbert *et al* 2012, Adams and Setterfield 2013), and these emissions reductions (and profits) could be further increased if the maximum emissions reduction potential is achieved (Russell-Smith *et al* 2009a).

Our model was necessarily general to encompass the broad scale of Australia's northern rangelands, so some details and dynamics were omitted that may be relevant at the property scale. Our estimates of safe stocking numbers were primarily determined by pasture growth (Scanlan *et al* 1994). Whilst this relationship is broadly representative, other factors can also influence the safe stocking rate at finer scales, particularly slope, the species composition of pasture, and the spatial distribution of grazing pressure within a property, amongst others (Orr and O'Reagain 2011). Additionally, land holders do not have perfect information about future pasture growth, so stock number may be unintentionally set above the carrying capacity of the property in a given year (O'Reagain *et al* 2014). This can result in land degradation, which can in turn impact pasture growth and the 'safe' number of livestock in subsequent years (Greenwood and McKenzie 2001, Hunt *et al* 2014). Here we modelled the application of safe stocking rates and did not simulate feedbacks to pasture growth from overstocking, however this remains an important land management issue for rangelands.

Our results may inform future modelling of land use change in the region under different global change scenarios (akin to Bryan et al. (2016a)). However, to give more reliable projections of land use change, these results need to be combined with realistic models of human behaviour (Rounsevell *et al* 2014). Although actions to mitigate greenhouse gas emissions become more profitable under some global outlooks, this is unlikely to be sufficient to actually instigate a change in management practices on many properties. Such a change would also need to overcome a varying range of risk aversion and attitudes towards adopting new practices (Rolfe and Gregg 2015). For instance, Australia-wide research has categorized primary producers into four typologies ranging from early adopters ("The first primary producers to try new things") to recalcitrant ("They don't listen to others, are less capable of adaptation") (Donnelly *et al* 2009). Data from cattle graziers in northern Australia's rangelands found that 85% of sampled pastoralists had low interest in adapting to climate change and were not strategic in their management (Stokes *et al* 2012, Marshall and Stokes 2014, Marshall *et al* 2014). Accordingly, the potential increase in profitability of greenhouse gas emissions abatement actions is unlikely to directly translate into management change in most cases, so risk aversion and barriers to adoption should also be incorporated.

Our study has focused on food production (livestock) and climate regulation (greenhouse gas emissions), yet the management strategies would also have impacts on biodiversity and other ecosystem services. Although extensive livestock grazing has lower environmental impacts (per unit area) than other more intensive land use options (such as cropping), it is not without issue (Steinfeld *et al* 2006). For example, a study in northern Australian rangelands found that runoff significantly increased on hillslopes with small patches of bare ground, even where they had relatively high mean cover (Bartley *et al* 2006). As a consequence, livestock production could have implications for hydrological ecosystem services in the region, as grazing pressure tends to be heterogeneous (O'Reagain and Scanlan 2013). Multi-paddock cell grazing systems not justified in northern Australia due to very low densities of cattle, making it difficult to homogenise grazing pressure (O'Reagain *et al* 2014). Whilst stocking at 'safe' levels are likely to reduce these negative hydrological impacts, they cannot be eliminated entirely (Bartley *et al* 2010). Similarly, livestock grazing has largely negative impacts on biodiversity in northern Australia by altering ecological communities and in some cases bringing invasive species (Garnett *et al* 2010, Woinarski *et al* 2011). These impacts are somewhat lessened at low stocking rates and are significantly improved with destocking (Lunt *et al* 2007, Legge *et al* 2011). Ideally impacts of livestock grazing on biodiversity and other ecosystem services should also be considered. However, it may not be possible to achieve these multiple objectives through financial incentives alone, and a more strategic planning approach may be required (Morán-Ordóñez *et al* 2016). Alternatively, adaptive management and collaborative planning could be used to engage key stakeholders and develop novel solutions to this complex problem (Sayer *et al* 2013, DeFries and Nagendra 2017).

In contrast to livestock grazing, planned early dry season burning is likely to have mostly positive impacts on biodiversity (Woinarski and Legge 2013). Having a diversity of time-since-burnt in patches across the landscape (pyrodiversity) is hypothesised to be optimal for biodiversity to accommodate the different responses of various taxa to fire (Martin and Sapsis 1992, Griffiths *et al* 2015). Some taxa are fire dependent, or at least resilient to frequent fire (such as ants), whereas others depend on long unburnt areas for survival (i.e., many small mammal species) (Andersen *et al*

2012). Controlled, early dry season burning can potentially manipulate the fire mosaic for both carbon and biodiversity benefits by reducing the extent of more severe fire late in the dry season (Russell-Smith *et al* 2013). However, this may come at the expense of pastoral production (and some species) if woody thickening occurs (Walton *et al* 2014), so a more strategic design of prescribed fires may be needed to deliver biodiversity benefits (through long unburnt areas), relative to solely managing for carbon (Andersen *et al* 2005). Therefore, fire management *exclusively* for carbon benefits may not be appropriate in areas with important biodiversity values or on some pastoral properties. Decision theory has been used to manage fire for multiple objectives (biodiversity and built asset protection) at the wildland-urban interface (Driscoll *et al* 2010, Williams *et al* 2017), and this approach may also be of benefit to manage the multiple objectives in extensively grazed tropical savannas.

Although not considered in this study, the implications of management activities on employment and health cannot be overlooked, particularly for the indigenous people of the region. Indigenous lands cover large areas in northern Australia (ABARES 2016) and includes a diverse array of management activities, which vary according to land tenure, cultural sites, and funding availability (Hill *et al* 2013). Although some indigenous landholders undertake pastoral activities, further development (such as expanding grazing in indigenous owned land) may provide limited benefits to indigenous people, and they are more likely to be adversely affected by associated declines in natural capital (Stoeckl *et al* 2013). In contrast, fire management is in line with traditional indigenous uses, and can also provide employment opportunities, particularly with a carbon market (Walton *et al* 2014). However, payments for ecosystem services may conflict the world views of some indigenous people which can limit adoption (Zander *et al* 2013).

5.5.1 Conclusions

Integrating multiple climate and economic drivers is often overlooked in assessments of ecosystem services, which can create misleading results and limit their utility for decision making (Runting *et al* 2017a). Here we incorporated multiple drivers (i.e., temperature increase, rainfall change, fire, productivity growth, and price trajectories for livestock, farm inputs, and carbon) to assess the greenhouse gas emissions and livestock production to 2050. The profitability of livestock production increased with growing demand, but rising farm input prices and new biophysical constraints posed by climate change counteracted these gains in some cases. Innovative strategies, such as changing fire management practices or nitrate supplementation were able to reduce greenhouse gas emissions, but they came with financial costs. The growing urgency to abate emissions in some global change scenarios resulted in prices for carbon that were able to compensate for the costs of controlled burning, but costs remained a barrier for nitrate supplementation, even with a carbon price.

Although our modelling is based on Australia's northern rangelands, our findings are likely to be relevant to other rangelands facing similar climatic and economic fluctuations. The low input and low productivity cattle grazing systems in northern Australia are fairly typical of grazing enterprises throughout the globe's tropical savannas (Steinfeld and Wassenaar 2007), which all face a likely increase in temperatures and uncertain changes in rainfall with climate change (IPCC 2013). Rising livestock prices, driven by a growing demand for beef, is also a global phenomenon that influences markets beyond northern Australia (McAlpine *et al* 2009). Constraining climate change to the less severe scenarios will require strong global action, producing substantial incentives for emissions abatement (Hatfield-Dodds *et al* 2015). Much of the grazing lands in northern Australia and elsewhere are already marginal for livestock production, so the opportunity to diversify income streams may prove vital in a changing climate.

6 Conclusions

Climate change and other global change drivers are having a significant impact on ecosystem services and their underpinning natural capital, and these impacts are likely to intensify over time (Scholes 2016). Consequently, incorporating the impacts of global change into assessments and decisions concerning ecosystem services is vital to ensure the continued supply of these services (Mooney *et al* 2009, Polasky *et al* 2011). Additionally, given the finite nature of conservation resources, it is also imperative that any solution is cost-effective to ensure resources are not squandered (Duke *et al* 2013). Ignoring these complexities could result in misleading outcomes of both assessment and decisions concerning ecosystem services (Bryan 2013). However, to date there have been relatively few attempts to incorporate global drivers of change into ecosystem services assessments, and even fewer into decision making. To address this gap, the overarching aim of this thesis was to develop and assess approaches to manage natural capital assets and ecosystem services under global change. To achieve this I integrate methods from environmental management, operations research, and economics, to incorporate multiple drivers and objectives into the management of ecosystem services. Specifically, four separate objectives were addressed: (i) to determine how climate change and other drivers have been incorporated into ecosystem service assessments and decisions (***chapter 2***); (ii) to determine the extent to which the costs of strategies to preserve natural capital assets are affected by climate change and payments for ecosystem services (***chapter 3***); (iii) to develop an approach to preserve natural capital assets and ecosystem services that are robust to the uncertain impacts of climate change (***chapter 4***); and (iv) to assess the costs and effectiveness of actions to manage ecosystem services under climate change and external economic drivers (***chapter 5***).

In this concluding chapter, I summarise the main findings from each previous chapter of this thesis, and discuss their implications for the management of ecosystem services under uncertain global change. I then synthesise the major contributions, discuss challenges and limitations, and recommend future research directions.

6.1 Main findings

6.1.1 Incorporating climate change into ecosystem services assessments and decisions: A review. (Chapter 2)

Climate change is a threat to the provision of ecosystem services (Scholes 2016), yet the precise nature of future impacts can be difficult to determine due to high uncertainties and other confounding drivers (IPCC 2014). Critically, there were no quantitative synthesis of drivers, methods, impacts or decisions related to ecosystem service assessments under climate change prior to my thesis. To determine how climate change and other drivers were incorporated into ecosystem service assessments and decisions (objective 1), I conducted a systematic literature review (chapter 2, Runting *et al* (2017a)). I found that the overall impacts of climate change were largely negative, although there was substantial variation across services, drivers, assessment methods, and localities, and in some cases the impacts were positive. In particular, carbon sequestration had the most variable response to climate change, and CO_2 fertilisation was responsible for the largest amount of variation across services. Substantial gaps were identified in the locations that were assessed, with most studies being focused on the USA and Europe. Given the variation in the impacts of climate change, further studies beyond these regions are essential to ensure an adequate understanding of impacts, rather than relying on averages or aggregates from other contexts. Somewhat concerningly, we found that the method used could impact the results. Specifically, studies that used expert elicitation gave more frequent negative results than studies employing empirical or quantitative modelling methods, and this effect was statistically significant. Although uncertainty was often incorporated in assessments, I found that this was largely limited to scenario analyses that incorporated variation in the magnitude of climate change. Numerous other sources of uncertainty exist, and ideally these would be incorporated to allow meaningful integration with decision making. The relatively few studies that incorporated decision making did not assess how well their proposed solutions performed under a range of uncertainties. For management or policy to ensure the delivery of ecosystem services, I recommend integrated approaches that incorporate multiple drivers of change and account for multiple sources of uncertainty are needed.

6.1.2 Costs and opportunities for preserving coastal wetlands under sea level rise. (Chapter 3)

Coastal ecosystems are particularly vulnerable to climate change through rises in sea level (Lovelock *et al* 2015). Pre-emptive planning to set aside key coastal areas for wetland migration is critical for the long-term preservation of ecosystem services, yet we have limited understanding of the economic costs and benefits of doing so. I used data and simulations from Moreton Bay (Queensland, Australia) to determine the extent to which the costs of strategies to preserve natural capital assets were affected by climate change (specifically sea level rise) and payments for ecosystem services (objective 2). I found that substantial changes in the distribution of coastal wetlands under seal level rise by 2100 increased the costs of protecting a given area target (relative to no sea level rise). The landward movement of coastal wetlands, combined with the positive association between land values and elevation, drove the increase in costs. In addition, the rate of sea level rise influenced the results - the higher the sea level rise projection, the higher the opportunity cost of expanding the protected area network. Despite the higher costs with sea level rise, payments for ecosystem services had the potential to substantially reduce the net cost of pre-emptive protection, and in many cases resulted in a profit in the long run. I also found that the potential cost savings from payments for ecosystem services could be further increased under different market conditions, most notably if prices for carbon increased. Although, higher rates of sea level rise again reduced the effect of payments for ecosystem services under all market conditions. Even in the cases were a profit was possible in the long run, the immediate costs to planning authorities was still high, as the payments for ecosystem services would not start flowing until the benefits materialised. Despite these short term challenges, I conclude there is substantial potential for payments for ecosystem services to fund the expansion of protected areas under climate change, particularly if planners take a long-term view of benefits and costs.

6.1.3 Risk-sensitive conservation planning under climate change: A case study of coastal ecosystem services under sea level rise. (Chapter 4)

The precise spatial and temporal impacts of climate change on ecosystem services are inherently uncertain (Scholes 2016, Runting *et al* 2017a), so the outcomes of planning long term conservation actions, such as designating protected areas, are subject to substantial risks. In order to explicitly incorporate these risks, I developed an approach to preserve natural capital assets and ecosystem services that is robust to the uncertain impacts of climate change (objective 3). Specifically, I incorporated a risk-sensitive resource allocation approach from finance, Modern Portfolio Theory, within a conservation planning algorithm. This approach extended previous applications of Modern Portfolio Theory to conservation by including multiple objectives, allowing the selection of discrete planning units, and specifying connectivity requirements among planning units. I applied this approach to a case study of conservation planning for coastal ecosystem services using a similar study area to ***chapter 3***. This application additionally incorporated uncertain rates of sea level rise, potential error in elevation data, uncertain rates wetland accretion and a range of other uncertain modelling parameters. I compared my new approach to planning for specific rates of sea level rise, but ignoring uncertainty (in both sea level rise and other parameters). I found that ignoring uncertainty was a high-risk strategy, even when planning for the highest rate of sea level rise, compared to our risk-sensitive approach. I ascertained that reducing the risk of the conservation also reduces the expected conservation returns, but the risk preference of the decision maker(s) will ultimately determine the specific level of risk to accept. My approach developed here is likely to be of use to decision makers with any degree of risk aversion, who also aim to achieve multiple conservation objectives. Although illustrated for coastal ecosystems under sea level rise, the problem formulation is adaptable to other contexts and uncertainties.

6.1.4 Managing livestock production and greenhouse gas regulation under global change in northern Australia. (Chapter 5)

Whilst accounting for the impacts of climate change is clearly important, it is also vital to consider the changing economic conditions occur in parallel with climate scenarios (Bryan 2013). Here I determined the costs and effectiveness of actions to manage ecosystem services under climate change and external economic drivers (objective 4), using an integrated systems modelling approach for the livestock production landscapes of northern Australia. I first assessed impacts on livestock production and greenhouse gas regulation from climatic drivers alone (i.e., changes in temperature, precipitation and fire), then included coupled external economic drivers (i.e., productivity growth, the costs of farm inputs, livestock price and carbon price projections). I found that while the profitability of livestock production increased with growing demand, rising farm input prices and biophysical constraints posed by climate change counteracted some of these gains, reducing the number of animals produced. Emerging strategies, such as planned early dry season burning or nitrate supplementation, were able to reduce greenhouse gas emissions, but they came with financial costs (i.e., lost profit). Higher carbon prices under some global change scenarios were able to compensate for the costs of controlled burning, but costs remained a barrier for nitrate supplementation, even with a carbon price. All results were spatially variable, indicating the importance of conducting spatially explicit assessments rather than relying on averages from other regions, or assuming homogenous patterns from point-based analyses. Perhaps most importantly, this work illustrates that coupled economic drivers (in addition to climatic drivers) can influence the viability of actions to manage ecosystem services under climate change. These economic drivers are particularly important to take into account when considering policies to influence the behaviour of landholders overtime.

6.2 Major contributions

My thesis draws from the disciplines of economics, operations research, and environmental management to advance the knowledge and practice of incorporating climate change and other global drivers into decision making for ecosystem services. Specifically, I focus on developing and assessing different management approaches to determine their effectiveness. The overarching contributions are detailed below.

I first established the prevailing impacts of climate change and other key drivers on a range ecosystem services, and ascertained the dominant approaches for determining these impacts (***chapter 2***). I then revealed the key gaps in these approaches. Most pertinently, I identified the need to integrate (i) multiple objectives, (ii) multiple drivers, and/or (iii) multiple sources of uncertainty, into decision making for ecosystem services (***chapter 2***). Subsequent chapters of the thesis were used to address these identified gaps:

(i) Previous research has found that incorporating multiple objectives is vital for balancing trade-offs where objectives compete (Moilanen *et al* 2011), and taking advantage of co-benefits where possible (Bryan *et al* 2016b). Similarly, I found that in the context of global change, incorporating multiple drivers was valuable both in cases where ecosystem services were largely synergistic (***chapter 4***), and where they were competing (***chapter 5***). This extends the findings of prior research using previously untested geographies in the context of global change.

(ii) Although previous research has incorporated multiple drivers when assessing natural capital or ecosystem services (e.g., Bateman et al (2013), Bryan et al (2015, 2016a), and Struebig et al (2015)), this has not previously been attempted for livestock production and greenhouse gas regulation in tropical rangelands (***chapter 5***). Assessing these services in tropical rangelands is particularly challenging due to the influence of climate on fire, amongst other factors (Bowman *et al* 2009). Here, I revealed that the complex interplay of multiple drivers resulted in limited economic potential for emissions abatement in this system.

(iii) Although uncertainty related to climate change is a focal theme of this thesis, I went beyond climate change uncertainty to incorporate other significant sources of uncertainty which are often overlooked. These additional sources of uncertainty (in the parameters for modelling coastal wetlands (***chapter 4***) and livestock production (***chapter 5***)) substantially increased the variation in projections of ecosystem services. Therefore, I recommend that model parameter uncertainty should not be overlooked in assessments or decisions relating to natural capital and ecosystem services.

This thesis also integrates methods from finance and economics with established methods for the assessment and management of ecosystem services. Some methods from economics have been regularly used in conservation planning or integrated assessments, such as calculating opportunity costs (as used in ***chapters 3, 4*** and ***appendix A***), or determining profits (as used in ***chapters 3, 5***, and ***appendix A***) (Naidoo and Adamowicz 2006, Naidoo *et al* 2006, Naidoo and Iwamura 2007). However, I also advanced the development of emerging economic approaches with conservation planning. Specifically:

- I illustrated how payments for ecosystem services can fund the expansion of protected areas under climate change. Previously, climate change has been treated as a threat to payments for ecosystem services schemes (Friess *et al* 2015), but I demonstrate that under climate change, markets for ecosystem services show substantial potential to preserve our natural capital assets (***chapter 3***). This further shows that the designation of protected areas and markets for ecosystem services can complement each other, rather than being competing approaches.
- I integrated Modern Portfolio Theory within a typical conservation planning framework to incorporate correlations in projected outcomes among sites to ensure a complimentary set of connected sites are selected (***chapter 4****)*. My approach also includes multiple objectives, discrete site selection, and ecological connectivity. This is a significant advance on previous applications of Modern Portfolio Theory to conservation, as these were either aspatial (Koellner and Schmitz 2006) or did not consider the multiple objectives, and spatial dependencies inherent in conservation problems (Ando and Mallory 2012a).

When managing ecosystem services in an era of global change, managers must consider a wide range of objectives, drivers and uncertainties. Together, these thesis chapters advance our understanding of how this can be accomplished.

6.3 Limitations and future research

The chapters in this thesis conceptualise and demonstrate the management of ecosystem services in complex environments. In this section, I discuss the primary limitations of these contributions, and suggest future research directions to advance this work.

6.3.1 Integrating ecosystem service flows and beneficiaries

Ideally, ecosystem service research incorporates how a service is supplied ('supply side'), along with its flow to the beneficiaries of the service ('demand side'), thus illustrating the importance of natural capital to people (Tallis *et al* 2012). In ***chapters 3***, ***4***, and ***5***, I have primarily focused on management for the supply side of ecosystem services, or natural capital assets. However, demand for ecosystem services was taken into account for carbon sequestration (***chapter 3***), greenhouse gas regulation (***chapter 5***), and livestock production (***chapter 5***) through market prices for these services. In these cases, incorporating demand was relatively straightforward, as modelling spatially explicit flows to the beneficiaries of the service was not required. Focusing on the supply side is a prevailing trend in ecosystem services research (Martinez-Harms *et al* 2015, Runting *et al* 2017a), despite the importance of demonstrating benefits to people for integration with planning and policy decisions (Daily *et al* 2009, Guerry *et al* 2015).

For many ecosystem services, such as storm protection (Arkema *et al* 2013), pollination (Ricketts and Lonsdorf 2013), or hydrological services (Brauman *et al* 2007), the spatial flows to beneficiaries are of vital importance. Accounting for service flows means the spatial configuration of areas of supply, relative to beneficiaries are of consequence (Mitchell *et al* 2015, Eigenbrod 2016), and can substantially change the relative importance of different areas of service provision (Bagstad *et al* 2012). Ideally, future research should expand on the methods developed and used in this thesis need to thoroughly incorporate the spatial flows of services. Specifically, the methods in ***chapters 3*** and ***4*** could be modified to include piecewise linear approximations of non-linear functions, or the incremental updating of parameter values (Golovin *et al* 2011), based on models of service dynamics. Such an approach has not yet been applied to planning for ecosystem services and would represent an important advance. In any case, the development of land use or management plans should entail iterative feedback with key stakeholders (beneficiaries), to ensure the social acceptability of solutions (Luck *et al* 2012, Arkema *et al* 2015). Incorporating both dynamic updating and stakeholder input into planning methods is a valuable direction for future research on the optimal management of ecosystem services.

6.3.2 How much complexity is enough?

Incorporating multiple uncertainties and drivers into assessment and planning for ecosystem services may not always be necessary. Some environmental decisions or policies may be insensitive to future changes. For example, in ***Appendix A*** (Runting *et al.*, 2015) my main finding (that cross-jurisdiction collaboration leads to efficiency gains when planning for multiple competing objectives) held true under an extensive sensitivity analysis, which varied commodity prices, opportunity costs, species viability, and the interpretation of public policy targets. However, the optimal spatial location of specific land use zones showed some variation in relation to these parameters. It is unlikely that the impacts of global change will reverse broad policy decisions surrounding land use and management, such as cross-jurisdictional collaborations (Kark *et al* 2009, Runting *et al* 2015), restrictions on broad scale land clearing (Evans 2016), or improved management of production systems (Laurance *et al* 2010, Brodie *et al* 2012). In these cases, a detailed assessment of drivers and uncertainties may be unnecessarily cumbersome, and I do not recommend that the complexities included in these thesis chapters be applied to every environmental decision.

Although incorporating the full range of complexity is not required in all cases, it can be difficult to determine in what contexts to include these complexities, and how much complexity to include (Boschetti 2008, Evans *et al* 2013). Thoroughly assessing and incorporating a range of drivers, uncertainties, and objectives can require substantial resources (i.e., time, money, and expertise). In many cases, assessing a range of drivers is a worthwhile investment as it can substantially change the management strategy. For example, in ***chapter 5***, I found that incorporating global economic drivers switched which management actions were the most profitable over a large spatial scale. However, in other cases, unnecessary resources may be allocated to the collection and incorporation of additional information which does not change the management strategy (or does not alter it enough to justify the additional cost) (Pannell 2006, Grantham *et al* 2008). Even where multiple drivers (***chapter 5***) and uncertainties (***chapter 4***) are incorporated, this does not exclude the potential of other drivers from having an impact on the system, and potentially the management outcomes (e.g. invasive species (Adams and Setterfield 2013)).

No individual assessment or project can include every complexity, so in most cases it is necessary to prioritise some drivers and uncertainties over others. The key drivers of change (or threats) are commonly determined by expert elicitation, either via a focus group, survey or more informal methods (Donlan *et al* 2010, Bohensky *et al* 2011, Carwardine *et al* 2012). However, experts are limited by the current state of knowledge, and can also be subject to biases (Martin *et al* 2012). A (partial) solution to this problem is to focus research efforts on the assessment of the relative and cumulative impacts of multiple drivers of change (e.g., Aber *et al* (2001)), to expand the currently limited knowledge base. Assessing both relative and cumulative impacts of multiple drivers could be useful in determining which combination of drivers has the most influence of management outcomes and should therefore be the focus of analyses. The primary drawback of this process is that the most important drivers are likely to vary across different locations, objectives, and types of management decisions, making the generalisation of findings potentially difficult. Alternatively, the value of including additional drivers (and their uncertainties) could be determined *a priori* using value of information analysis – a method which determines the value in collecting additional information for decision making (Runge *et al* 2011). However, the application of this method may similarly require additional resources (i.e., time and expertise), that are beyond the scope of many projects. Nonetheless, determining the optimal level of complexity to include in decision making for ecosystem services remains an important focus for future research.

6.3.3 Unknown unknowns

I have illustrated that prioritising and incorporating *known* drivers and uncertainties into management decisions concerning ecosystem services is a useful, but challenging, task. However, even the most sophisticated models of ecosystem services do not include deep uncertainty (‘unknown unknowns’ or ‘black swan’ events), which may have catastrophic impacts (Makridakis and Taleb 2009, Farley and Voinov 2016). Potential examples of ‘black swan’ events include armed conflicts, extreme drought, earthquakes, and terrorism, although even these risks can be quantified and incorporated into planning in some cases (e.g., armed conflict risk in Hammill et al (2016)). Whilst such events have low predictability, rare events are inevitable, given enough time (Taleb 2007). Diversification (such as in ***chapter 4***) and other decision-theoretic methods such as info-gap (Regan *et al* 2005, Moilanen *et al* 2006), may help to reduce risk from these events, but these risks cannot entirely be eliminated through either method (Sniedovich 2007, Hummel *et al* 2009).

Alternative methods, or further development of existing methods, are required to explicitly account for this type of uncertainty. For instance, typical scenario thinking and development (i.e., based on trends) can be reframed to challenge the perceived bounds of uncertainty (Wright and Goodwin

2009). Similarly, methods for strategic foresight can encourage thinking that is unbound by previous experiences, can help to highlight otherwise unanticipated emerging threats to incorporate within scenarios or decision-making (Cook *et al* 2014). However, further research is needed to demonstrate how futures thinking, or other methods for addressing deep uncertainty, can be integrated with spatial planning approaches. In this context it is important to keep in mind the benefits of exploring deep uncertainty relative to learning more about known uncertainties. A framework exists for allocating ecological monitoring effort among these two types of uncertainties (Wintle *et al* 2010), and further research could potentially extended this to decision making for ecosystem services.

6.4 Concluding remarks

Incorporating the impacts of global change into ecosystem service assessments and management decisions is critical to ensure their continued provision (Polasky *et al* 2011, Nelson *et al* 2013). Developing new approaches, and testing the performance of existing approaches in different contexts, is vital to ensure we are adequately equipped to adapt to climate change and associated complexities. This thesis advances our understanding of how to manage natural capital assets and ecosystem services that are impacted by climate change and other global drivers, particularly where there are multiple objectives, multiple drivers, or multiple uncertainties. In doing so I provide tangible solutions to manage our environment in an era of global change.

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Appendix A: Alternative futures for Borneo show the value of integrating economic and conservation targets across borders

This section is reproduced from the following paper, with some alterations to formatting and structure:

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A.1 ABSTRACT

Balancing economic development with international commitments to protect biodiversity is a global challenge. Achieving this balance requires an understanding of the possible consequences of alternative future scenarios for a range of stakeholders. I employ an integrated economic and environmental planning approach to evaluate four alternative futures for the mega-diverse island of Borneo. I show what could be achieved if the three national jurisdictions of Borneo coordinate efforts to achieve their public policy targets and allow a partial reallocation of planned land uses. I reveal the potential for Borneo to simultaneously retain ~50% of its land as forests, protect adequate habitat for the Bornean orangutan (*Pongo pygmaeus*) and Bornean elephant *(Elephas maximus borneensis*), and achieve an opportunity cost saving of over US$43 billion. Such coordination would depend on enhanced information sharing and reforms to land-use planning, which could be supported by the increasingly international nature of economies and conservation efforts.

A.2 INTRODUCTION

All United Nations member states have sanctioned national efforts to pursue environmental sustainability under the Convention on Biological Diversity and the Millennium Development Goals. Simultaneously, states have set ambitious national targets for economic growth, development and trade, often without assessing how these targets align or conflict with sustainability agendas. Balancing the needs for economic development with international commitments to protect biodiversity is a global challenge. Achieving this balance will require a whole-landscape approach to land-use planning that incorporates the targets sought by multiple sectors (DeFries & Rosenzweig 2010). The potential for systematic planning approaches to deliver large gains in economic and environmental efficiency has so far been demonstrated in efforts to re-design protected area networks within (Fuller et al. 2010) and across (Kark et al. 2009) political borders. We now need to understand whether this potential can be realised in regions with multiple land-uses and multiple, often conflicting, objectives. Sustainable allocation of land-uses will require a dialogue on potential futures and an understanding of the possible consequences of alternative strategies for diverse sectors (Tress & Tress 2003; Game et al. 2014).

Tropical forests regulate regional and global climate, provide a wide range of ecosystem services to over a billion people, and support ~50% of described species (World Bank 2001; Dirzo & Raven 2003; Bonan 2008). The forests of Borneo, the third largest island in the world, have an average aboveground biomass that is 60% higher than the Amazonian average (Slik et al. 2010). The island harbours an estimated 14,423 plant and 1,640 vertebrate species, of which 28% are endemic (Table A.1) and 534 (3%) are considered to be threatened with extinction (IUCN 2012). The extent of forest on Borneo declined by 16.8 million ha (30%) from 1973-2010 because of agricultural expansion and ENSO-induced wildfires (Gaveau et al. 2014). Indonesia and Malaysia are major exporters of palm oil; in 2012 these countries collectively produced >80% of the global supply (FAO 2013). Furthermore, the governments of Malaysia and Indonesia seek to increase the area of oil-palm and industrial timber plantations (ITP) on Borneo by 7.1 million hectares over the next two decades. The planned expansion of oil-palm plantations in Indonesian Borneo alone is projected to contribute carbon dioxide emissions (CO_2) of 0.12–0.15 GtC yr^{-1} from 2010 to 2020, equating to approximately 34% of Indonesia's total land sector emissions (Carlson et al. 2013). High rates of forest conversion and degradation have prompted inter-governmental agreements between Indonesia, Malaysia and Brunei Darussalam to protect and sustainably use the forests that remain in Borneo (Proctor et al. 2011). For example, the Borneo Initiative is a project focused on sustainable forest management (The Borneo Initiative 2013), and the Heart of Borneo initiative aims to sustainably manage ~20 million hectares of the mountainous core of the island (Government of Brunei Darussalam, Government of Indonesia, and Government of Malaysia 2009). While political coordination across borders will likely improve the efficiency of meeting economic and conservation goals, these potential gains have not previously been quantified.

Table A.1 | Biological and socio-economic background for Borneo. Panel (a) shows the species occurring in Borneo, and number of endemics. Plant species counts are extrapolated estimates made by Roos et al (2004). Panel (b) shows a comparison of the three nations on Borneo across selected indicators. The corruption rank is out of the 177 countries assessed, with 1 being the least corrupt (Transparency International 2013). Gross domestic product (GDP) per capita is measured in purchasing power parity (PPP) equivalent to 2011US$ (The World Bank Group 2015).

a

Taxa	***Total number***	***# endemics***	***Source***
Plants	14,423	4,089	(Roos et al. 2004)
Frogs	141	88	(Inger & Voris 2008)
Reptiles	276	89	(Uetz et al. 2013)
Terrestrial mammals	196	40	(Corbet & Hil 1992)
Freshwater fish	394	149	(Kottelat 1989)
Birds (resident and migratory)	633	53	(Myres 2009)

b

Indicator	***Indonesia***	***Malaysia***	***Brunei***
Area on Borneo (km^2)	548,005	198,161	5,770
% of area protected	20%	9%	22%
Corruption rank	114	53	38
GDP per capita (PPP)	$9,561	$23,338	$71,777
Type of government	Presidential democratic republic	Constitutional elective monarchy	Absolute monarchy

We explored four alternative futures for Borneo, each representing a set of policy objectives and a planning strategy: (1) baseline (current land-use allocations are executed); (2) uncoordinated, state-based planning to achieve policy targets (with the Malaysian states of Sabah and Sarawak treated separately); (3) coordinated planning in the mountainous interior of Borneo, with state-based planning outside this area; and (4) integrated planning across all four states (allowing for both jurisdictional coordination and the reallocation of some land-uses) to achieve either (a) existing public policy targets or (b) alternative biodiversity targets seeking to achieve representative protection of dominant vegetation types (Table A.2). For each scenario (except the baseline), we identified land-use configurations that achieve the stated targets. We evaluated each scenario by determining the opportunity costs of meeting existing policy targets for key economic and conservation features, namely forest cover, protected areas, Bornean orangutan (*Pongo pygmaeus*),

Table A.2 | A brief description of scenarios and the socio-political challenges involved with implementing them.

Scenario Name	*Description*	*Challenges*
1. Baseline	The current land-use allocation in each state is assumed to be fully executed (e.g., all oil-palm concessions are planted).	Inefficient: Some planned plantations are in unsuitable locations; conservation opportunities are missed.
2. State-based planning	State or national targets are sought within each state. Minimal changes can be made to existing land-use allocations.	States must adhere to their stated targets. This may be difficult in practice due to corruption and vested interests.
3. Coordinated planning inside the core, with state based planning outside	Coordination between states within the mountainous interior of Borneo. State-based planning and targets are assumed outside of this area.	As per scenario 2, but all states must implement the agreed upon (but non-binding) vision of the Heart of Borneo.
4a. Integrated planning	Uses the combined targets from scenario 2 but ignores state boundaries and modifies land-use allocations where possible.	As per scenario 2, but states must agree on island-wide targets. Implementation will require an appropriate institutional platform, and compensation mechanisms or payment schemes.
4b. Integrated planning alternative conservation targets	As per scenario 4a, but 70% of the extant distribution of each forest type must be protected overall. The faunal targets were set at 70% of the distribution of each species to correspond to the forest cover target.	As per scenario 4a, but this scenario highlights that current conservation targets are inadequate. Extensive consultation is required to specify island-wide conservation targets that capture a range of biodiversity features and the needs of local communities.

Bornean elephant (*Elephas maximus borneensis*), oil-palm and ITP (Table A.2). We also evaluated the scenarios in terms of the extent of land allocated to conventional (CL) or reduced impact logging (RIL) and the potential for reducing CO_2 emissions relative to the baseline scenario. We reveal the potential for Borneo to simultaneously: retain ~50% of its land as forests, protect adequate habitat for orangutan and elephant, and achieve an opportunity cost saving of over US$43 billion. The value of integrating economic and conservation goals through trans-boundary collaboration will be substantial wherever the costs and opportunities for achieving goals vary across borders.

A.3 METHODS

A.3.1 Land-use decision support tool

The planning goal was to meet a set of conservation and economic targets, while minimising the opportunity cost of allocating land to particular uses (for scenarios 2-4). We used Marxan with Zones conservation planning software, which uses simulated annealing as the optimisation algorithm to find multiple, near optimal solutions for this land-use planning problem (Watts et al. 2009). This algorithm also accounts for the impact of undesirable combinations of adjacent land-uses (e.g. avoids placing oil-palm plantations adjacent to protected areas, where possible). Each scenario (and scenario variation) was run 1000 times to ensure near-optimal solutions were found. We incorporated the relative probability of deforestation and assumed benefits were delivered in perpetuity (i.e. if an area is re-zoned protected, it is expected to remain forested indefinitely although we acknowledge that this may not be the case over long time frames under climate change (Struebig et al. 2015)). We also discounted costs and profits in perpetuity (i.e. assuming that the revenue from each land-use will continue indefinitely), but did not include dynamic factors, such as commodity price fluctuations.

We accounted for the contribution to targets and opportunity costs of meeting these targets in five general land-uses: 1. protected areas; 2. logging (CL or RIL, depending on scenario); 3. ITP for pulp and paper (monocultures of fast growing trees); 4. oil-palm; and 5. other non-forested land-uses not incorporated in the above (Table A.5). This “other non-forest” category represents the land remaining for other development (i.e. urban, mining, or other agriculture) after achieving the public policy targets. The “other non-forest” category was not further disaggregated or explicitly modelled due to the spatial dominance of the first four categories in the landscape. Mining, for example,

Table A.3 | Conservation and economic targets for Sabah, Sarawak, Kalimantan and Brunei Darussalam. Sources are provided in Table A.4.

Target	***Sabah, Malaysia***	***Sarawak, Malaysia***	***Kalimantan, Indonesia***	***Brunei Darussalam***
Forest cover	50% of land area (37,000 km^2)	50% of land area (61,885 km^2)	45% of land area (240,587 km^2)	75% of land area (4,337 km^2)
Protected areas	17% of land area (12,571 km^2)	17% of land area (21,041 km^2)	17% of land area (90,888 km^2)	55% of area as "national forest estate" (3,180 km^2)
Orangutan	No conversion of forest with significant orangutan populations	No conversion of forest with significant orangutan populations	Stabilise all orangutan populations by 2017	N/A
Elephant	Secure long-term viability of elephant populations in the state	N/A	None	N/A
Reduced impact logging	All commercial forest reserve needs to be FSC certified	No directive outside of the Heart of Borneo area	All production forest to be converted to reduced impact logging	All exploitation forests follow sustainable practices
Oil-palm plantations	2.1 million ha	2 million ha	Double production (to 6.9 million 'productive hectares')	None
Industrial timber plantations	Increase by 837 km^2 (to 1,778 km^2)	Increase by 1,414 km^2 (to 2,883 km^2)	Increase by 13,900 km^2 (to 20,186 km^2)	None

while having significant localised impacts, was found to account for only a minor proportion of overall deforestation in East Kalimantan (Smajgl et al. 2009). The classes of protected areas included were specific to each country. For Brunei we accounted for forest reserves, national parks and wildlife sanctuaries. For Kalimantan we accounted for protection forest, national parks, nature reserves, recreation/community parks and wildlife sanctuaries. In Sabah we accounted for protection forest reserves, virgin jungle reserves, wildlife reserves, Sabah parks, wildlife sanctuaries and wildlife conservation areas. In Sarawak we accounted for wildlife sanctuaries, national parks, protection forest, communal forest, forest reserves, hunting reserves, virgin jungle reserves and parks. We used hexagonal grids of 10 km^2 (i.e.1.7 km in-circle radius) as the base spatial unit for the analysis. We also ensured that the mean land-use 'patch' size for each solution was within ±5% of the mean of the baseline scenario (28,216 ha).

We analysed targets for four geopolitical units: the country of Brunei Darussalam; the two Malaysian states of Sabah and Sarawak; and Kalimantan, the Indonesian part of Borneo. We did not analyse Kalimantan at the level of provinces, because despite a process of decentralisation in Indonesia, the five provinces of Kalimantan have less direct authority over their land resources compared to Brunei, Sabah, and Sarawak. State governments in Sabah and Sarawak largely decide on the allocation of budgets and land-uses, whereas Kalimantan depends on national level policy to inform these decisions.

A.3.2 Scenarios

<u>Scenario 1: Baseline</u>

This scenario represents existing land-use allocations and is based on the following assumptions:

1. Urban and mining areas cannot be changed to other land-uses.
2. All oil-palm and ITP concessions are planted.
3. All areas designated for limited production or production forests become active.
4. All classes of protected areas remain protected.

The data on existing land-use allocations were compiled in accordance with Wich et al. (2012), including industrial oil-palm plantation concession data for Kalimantan compiled by Carlson et al. (2013) and data for protected areas in Sabah from the Sabah Forestry Department (2013). Given the dearth of spatial information on oil-palm concessions in Sabah, we assumed land classified as conversion forest would be converted to oil-palm, unless another concession type was indicated.

This is likely to be an overestimation of oil-palm concessions in Sabah, but is appropriate for this scenario as it represents the worst case. We acknowledge that the full execution of existing land-use allocations may not be desirable due to community conflicts, low productivity and environmental issues.

Scenario 2: State-based planning

This scenario reflects a state based planning approach to achieve targets (Table A.2, Table A.4). The following land-use transition rules apply based on current policy or practice (Figure A.9a):

1. Urban and mining areas cannot be changed to other land-uses.
2. Current planted ITP and oil-palm plantations remain.
3. All classes of protected areas remain protected.
4. New protected areas can occur where there is forest cover (i.e. intact, logged, agroforest/regrowth, severely degraded).
5. New oil-palm plantations can be established anywhere except urban areas, mining areas, areas not suitable for oil-palm (e.g., land with a slope above 45° (Table A.6b)), and planted ITP. This can include severely degraded grasslands, where suitable.
6. New ITP can be anywhere except urban areas, mining areas, areas not suitable for oil-palm, oil-palm concessions, and planted oil-palm.
7. Current oil-palm concessions can only become oil-palm or "other non-forest".
8. Land that is not suitable for oil-palm can only become "other non-forest", protected, or logging.
9. Logging can only occur where there is sufficient forest cover (i.e. not agroforest/regrowth or severely degraded forest types (Hoekman et al. 2010)).
10. "Logging" can be either CL or RIL in Sarawak and only RIL in the other states, to reflect their targets (Table A.2). CL can be converted to RIL and vice versa.

Scenario 3: Coordinated planning within the mountainous core

This scenario reflects the vision of the Heart of Borneo initiative, where coordinated planning between states occurs within a defined area in the mountainous interior of Borneo. Land-use transition rules within the defined Heart of Borneo area follow those stated in WWF's vision for a "Green Economy" (Dean & Salim 2012) including:

1. Standing primary and secondary forest cannot be developed.
2. Active logging concessions are converted to RIL.

3. Inactive logging concessions are not logged.
4. Oil-palm and ITP expansion can only occur where a concession already exists and the land is degraded/idle, and excludes development in peatland, swamp forest, and protected areas.
5. Urban and mining areas cannot be changed to other land-uses.

As the Heart of Borneo initiative does not provide land-use transition rules beyond the defined Heart of Borneo, we have applied the land-use transition rules from scenario 2 for the remainder of the island (Figure A.9a).

Scenario 4: Integrated planning

This scenario reflects coordinated planning between states with the land-use transition rules employed for scenario 2, but with the following relaxations (Figure A.9b):

1. Protected areas need not remain protected.
2. Oil-palm and ITP concessions can be protected or logged where there is current forest cover (i.e. intact, logged, agroforest/regrowth, severely degraded).
3. ITP can be established on oil-palm concessions.
4. Oil-palm and ITP concessions can become ‘other non-forest’.

This scenario (Scenario 4a) was also modified to include ecosystem-based targets, representing a more integrated approach to conservation. In this modified scenario (Scenario 4b), 70% of the remaining extent of each forest type (i.e. montane, lowland, peat swamp, swamp, riverine, mangrove, and shrubland (Miettinen et al. 2012)) must be protected overall. The targets for orangutan and elephant were reduced to 70% to reflect the forest type target. The aim of this was to encompass a greater range of conservation features not specifically mentioned in government policy documents, whilst still allowing for the expansion of other land-uses.

For all scenarios, the opportunity costs were derived by discounting into perpetuity (see ‘opportunity costs’ below). Similarly the expected benefits (i.e. habitat for endangered species) are expected to remain in perpetuity.

A.3.3 Opportunity Costs

The following equation was used to determine the opportunity cost of each land-use change (adapted from Naidoo and Adamowicz (2006)):

$$L_m = \sum_{i=1}^{I} \left\{ \sum_{k=1}^{K} \left[P_{ik} \left(\frac{R_{ik}}{\delta} + C_{ik} \right) \right] - \left[\frac{R_{im}}{\delta} + C_{im} \right] \right\}$$

Where L_m is the opportunity cost of land-use *m* (L_m is >= 0), P_{ik} is the probability that parcel *i* will be converted to land-use *k*, R_{ik} is the average annual profit (or loss) associated with land-use *k* for parcel *i, δ* is the discount rate, C_{ik} is the profit (or loss) from converting parcel *i* to land-use *k*, R_{im} is the average annual profit from land-use *m* for parcel *i*, and C_{im} is the profit (or loss) from converting parcel *i* to land-use *m*.

In the absence of complete information on the probability of future land-use (P_{ik}), we used the probability of deforestation (detailed below) and assumed that the most lucrative alternative land-use would be conversion to oil-palm for deforested areas, or RIL for those areas that are to remain forested. Specifically, for deforested areas we used the net present value (NPV) of oil-palm production (average annual oil-palm profits discounted into perpetuity, plus profits from timber harvested during conversion, less the administrative costs of conversion) less the NPV of the selected land-use. For those areas which would remain forested, we used the NPV of RIL (annual RIL profits discounted into perpetuity, less administrative costs), less the NPV of the selected land-use. For the discount rate (*δ*) we used 10%, as this is consistent with other studies in the region (Edwards et al. 2011; Fisher et al. 2011a; Venter et al. 2013).

Logging Profit

The estimated profit from timber harvesting was obtained from data on timber yields, costs and revenues for CL and RIL (Table A.7).The mean value per hexagonal 10 km^2 grid cell varied, depending on:

1) Forest condition. Values for forests that have been logged previously were estimated by reducing the volumes from intact forest by 46% for Kalimantan and Sarawak (based on the meta-analysis by Putz et al. (2012)), and by 70.4% for Sabah (based on data from the Yayasan Sabah Forest Management Area (Fisher et al. 2011b)). Volumes extracted from intact forests in Sabah were generally much higher than in Kalimantan and Sarawak (c. 117-138 vs. 25-90 m^3 ha^{-1} for CL, or 106 vs. 28-48 m^3 ha^{-1} for RIL). The larger reduction factor for the volume obtainable from logged forests in Sabah partly reflects this more intense initial logging. Estimated volumes for timber from previously logged forest were much more similar across states (37.8, 23.6 and 23.5 m^3 ha^{-1} for

Sabah, Sarawak and Kalimantan, respectively). Areas of open agroforests, regrowth and severely degraded burnt forests were considered unlikely to be profitable for timber extraction, due to the presence of relatively few mature trees (Slik et al. 2002).

2) *Harvestable area*. Profits are usually reported per *harvested* hectare, as distinct from all hectares in a given management unit. For CL and RIL we therefore excluded all areas with a slope greater than a threshold slope specific to the state and logging method, and for RIL we also excluded areas within specified buffering distances of water bodies or watercourses.

Slope: Within each hexagonal 10 km^2 grid cell, we excluded all 90 m pixels with slopes greater than a value set for RIL or CL in each state. For RIL these values were > 16.7 degrees for Kalimantan (Sist et al. 1998) and Sarawak (Richter 2002), and > 25 degrees for Sabah (Lohuji & Taumas 1998). For CL this was > 25 degrees for all states (ECD 2002). It is possible to use skyline (aerial) yarding for RIL on steeper slopes (estimated 16.7 – 35 degrees (Sist et al. 1998)), however this practice is not yet widespread and we could not find sufficient financial information on costs and yields to enable its inclusion in this study. Similarly, helicopter logging can be used on steep slopes (though damage from felling and retrieval on slopes > 25 degrees may often exceed RIL principles). However, it involves very high costs and safety risks, and requires very tight co-ordination of felling and retrieval operations. Its use remains rare (Thang & Chappell 2005; Asia-Pacific Forestry Comission 2006; Bryan et al. 2013), and we found only two examples of its operation (one in Sarawak, and one in the Yayasan Sabah forest management area).

Buffering of water bodies: For RIL only, buffers of 100 m were placed around all water bodies, coastlines and large rivers (>= 50m wide) (Sist et al. 1998). The remaining rivers in the HydroSheds dataset were buffered by 40 m (Sist et al. 1998). The rivers in the HydroSheds dataset have minimum catchment areas of 20 km^2 (Lehner et al. 2006), and so to allow for buffering of watercourses smaller than this threshold, we applied a uniform reduction factor of 12.2% to the remaining harvestable area in each hexagonal 10 km^2 grid cell (based on the required area for buffering small watercourses in three reserves in Sabah with moderate rainfall (Pinard et al. 2000)).

The profit per hectare harvested (Table A.7) does not represent the NPV of logging. Logging companies with selective logging concessions do not harvest all of the concession area in the first year of operation, rather, a fraction of the area is harvested to ensure a continued revenue stream over the cutting cycle length (Sabah Forestry Department 2009; Edwards et al. 2014). Therefore, we divided the profit per hectare harvested by a cutting cycle length of 30 years (which is within the range of other studies (van Gardingen et al. 2003; Sabah Forestry Department 2009; Fisher et al. 2011a; Bryan et al. 2013)) to give an average annual profit per hectare. When applied to the harvestable area, this spatially explicit value represents R_i for the different types of logging. Logging operations incurred additional costs when the area to be logged was not initially covered by a logging concession. In these cases we applied an additional, once-off cost of \$17.25 ha^{-1}, to represent official and unofficial administrative costs (Art Klassen, pers. comm. 4 June 2014).

Plantation Profit

Oil-palm suitability was estimated by classifying a variety of biophysical properties of land units into five categories based on their suitability for oil-palm production (Table A.6a). If any given pixel had at least one of the biophysical properties classed as 'not at all suitable', it was excluded from further analysis. The remaining pixels were summed into a cumulative suitability map, which was then tertiled into 3 suitability classes (with 1 being the most suitable). The average annual profit for oil-palm production was derived from industry specific finance models (CH Williams Talhar and Wong Sdn Bhd 2011) based on state averages (for Sabah, Sarawak, and Kalimantan) of production per hectare of fresh fruit bundles and based on a crude palm oil price of \$800 per tonne (Table A.6b). Different scenarios of yield (full yield, 25% less, and 50% less) were applied to the 3 suitability classes to produce a Borneo-wide layer of potential revenue from oil-palm production (which was summarised at the planning unit level and used as R_i for oil-palm in equation 1). Oil palm is particularly well adapted to the humid tropics, which combined with growing demand, means revenues are likely to continue well into the future (Villoria et al. 2013). Oil-palm production was therefore measured in productive hectare equivalents (i.e. one hectare of oil-palm planted on land with 50% productive capacity equates to half a hectare of oil-palm production).

The average annual profit of industrial timber plantations (adjusted to 2009 US\$) was based on estimates from the Indonesian Forest Climate Alliance (2008). This attributed a different average annual profit to mineral (\$283.04) and peat (\$177.08) soils, due to the difference in productivity of these soil types. Any areas that were 'not at all suitable' for oil-palm were also considered to be unsuitable for ITP and were excluded from the calculation. The final values were summarised at the planning unit level and used as R_i for ITP in equation 1.

An additional, once-off cost (in year 0) was attributable in the cases where plantations were allocated on land that does not currently have a relevant concession (allowable in scenarios 2-4). For oil-palm, there are many steps involved in obtaining a licence. As official figures were unavailable, we estimated this value at \$907.58 per hectare (2009 US\$) using unofficial sources

(Borneo Climate Change 2013). For industrial timber plantations we estimated this value at $154 per hectare based on official guidelines (Republik Indonesia 2009).

In addition to revenues from oil-palm or industrial timber production, significant additional revenue can arise from timber harvest during conversion from forest to plantations (Venter et al. 2009). This was a once-off profit attributable to year 0 (i.e. it was not discounted). Timber revenues from clear-felling before conversion to oil-palm were estimated from logging revenues for each state and forest type (intact or previously logged), as given in the description of timber harvesting profits, combined with estimates of the percentage of additional timber that could be obtained from clear-felling rather than selective logging (Table A.7). The multiplication factors were estimated from data on timber harvesting profits (revenues minus costs) from three rounds of logging in the Yayasan Sabah Forest Management Area (from an area of approximately 310,000 ha) (Fisher et al. 2011b). That study reported values from logging in intact forests, from logging in previously logged forests, and from clear-felling of twice-logged forests. We assumed that the total volumes attainable by clear-felling an intact forest, or a logged forest, would be similar to the sum of volumes from sequential logging rounds reported in that study. For example, for intact forests, we assumed the amount that could be clear-felled in a single cut is similar to the sum of volumes reported from the first and second selective logging events, and the final clear-felling of the remnant stand. This calculation also assumes that levels of damage or wastage would be similar whether the felling occurs in sequential rounds or as a single clear-cut. It also does not account for possible regeneration between logging events, although this may have been small given the lengths of time between rotations in the Yayasan Sabah Forest Management Area (mean 16 years from first to second cut, and 1-7 years from second cut to clear-felling) (Fisher et al. 2011b). For Sarawak and Kalimantan, we modified the selective logging to clear-felling ratios to account for the higher relative volumes remaining after each logging round in these states (yields from logged forests being approximately 54% of yields from intact forest, compared to approximately 28% in Sabah). These clear-felling profits, less administrative start-up costs, form C_i for oil-palm or ITP in the opportunity cost equation above.

Protected area costs

The average annual management costs for protected areas (per hectare) was based on the optimal management of large Indonesian terrestrial national parks (approx. 120,000 ha) (McQuistan et al. 2006). This value (of 2004 US$6.17 ha^{-1} yr^{-1}) was similar to other estimates (Wilson et al. 2010; Kementrian Kehutanan 2013) and was adjusted to 2009 US$ ($7.01 ha^{-1} yr^{-1}) . The estimate includes field and administrative staff, equipment and infrastructure maintenance (McQuistan et al.

2006). This "loss" forms R_i for protected areas in equation 1. Additional start-up costs arise when a new protected area is established, which was estimated at $50 per hectare (Wilson et al. 2010). Were applicable -$50 ha^{-1} forms C_i for protected areas in the opportunity cost equation above.

Probability of deforestation

We employed a tree cover loss map for the period 2000–2010 (60x60m grid cell size) as the base dataset for modelling the probability of deforestation (Hansen et al. 2008; Broich et al. 2011). In this dataset 'tree cover' is defined as areas of trees (≥5m height) with >25% canopy cover and 'tree cover loss' as the removal of tree stands. We restricted our analysis to losses of intact forest cover that existed in year 2000. We randomly sampled 3,391 cells (of 6,234 available at a 1 km^2 resolution) and, of these, 451 cells had lost at least 20 hectares of forest. An equal number of cells with no forest loss were also randomly selected. The sub-sample of 902 cells was analysed using logistic regression, with elevation (Rabus et al. 2003), distance to cities (cities were defined as having a constructed surface area density greater than two per cent, using data from Sutton et al. (2010)), soil type (peat or mineral), and land-use (protected area, logging concession, limited production forest, production forest, conversion forest, monoculture industrial timber plantation or oil-palm plantation concession (Carlson et al. 2012; Wich et al. 2012)) employed as explanatory variables. The final model (R^2 of 0.68) included elevation and land-use as the most significant explanatory variables ($p < 0.05$), with forest at low elevations, in oil-palm plantation concessions and with conversion forest status having the strongest relationship with areas that have been cleared. The spatial layers of each of these variables were weighted by their respective coefficient to produce a relative probability map of deforestation.

A.3.4 Conservation objectives

The Bornean orangutan (*Pongo pygmaeus*), Bornean elephant (*Elephas maximus borneensis*) and forest cover had quantifiable governmental targets for their protection (Table A.3, Table A.4). The distribution of the Bornean elephant and orangutan was determined using Maximum Entropy Modelling (MaxEnt) (Phillips & Dudík 2008) (Figure A.1c and d). For the orangutan, this was supplemented using local knowledge, details of which can be found in Wich et al. (2012). For the elephant, location data (n=112) were collated from ground surveys and opportunistic sightings throughout the known elephant range between 1999 and 2011. Eleven spatial variables were

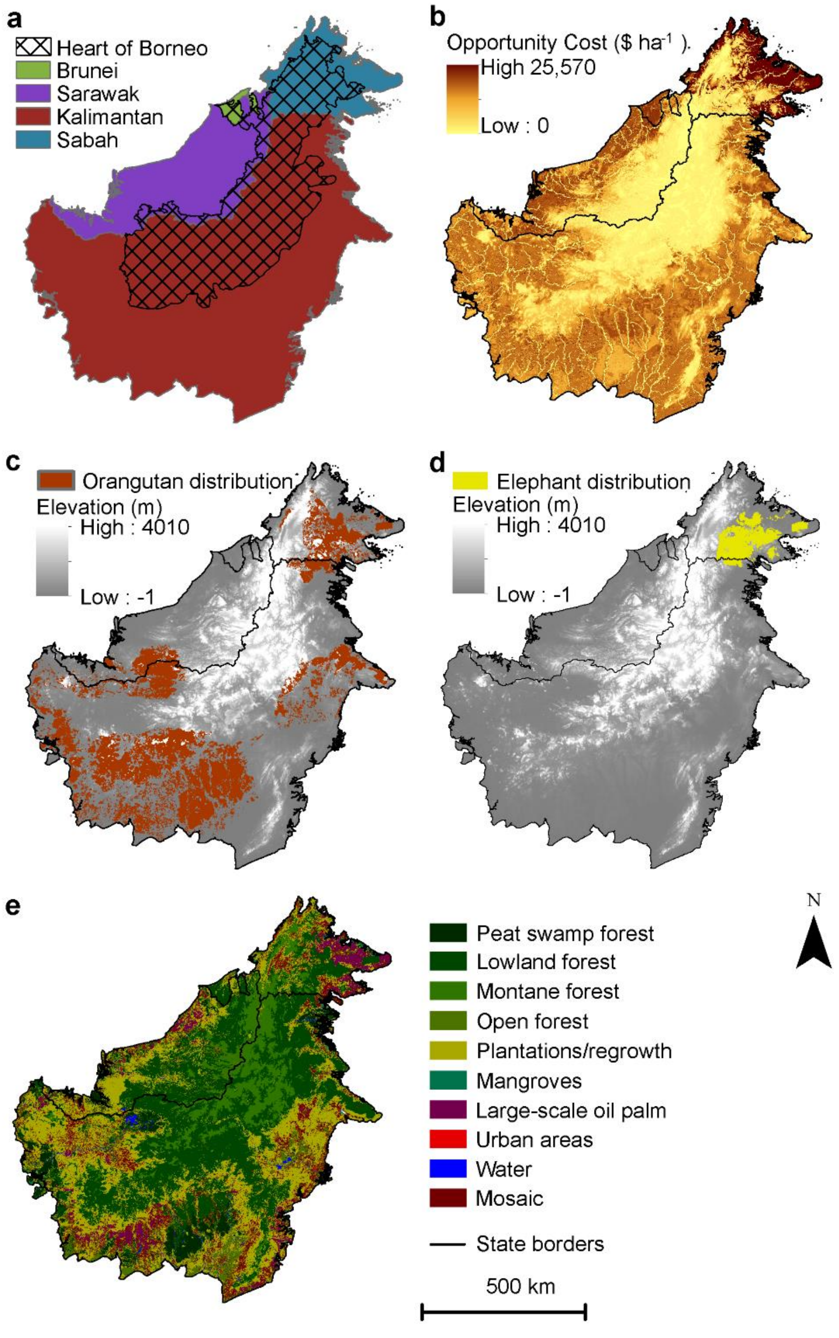

Figure A.1 | Context of Borneo: (a) Bornean states and the planned area for the Heart of Borneo initiative; (b) the opportunity cost (per hectare) of designating land as 'Protected'. An opportunity cost layer was developed separately for each of the possible land-uses; (c and d) the distribution of orangutan and elephant respectively; (e) current land-use and land cover (Miettinen et al. 2012). The orangutan distribution map is based on a predictive model, and is continually updated as new information becomes available on the presence and absence of the species from different regions. For example, we note that in 2015–2016 additional surveys in Sarawak will be carried out by the Wildlife Conservation Society.

identified as important for determining the suitability of elephant habitat. These included: four climatic variables, precipitation annual range, precipitation seasonality, temperature annual range, and temperature seasonality (WorldClim, ver. 1.4 dataset; http://www.worldclim.org); road density using 1999 to 2002 Landsat digitised data (Wich et al. 2012); soil data (FAO/IIASA/ISRIC/ ISSCAS/JRC, 2012); land cover (Hoekman et al. 2010); above-ground carbon stock (Baccini et al. 2012) that was converted into Mg CO_2 ha^{-1}; and three topographic variables, elevation (WorldClim, ver. 1.4 dataset), ruggosity and slope generated from elevation data (Jenness 2012). All spatial data were reclassified to 30 arc-seconds (approximately 1 km^2 resolution). MaxEnt was set to measure variable importance through jack-knifing, employed the logistic output algorithm, and default "auto feature" options. The model was validated with cross-validation with 10 replicates (Marmion et al. 2009) and measured performance using the mean area under the receiver operating characteristic curve (AUC = 0.977). Precipitation annual range, road density, soil types, and temperature annual range were identified as important explanatory variables for elephant (contributing 38.2%, 16.7%, 15.1%, and 9.8% respectively). A threshold probability of occurrence was determined using the maximum sensitivity plus specificity to derive a binary map of presence/absence. This was then clipped to the known distribution of elephant within 'forest'. Here, forest was defined to include areas that have intact, logged, severely degraded logged forest or areas with forest regrowth or agroforestry (modified from 2010 SarVision data (Hoekman et al. 2010): logged forests were defined as those within 5km from a satellite-visible logging road).

Carbon

We evaluated the change in carbon stock for each scenario relative to the current land-use plan (scenario 1). We calculated potential CO_2 emissions as the difference in time averaged CO_2 relative to a simple baseline scenario in which any area of existing forest is converted to oil-palm. Emissions from this conversion are assumed to equate to the extant aboveground carbon (Baccini et al. 2012) and including peat carbon if on peat soil. Carbon was converted to CO_2e using an emissions factor of 3.67 (IPCC 2006; Pendleton et al. 2012). Peat soil carbon net emissions were estimated using net CO_2 fluxes for a 25-year period (Hergoualc'h & Verchot 2013), which considers all inputs and outputs (and a single fire during forest clearance), giving an estimate of 1503 Mg CO_2e ha^{-1} over a 25-year time horizon. Below-ground carbon was not considered for mineral soils, due to a lack of data for all land-use transitions, and the comparatively small changes in time-averaged carbon stocks on most mineral soil types (e.g. converting primary forest to oil-palm would emit 32.0 Mg CO_2e ha^{-1} over 49 years on mineral soils (Don et al. 2011), compared to 1503 Mg CO_2e ha^{-1} for the same conversion over 25 years on peat soils).

We assumed protected areas would retain extant aboveground and peat carbon, and sequester carbon through natural regeneration. For degraded forest and forest regrowth with extant aboveground carbon contents less than intact forest, we assumed regeneration would increase aboveground carbon stocks to equal that of the average for intact forest. For severely degraded logged forests, we assumed protection would only increase the stock of carbon by 5%. Most of this class is in East Kalimantan Province and these forests were severely burned twice, in March-April 1983 and March-April 1998 (i.e. during the two most intense El Niño fire pulses on record, also declared national disasters in Indonesia (Dennis et al. 2005)). Because of further burning, these areas have exhibited limited natural regeneration, showing high levels of cover by invasive grass species, and are unlikely to regain significant quantities of forest cover or biomass without active restoration (Kartawinata 1993). Active restoration was not considered in these analyses (i.e. we assumed no carbon benefits from protection of lands that currently have no forest cover).

RIL was assumed to result in a reduction of 30% of above ground carbon, relative to intact forest, and CL a reduction of 60% (Carlson et al. 2012), relative to intact forest. CL was also assumed to emit approximately 347.5 Mg CO_2 ha^{-1} if on peat soils due to soil disturbance (Hergoualc'h & Verchot 2013). Plantations (for industrial timber or oil-palm) were assigned no net change when planted on non-forest areas (0 Mg CO_2 ha^{-1}), because the carbon sequestered in industrial timber and oil-palm plantations is ultimately released when trees are harvested. For the "other non-forest" land-use class, we assumed worst case carbon emissions (i.e. that of oil-palm).

A.3.5 Variations

We determined if the impact of alternative interpretations of public policy targets on the results, along with the impact of variations in opportunity costs (Table A.8). Whilst the main analyses attempted to conserve all the remaining distribution of orangutan, we also considered the impact of preserving only the patches that were considered to be viable. Viable orangutan populations were determined by calculating their density in each $1km^2$ grid cell via expert elicitation, then grouping grid cells of breeding population presence into contiguous patches (approx. 2000 patches) (Wich et al. 2012). Any of these contiguous patches that contained fewer than 250 individuals were removed, as this is considered to be the minimum viable population size for orangutan in areas with low hunting pressure (Marshall et al. 2009). We also varied the definition of 'forest' cover, as this was not clearly specified in state government policy documents. The strict forest cover target could be met by the intact, logged or mangrove forest cover classes. The moderate and broad forest cover targets could additionally be met by the agroforest/regrowth forest class, and severely degraded logged forest could also contribute to the broad forest cover target.

We also considered the impact of assumptions about the discount rate, along with profits from oil-palm, industrial timber plantations, conventional logging, and reduced impact logging. We did not consider the impact of changes to once-off administrative costs or protected area management costs, as these were insignificant relative to the opportunity cost of oil-palm production. We varied the profits for oil-palm plantations, ITP, CL and RIL by ±50% for each land-use separately and all together (Table A.8). The upper estimate for oil-palm plantations was increased by 55%, to incorporate the previous peak in the fluctuations in the price of crude palm oil. We also applied a variation where the oil-palm profits in Kalimantan and Sarawak matched that of Sabah, to represent a case where the management practices, environmental conditions and infrastructure is consistent across states. The cutting cycle length for both types of logging were altered by ±10 years and incorporated in the upper and lower estimates (i.e. the lower logging estimate represents a 50% reduction in the profit per hectare harvested and a cutting cycle length of 40 years, whilst the upper logging estimate represents a 50% increase in the profit per hectare harvested and a cutting cycle length of 20 years). We varied the discount rate (of 10%) by ± 5% in absence of other variations and together with the extremes of variations in profits (Table A.8).

A.3.6 Classification Uncertainty

To visualise the spatial uncertainty in zone allocation, we calculated the classification uncertainty (adapted from Levin et al. (2013)):

$$U_i = \left(1 - \left[\frac{\frac{M_i}{S_i} - \frac{1}{n}}{1 - \frac{1}{n}}\right]\right)$$

Where U_i is the classification uncertainty for planning unit i; M_i is the maximum set membership (the greatest number of times the planning unit was allocated to a particular zone) for planning unit i; n is the total number of zones (in this case 6); and S_i is the total number of runs. In this case the total number of runs was 21,000 (i.e. the number of parameter variations for each scenario (21), multiplied by the number of runs per solution (1000)). Planning units that had been allocated to each zone an equal number of times (across all the parameter variations and repetitions) would receive a value of 1, whereas planning units that had been allocated to only one zone were given a value of zero. This enabled a spatial depiction of the uncertainty, or variability, in the land use allocations for each scenario.

A.4 RESULTS

A.4.1 Protecting the mountainous interior of Borneo

The aspirations of the highest profile conservation initiative in Borneo (the Heart of Borneo) are reflected in scenario 3, with coordinated efforts focused on the mountainous and heavily forested interior of Borneo, and state-based planning outside of this core region (Figure A.2a and Figure A.2c). This scenario incurs the greatest opportunity cost for meeting the policy targets, as 51% of land on Borneo would be required for protection or reduced-impact logging (Figure A.3a and Figure A.4). Whilst large tracts of land remain forested under this scenario, much of the lowland habitat for orangutan and elephant is converted to non-forest use, as these areas fall outside of the core region and existing protected areas (Figure A.1 and Figure A.2). Despite these limitations, this scenario substantially improves upon conservation targets relative to the baseline scenario (scenario 1), which could result in only 25% of land protected or managed for reduced-impact logging and the remainder being converted to non-forest use or conventional forestry (Figure A.4b).

A.4.2 Integrated planning achieves targets more efficiently

Integrated planning both within individual states and across jurisdictional borders could enable substantial savings while meeting targets across diverse sectors. If states coordinated their plans and allowed more flexible changes to existing land-use allocations (scenario 4a), this would offer an opportunity cost saving of at least US$43 billion with the same level of target achievement as other scenarios Figure A.3b), or, for a similar opportunity cost, would enable substantially higher achievement of all targets (Figure A.5). Additionally, integrated planning was the closest to meeting conservation targets while requiring less land for protected areas, and delivering the greatest area of reduced-impact logging (Figure A.5 and Figure A.4b).

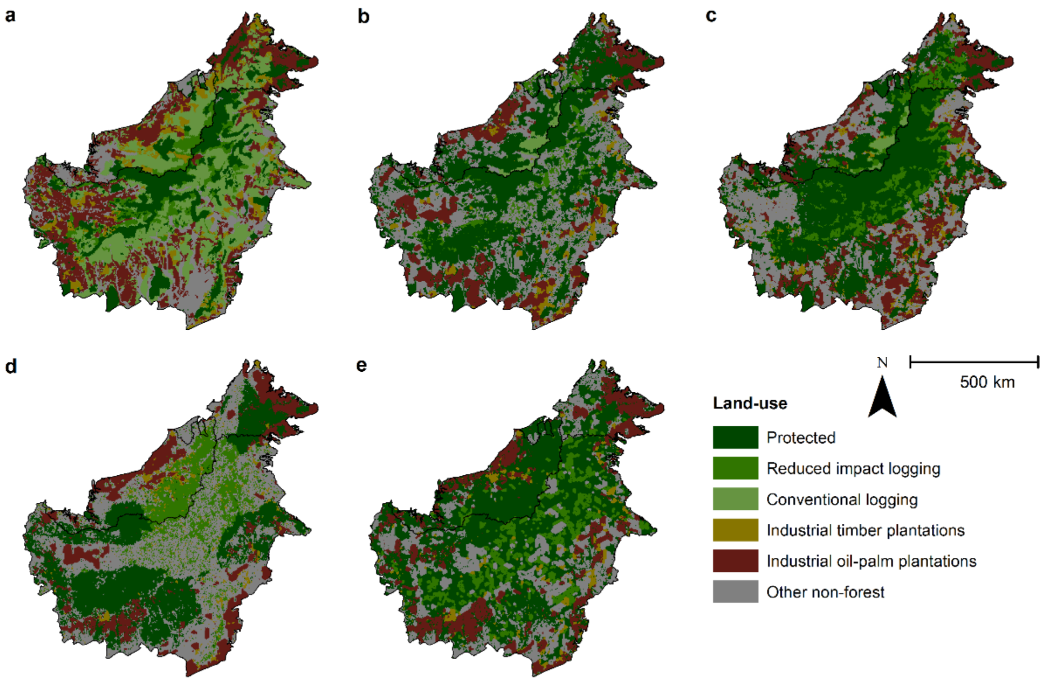

Figure A.2 | Future land-use options under each scenario: (a) baseline (scenario 1); (b) state-based planning (scenario 2); (c) coordinated planning within the mountainous core, with state-based planning outside (scenario 3); (d) integrated planning with existing state targets (scenario 4a); and (e) integrated planning with alternative public policy targets for biodiversity (scenario 4b).

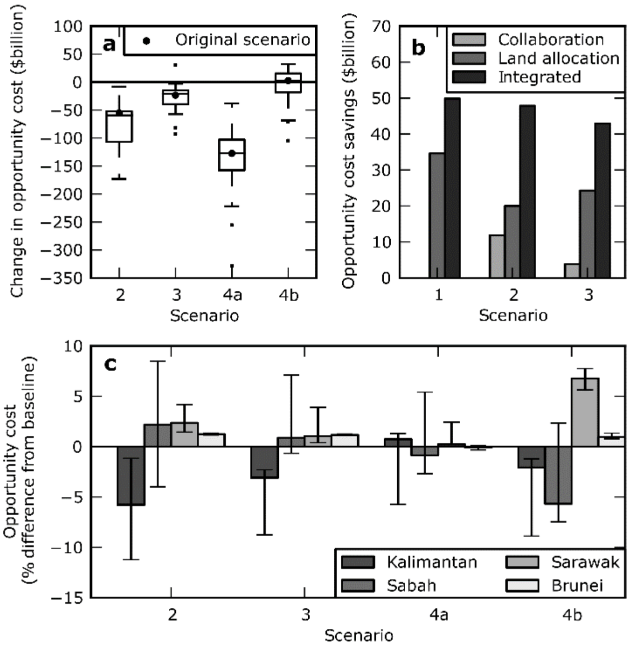

Figure A.3 | Changes in opportunity costs under the alternative planning scenarios. (a) Comparing opportunity costs relative to the baseline (scenario 1), integrated planning (scenario 4a) resulted in the lowest opportunity cost, whereas extending the conservation targets (scenario 4b) was the most expensive. Box plots show the variation in opportunity costs when altering the economic parameters and assumptions about public policy targets. While this variation was considerable, it affected all scenarios similarly, such that integrated planning had the lowest opportunity cost for any given set of parameters and assumptions. (b) Exploring the effects of coordination and/or allowing more flexible changes to existing land allocations on the opportunity cost for scenarios 1, 2 and 3. Savings are expressed relative to the opportunity cost of each scenario when it is implemented without full coordination, and allowing fewer changes to the existing land allocation. (c) The distribution of opportunity cost among states differed in each scenario, compared to the baseline case (scenario 1). Although each state's opportunity cost differed by a maximum of +/- 7% between scenarios, this is still likely to create challenges for collaborative efforts. The error bars represent the minimum and maximum opportunity cost change when altering the economic parameters and assumptions about public policy targets.

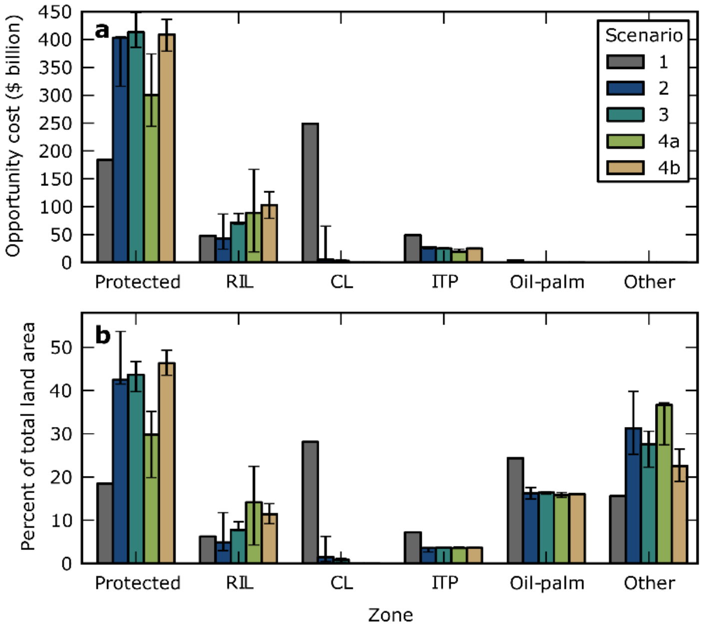

Figure A.4 | Allocation of land-uses across scenarios. (a) The contribution of each land-use zone to the opportunity cost. (b) The percent of total land area allocated to each land-use under alternative scenarios. CL and RIL refer to conventional logging and reduced impact logging respectively. ITP refers to industrial timber plantations. Solid bars represent the result from each scenario, and the error bars represent the minimum and maximum when altering the economic parameters and assumptions about public policy targets. The baseline (scenario 1) shows no variation, as it assessed the existing land-use allocations.

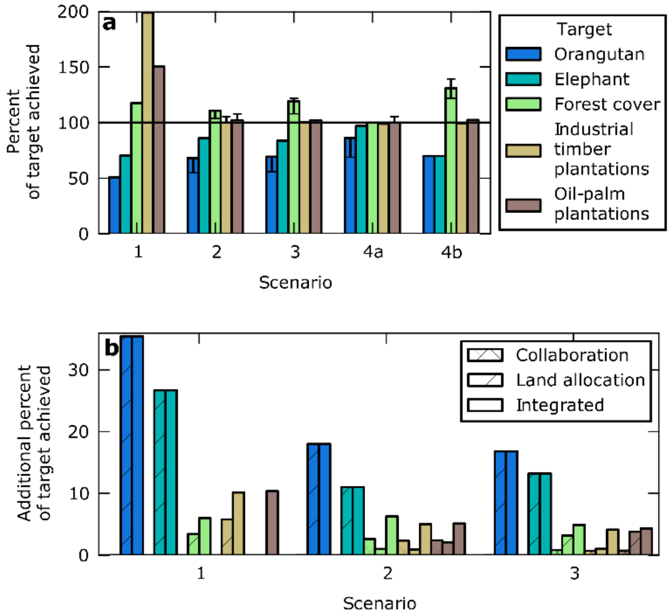

Figure A.5 | Variation between scenarios in terms of their achievement of public policy targets. (a) All scenarios achieved the economic targets (i.e. industrial timber plantations and oil-palm plantations), but no scenarios achieved the species conservation targets. Integrated planning (scenario 4a) performed the best in terms of minimising the overall target shortfall. The target for protected areas is not shown, because the target of 17% by land area was met in the baseline scenario, and was greatly exceeded in scenarios 2, 3 and 4 due to the orangutan and elephant habitat requirements. The error bars represent the minimum and maximum change in target achievement when altering the economic parameters and assumptions about public policy targets. (b) More of the species conservation targets can be achieved when planning involves coordination between Bornean states, and/or allowing more flexible changes to existing land allocations. Allowing more flexible changes to existing land allocations resulted in substantial gains for species conservation targets because much of the orangutan and elephant habitat overlaps with unplanted concessions for industrial timber or oil-palm. Allowing these areas to become protected or logged forests dramatically increases the scope for achieving the targets for these threatened species.

A shift away from state- or species-focused approaches to a more collaborative, ecosystem-based approach could deliver substantial dividends for climate change mitigation and for biodiversity conservation. Integrated planning reduces CO_2 emissions from land-use change relative to the baseline, and out-performs other scenarios if the forest cover target is modified from a target for total forest cover (regardless of forest type), to a target of conserving 70% of the remaining extent of each forest type (scenario 4b, Figure A.6). With a 'total forest' target (scenario 4a), protected areas are concentrated within the remaining extent of orangutan and elephant distributions, with limited protection of upland forests (Figure A.1, Figure A.2, and Figure A.7), and emissions reductions are ~16%. In contrast, if forest cover targets require conservation of each forest type (scenario 4b), then it is possible to achieve a 53% reduction in emissions compared to the baseline (Figure A.6). This scenario therefore offers emissions reductions that are substantially higher (53% vs. 40%) than would be possible if protection was concentrated in the mountainous core of the island (scenario 3), even though opportunity costs remain similar.

A.4.3 Integrated planning requires some reassignment of land-uses

Our alternative futures reveal that public policy targets can be more efficiently achieved through coordination and modifications to existing land-use allocations. Integrated planning across Borneo (scenario 4a) could require protection of 8.6 million hectares of land that is currently designated for logging (with or without an existing concession), along with 4.3 million hectares of un-planted oil-palm concessions and 1.3 million hectares of un-planted industrial timber concessions (Figure A.10). Despite this substantial re-allocation of land-uses, the opportunity costs to each state remained similar to the baseline scenario (each state's opportunity costs differed by a maximum of ±7% across all scenarios; Figure A.3c). Nonetheless, even small differences in opportunity costs may create challenges for collaboration. There are also some substantial differences across states in the land allocations required to meet targets (even if total opportunity costs are similar). For example, in scenario 4b, the extent of protected areas is increased by 58% (compared to baseline) in Sarawak, compared to 20% in Kalimantan and 14% in Sabah, which partly reflects their existing protected area estate, and differences across states in opportunity costs of logging and plantations (Figure A.11).

The allocation of land-uses within each of the scenarios changed with variation in parameter values and multiple model runs (Figure A.8). Whilst the spatial allocation of protected areas and RIL varied only slightly (reflecting the limited spatial ranges and habitat requirements of orangutan and elephant), the allocation of the other land-uses was relatively flexible, reflecting the much greater availability of land suitable for oil-palm and ITP. This flexibility in the allocation of land to oil-palm and ITP means that the land-use scenarios presented here (Figure A.2) could be adjusted to accommodate local needs without compromising overall economic targets.

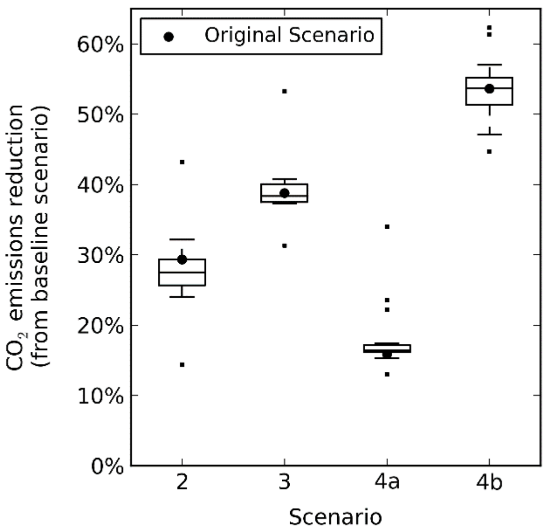

Figure A.6 | The percentage of CO_2 emissions reduction from the baseline scenario. The variations from the original scenarios were obtained by altering the economic parameters and assumptions about public policy targets.

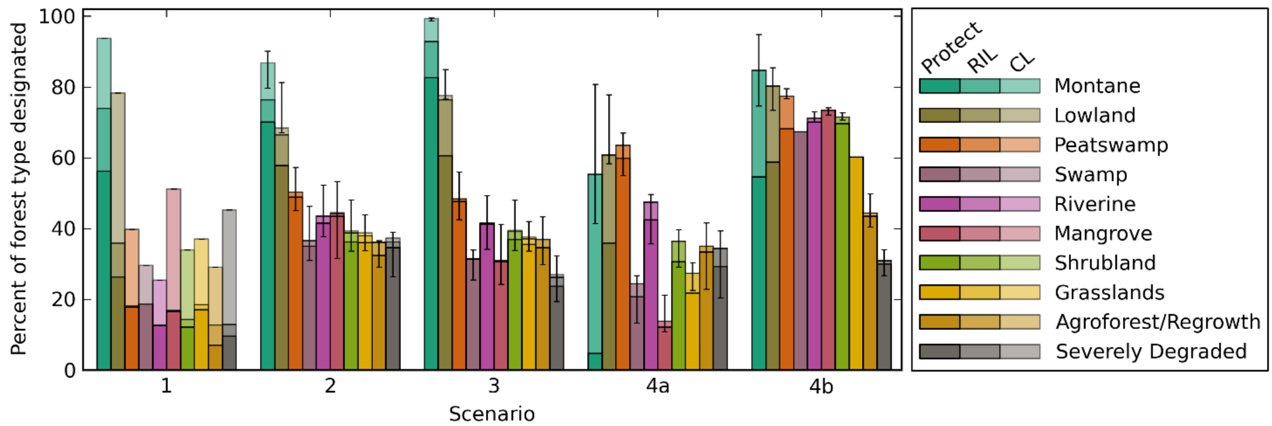

Figure A.7 | Representation of individual forest types. This shows the percentage of the extent of each forest type that is designated for protection, reduced impact logging (RIL), or conventional logging (CL). While all scenarios have a general 'forest cover' target, this will not ensure representation of each forest type. Under scenarios 1 to 3, protected areas are concentrated in the montane forest type. Scenario 4b specifically targets each forest type individually and consequently has the most equitable representation. Forest types and extents were defined by Miettinen et al (2012) for the year 2010. Error bars represent the minimum and maximum values when altering the economic parameters and assumptions about public policy targets.

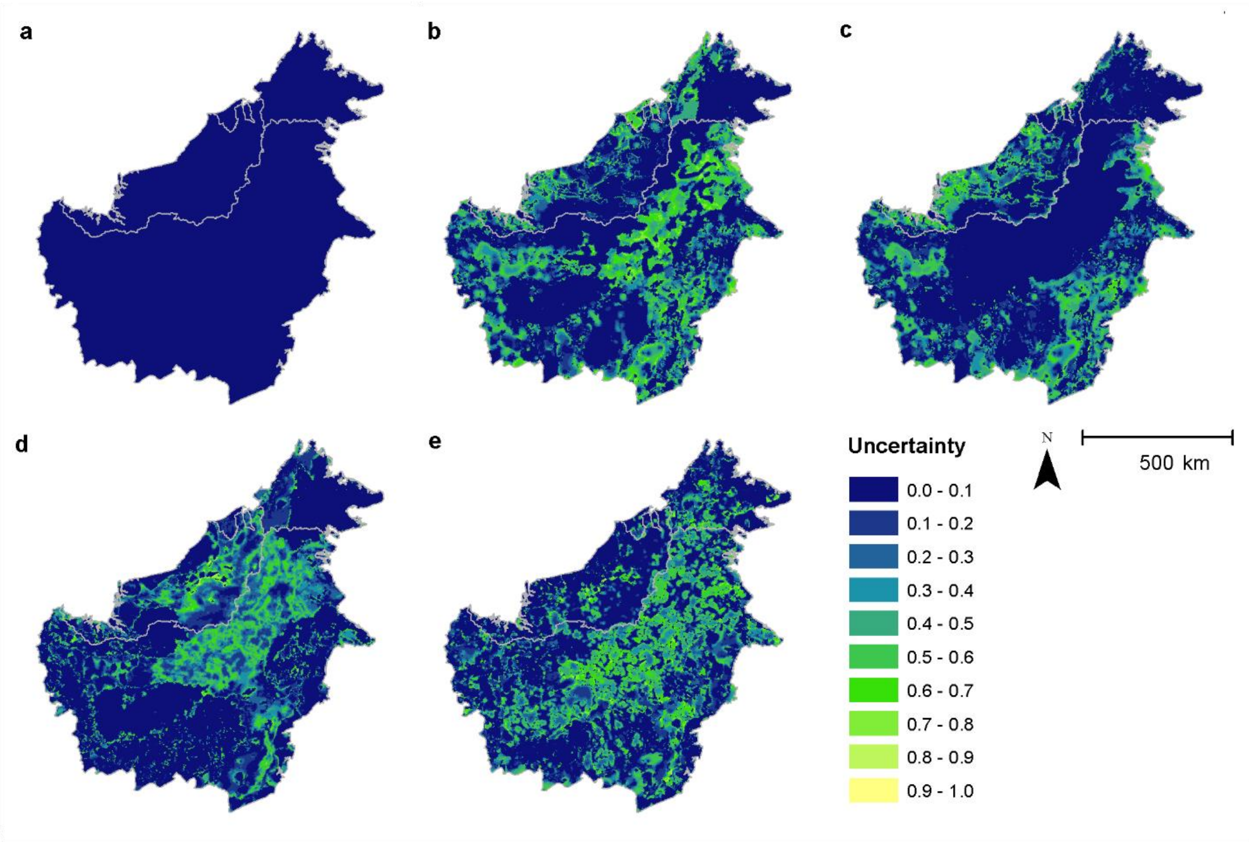

Figure A.8 | The classification uncertainty under each scenario. (a) baseline (scenario 1); (b) state-based planning (scenario 2); (c) coordinated planning within the mountainous core, with state-based planning outside (scenario 3); (d) integrated planning with existing state targets (scenario 4a); and (e) integrated planning with alternative public policy targets for biodiversity (scenario 4b). This shows the uncertainty of allocating a planning unit to the final land-use zone. This is a combination of the classification uncertainty from multiple runs with the same input parameters, along with the variation in input parameters. There is no uncertainty surrounding zoning in scenario 1, as this scenario is based on implementing the existing land-use allocations.

A.5 DISCUSSION

Integrated land-use planning has the potential to achieve a wide range of targets in a cost-effective manner, but the effectiveness of any planning process also depends critically on the adequacy of public policy targets. For example, the integrated planning scenario (scenario 4a) would cost-effectively make progress towards the stated species conservation targets (Figure A.3a), but the allocation of protected areas would be biased toward habitat favoured by orangutan and elephant (Figure A.1c and d, Figure A.2d) and potentially at the expense of other species or the livelihoods of local people (Abram et al. 2014). Whilst ignoring existing targets could lead to substantial savings (Figure A.12), it could result in poor conservation outcomes (Figure A.13). In contrast, if targets existed for each major vegetation type (scenario 4b) then greater geographic representation of the various habitats would be ensured (Figure A.7), and this would also substantially enhance opportunities to reduce CO_2 emissions from land-use change (Figure A.6). To facilitate integrated planning, Borneo-wide targets would need to be fully backed by all of the governments of Borneo, be developed in the context of other aligned or potentially conflicting goals, and respect political and economic sovereignty. This issue is not unique to Borneo – developing quantifiable targets to achieve ecologically sustainable development is a global challenge (Maxwell et al. 2015).

Given the vast spatial extent of Borneo and the multitude of factors included in this analysis, we acknowledge that the data and assumptions will not capture local variation and nuances, particularly in relation to opportunity costs. We have not, for example, accounted for the potential that one land-use type might have a greater rate of change in profitability over time, or that the spatially explicit probability of conversion might change over time. Furthermore, a fully functioning market for carbon would likely reduce the relative opportunity costs of the scenarios that offer higher emissions reductions. However, we found that large variations in input parameters (including alternative interpretations of public policy targets) would not change the overall conclusions (Table A.9). We have also not attempted to analyse all potential futures, but rather we reveal the possible outcomes of an illustrative set of planning options.

We found that changing the status of unplanted oil-palm and industrial timber concessions will be vital for making progress towards species conservation targets (Figure A.5). We acknowledge that re-allocating undeveloped land would not be trivial, and will require a thorough evaluation of tenure and governance arrangements in all stages of the planning process (McCarthy & Cramb 2009). Careful consideration of the appropriate institutional and incentive structures will be vital and require consultation beyond state and intergovernmental bodies to include the business sector, local communities, and the wider public. To realise conservation and economic goals on the ground, institutional arrangements would also need to ensure that incentives reach key actors at a district or local level (Ardiansyah & Jotzo 2013).

Implementing an integrated planning approach (scenario 4a and 4b) requires both new protected areas to be designated and managed, and also for some existing protected areas to be reallocated to other land uses (Figure A.10). This process of protected area downgrading, downsizing, or degazettement (PADDD) may risk undermining the perceived permanence of other protected areas (Forrest et al. 2015). Despite this issue, PADDD may be an essential part of land-use planning reform and substantial efficiency gains and improved biodiversity outcomes could be achieved by re-allocating underperforming protected areas (Fuller et al. 2010). Globally, protected areas are biased towards areas that have limited development potential (such as remote areas, or those with steep slopes or high elevation) (Joppa & Pfaff 2009). This is also true on Borneo, where protected areas are concentrated in the mountainous interior, resulting in a biased representation of forest types (i.e. montane forests above all other types, Figure A.7). In other locations the effectiveness of protected areas is reduced by surrounding land uses (Gaveau et al. 2014). Laurance et al. (2012) found half of protected areas in the world's tropical forests are ineffectively managed, resulting in a loss of biodiversity – a process that was strongly influenced by the surrounding landscape. Reallocating protected areas within the context of whole-landscape land-use planning may outweigh the risks associated with PADDD. However, a broader range of conservation targets must be developed and assessed before determining the optimal allocation of protected areas.

The capacity to effectively implement public policy targets varies significantly among the geopolitical units of Borneo (World Bank Group 2013). Trans-national coordination would need to overcome constraints related to governance efficacy, efficiency, regulatory quality, sovereignty commitments, and control of corruption. Furthermore, the history of cooperation between Brunei Darussalam, Malaysia, and Indonesia has involved significant challenges (Colchester 1993; Sparke et al. 2004). Substantial complexity is added by sectorial control of different land-use types (e.g. forestry, agriculture, and mining), the related political territoriality, and by varying social acceptability of land-use changes (Meijaard et al. 2013). A socially equitable distribution of land-use might be well received by local communities, but deriving such a land-use plan will require quantification of institutional and individual costs and constraints not yet captured in our analysis. Innovative mechanisms, such as land swaps and payments for conservation or opportunities foregone between geopolitical units (states, provinces, districts) may be required for the direct and indirect benefits of integrated planning to be realised (Drechsler et al. 2010).

Our results confirm that there is a strong justification for expanding upon existing efforts for collaboration across the political borders of Borneo. This finding is in line with Kremen et al. (2000), who found that operating at the national scale was ineffective in achieving conservation outcomes. Our study has demonstrated that restricting coordination to within the mountainous interior (i.e. the Heart of Borneo, scenario 3) fails to realise the benefits of wider coordination and will not meet public policy targets. Whilst the Heart of Borneo initiative reflects the sentiment of coordinated planning, stronger and more geographically distributed efforts are needed to avoid irreversible biodiversity loss, achieve equitable benefits among diverse stakeholders, and maximise efficiency across multiple sectors. A binding agreement on land-use may be necessary to ensure that jointly developed plans are implemented in each national jurisdiction. Such an agreement could be facilitated by a regional intergovernmental platform (such as ASEAN [The Association of Southeast Asian Nations], the tri-national collaboration regarding the Heart of Borneo, or BIMP-EAGA

[Brunei Darussalam-Indonesia-Malaysia-Philippines East ASEAN Growth Area]) and should serve to give each jurisdiction the confidence that their interests are being treated equitably. The agreement could include joint targets for sustainable management of forests, facilitate technical exchange on how to achieve these targets, bring cross-border protected areas under joint management, and address cross-border trade and flow of labour. Whilst designing such an agreement will involve many challenges, a non-binding agreement risks weak implementation and the adverse environmental impacts from poorly regulated agricultural expansion and extractive industries (Harrop & Pritchard 2011).

Our study is based on the fundamental assumption that governments seek to achieve their stated public policy targets, and that all targets are weighted equally. The reality, however, is that there will be far greater governmental support for increasing profits from oil-palm and other lucrative activities, as opposed to meeting conservation targets (e.g. the Indonesian government's target to stabilise all wild orangutan populations by 2017) (Meijaard & Sheil 2008). This situation is reinforced by the close and well-protected ties between industry (e.g. oil-palm, forestry, mining etc.), and politicians (Leuz & Oberholzer-Gee 2006; Dieleman & Boddewyn 2011); the intertwining relationships between, rather than independence of, the executive, legislative and judicial branches of government (Romano 2003); and corruption in both Indonesia and Malaysia (Siddiquee 2009; Butt 2011). Opposing these barriers, however, are potentially powerful democratic forces, such as the growth of local non-government organisations and the relative freedom of speech and information, especially in Indonesia (Blunt et al. 2012). Access to information is an important pre-cursor to change in political and civil society, including the potential for policy reform and implementation of innovative solutions (Romano 2003).

All countries on Borneo are struggling to develop and implement strategies that achieve sustainability despite their stated commitments to green growth and sustainable development. For example, the Sabah government has committed to certifying all its remaining natural forest timber concessions under the criteria of the Forest Stewardship Council or the Malaysian Timber Certification Council (Table A.3, Table A.4). However, the over-logged forests in Sabah raise limited net revenue, requiring that operations be scaled back until forests have sufficiently recovered to once again produce commercial timber (Reynolds et al. 2011). Alternatively, authorities could potentially generate income from avoided deforestation (requiring the development of a regulatory framework that aligns with international criteria for carbon trade), or from intensification of plantation production. The latter would require new spatial plans that allow plantation development within commercial forest reserves, along with stringent safeguards to minimise impacts on other targets (e.g., targets included in the State action plans for elephant, and orangutan, the Sabah Biodiversity Strategy (2012-2022), Sabah Tourism Masterplan (2011-2025), and the Sabah Structural Plan (2013-2033)). It may also be necessary to alter existing legislation which can require landholders to clear any forest on their land within a specified time period (usually three years) (State of Sabah 2010). Certification through the Roundtable on Sustainable Palm Oil (RSPO) has the potential to minimise adverse environmental impacts from oil-palm expansion, but significant high-level reforms to its monitoring, enforcement and auditing processes are needed for this to be an effective option (Laurance et al. 2010). Obstacles such as these will need to be overcome before the benefits of land-use policy reform can be realised.

New mechanisms are required to ensure effective implementation of the targets evaluated here. In some districts, for example, targets for watershed management or wildlife conservation will require new or expanded protected areas. Under such circumstances, a payment scheme to reward districts (or states or countries) for delivering these goods and services may incentivise protection. Payments for environmental services schemes have been piloted in Indonesia (Fauzi & Anna 2013) but have primarily been initiated by private enterprise. A regulatory framework to facilitate payments between districts is being drafted under the government regulation on environmental management, but is still awaiting endorsement (Prasetyo et al. 2009). A broader regulatory and institutional framework that encompasses such schemes and new market-based mechanisms will be essential to deliver effective land-use planning and land management.

The potential benefits from integrated planning within and between countries are not unique to the island of Borneo; many other jurisdictions across the globe have committed to land-use allocations that are proving sub-optimal. For example, Australia has devoted over half of its land mass to low productivity pastoralism with inflexible leasehold arrangements (Hamblin 2009), and China's farmland protection policy has led to a clustering of incompatible land-uses (Lichtenberg & Ding 2008). Trans-national collaboration may also be beneficial in the Congo Basin - a globally significant forest area spanning six central African countries with varying deforestation rates, with competing potential uses of the forest area (Somorin et al. 2012). Such an approach will also be instrumental in conserving the habitat of migratory species, such as the American redstart (*Setophaga ruticilla*) (Martin et al. 2007), and also where species ranges span national borders, such as larger bodied mammals in the Albertine Rift, Africa (Plumptre et al. 2007).

Achieving the Millennium Development Goals and post-2015 Sustainable Development Goals will require innovative solutions to complex land-use planning and policy problems (United Nations

2012). An analysis of alternative futures can help visualise the outcomes of different approaches. The Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) will also employ scenarios to address multi-scaled policy problems that encompass the natural and social sciences (Perrings et al. 2011). Through evaluation of alternative futures we found that coordination between countries would enhance the efficiency of achieving a diverse suite of national and international policy targets, which will be relevant wherever biodiversity and industries extend across borders. Integrated planning also improves efficiency when there is variation within and between countries in the costs and opportunities for implementing policy (Fuller et al. 2010). An alternative future for the tropical forests of Borneo that captures the benefits of coordination and integrated planning could enhance both conservation and economic outcomes.

A.6 SUPPORTING INFORMATION

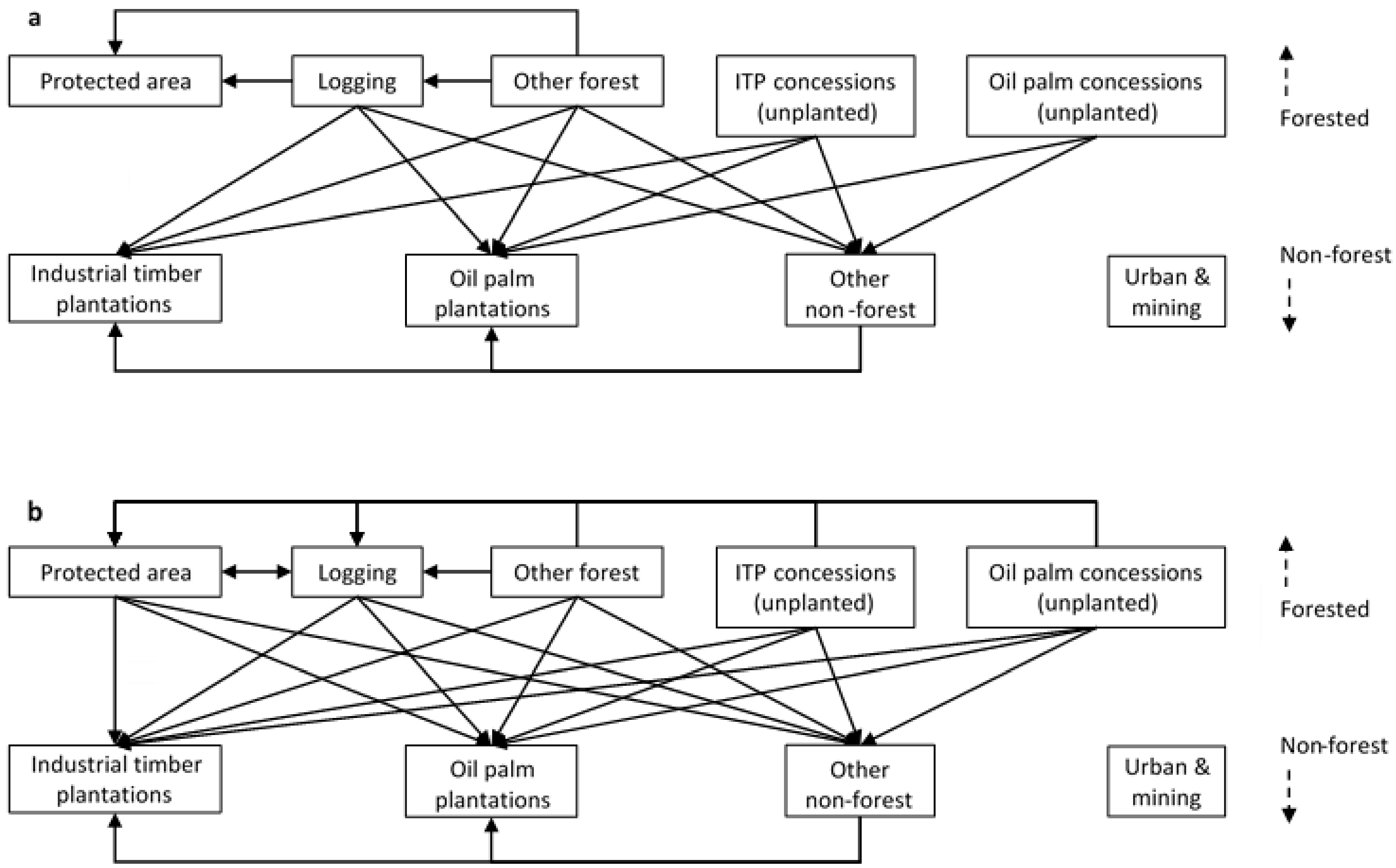

Figure A.9 | Possible land-use transitions for scenarios 2 (panel a) and 4 (panel b). Arrows show the changes in land-use allocation that are possible under each scenario (and whether uni-or bi-directional). Urban and Mining lands are not changeable, and so have no connecting arrows.

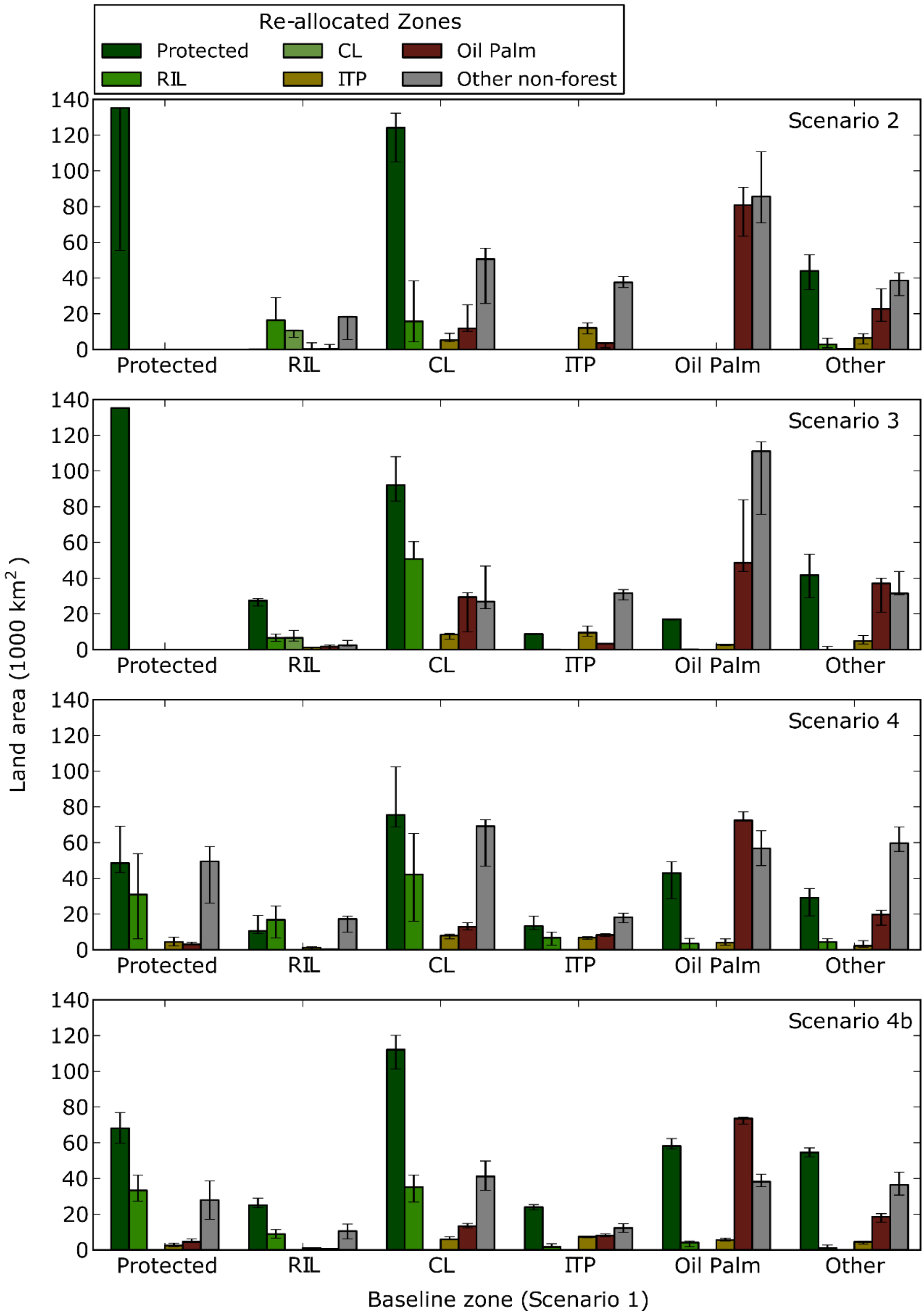

Figure A.10 | The re-allocation of land-use under different scenarios. Scenario 1 represents the existing land-use plan so was used as the baseline. Notably, there is a reallocation of protected areas in the integrated planning scenarios (4a and b). The error bars represent the minimum and maximum values when altering the economic parameters and assumptions about public policy targets.

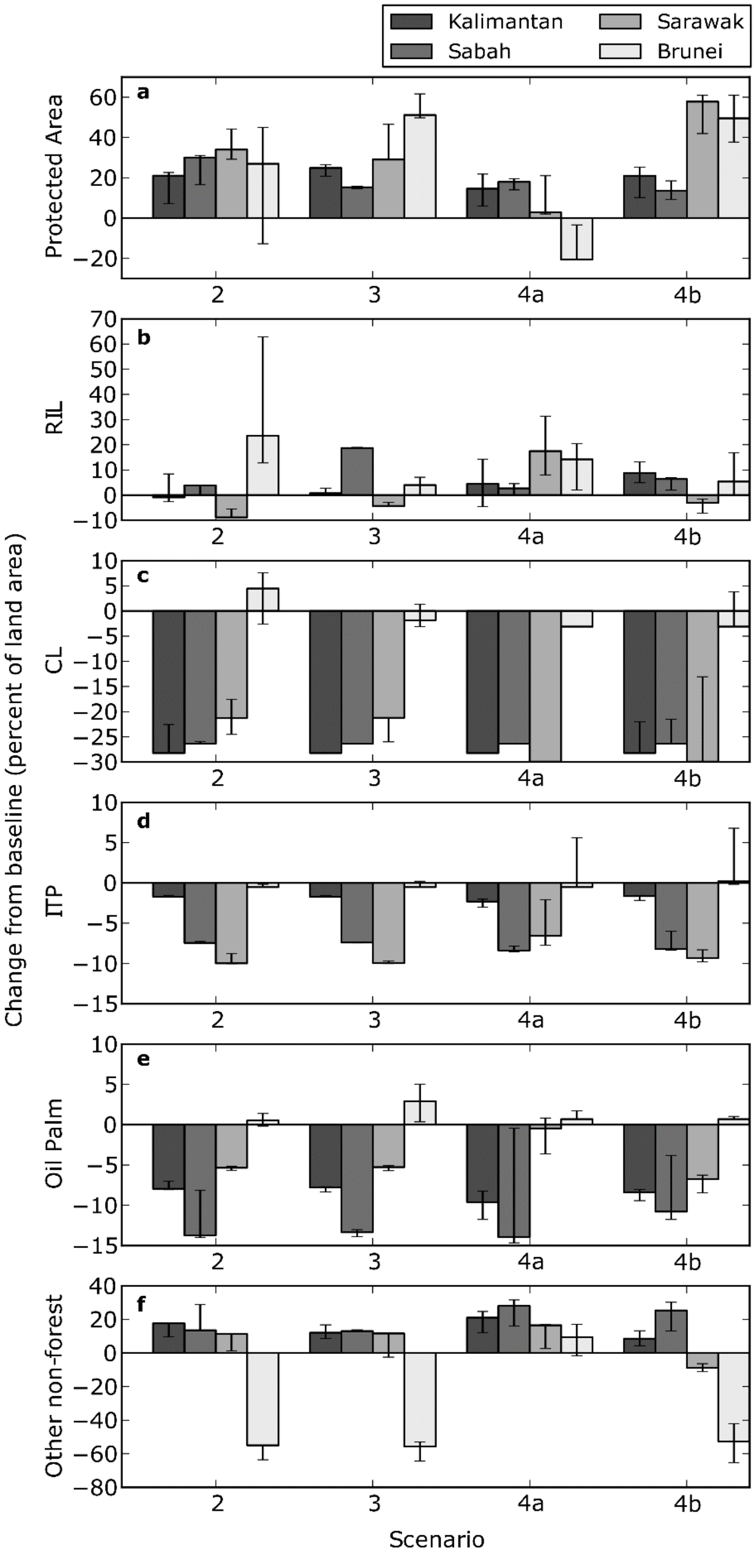

Figure A.11 | The change in the distribution of land-use zones across Bornean states when compared to the baseline scenario. This is shown for: (a) protected areas, (b) reduced impact logging (RIL), (c) conventional logging (CL), (d) industrial timber plantations (ITP), (e) oil-palm plantations, and (f) other non-forested land-uses. Error bars represent the minimum and maximum values when altering the economic parameters and assumptions about public policy targets.

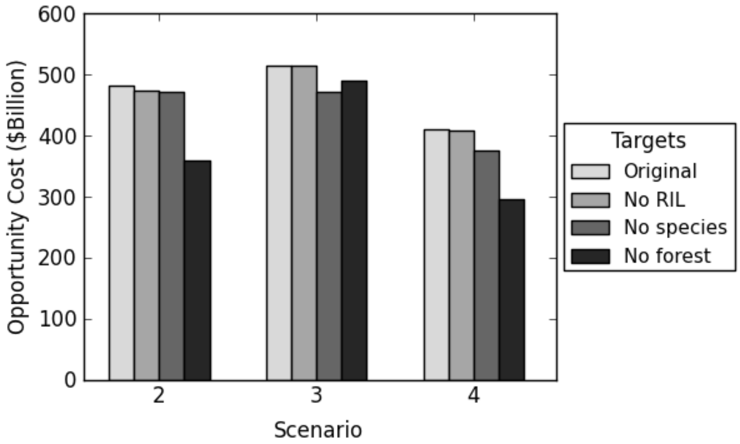

Figure A.12 | The opportunity costs across scenarios when omitting targets for reduced impact logging (RIL), species (orangutan and elephant), and forest area. Removing the requirement for RIL had only a minor reduction in the opportunity cost for each scenario, whereas removing the species or forest targets resulted in larger opportunity cost savings. Scenario 1 was not included as the land-use allocation cannot be altered, therefore changing the targets does not have an impact. Scenario 4b was also excluded, as this scenario was already a variation on the targets in Scenario 4a.

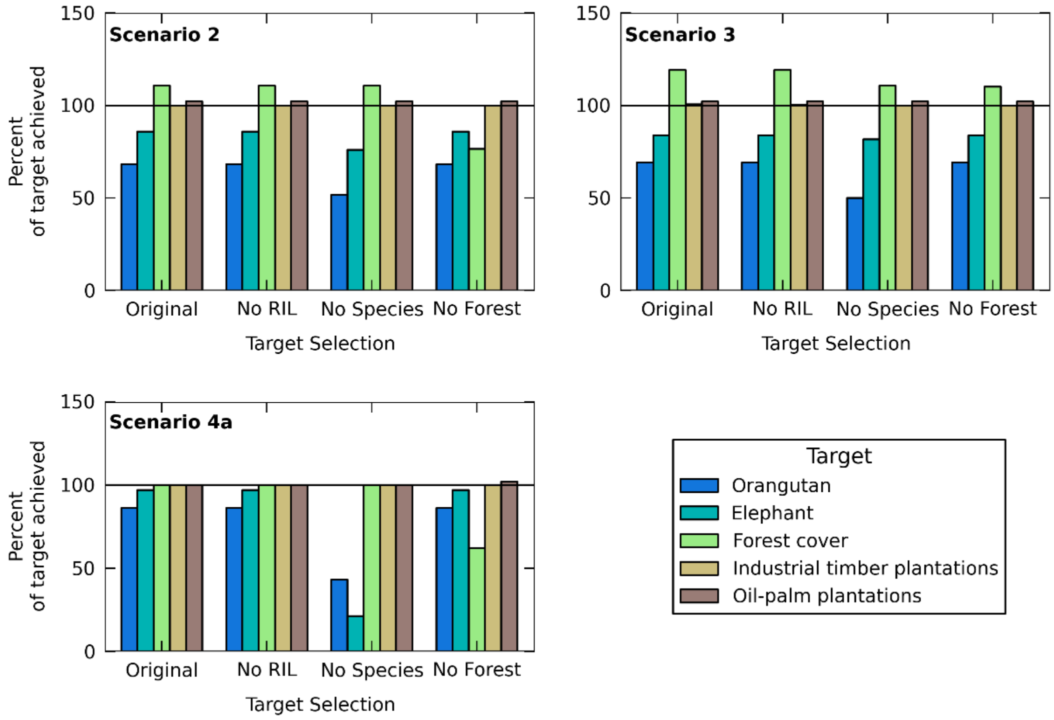

Figure A.13 | Target achievement across scenarios when omitting targets for reduced impact logging (RIL), species (orangutan and elephant), and forest area. Removing the requirement for RIL had only a negligible reduction in the target achievement for each scenario, whereas removing the species or forest targets resulted in poor conservation outcomes. Scenario 1 was not included as the land-use allocation cannot be altered, therefore change the targets does not have an impact. Scenario 4b was also excluded, as this scenario was already a variation on the targets in Scenario 4a.

Table A.4 | Sources used to derive the public policy targets. In some cases relied on the reporting of targets in the media due to the inaccessibility of government documents.

Target	*Sabah, Malaysia*	*Sarawak, Malaysia*	*Kalimantan, Indonesia*	*Brunei Darussalam*
Forest cover	In 1992 Malaysia pledged 50% forest cover for the country at the Rio Earth Summit	In 1992 Malaysia pledged 50% forest cover for the country at the Rio Earth Summit	Declared by the Indonesian President (President of the Republic of Indonesia 2012)	Declared by the Government of Brunei Darussalam (Government of Brunei Darussalam 2008)
Protected areas	From the Convention on Biological Diversity (CBD 2010)	From the Convention on Biological Diversity (CBD 2010)	From the Convention on Biological Diversity (CBD 2010)	Declared by the Forestry Department (Government of Brunei Darussalam Forestry Department 1989)
Orangutan	From Sabah’s Orangutan Action Plan (Sabah Wildlife Department 2011a)	From Sarawak’s Orangutan Strategic Action Plan (Gumal & Tisen 2010)	Declared by the Ministry of Forestry (Soehartono et al. 2007)	N/A
Elephant	From Sabah’s Elephant Action Plan (Sabah Wildlife Department 2011b)	N/A	None	N/A
Reduced impact logging	Forestry director’s message (Mannan 2012)	N/A	Declared by the Minister of Forestry and the Indonesian President (Ministry of Forestry of the Republic of Indonesia 2002; President of the Republic of Indonesia 2007)	Declared by the National Forestry Policy of Brunei Darussalam (Brunei Forestry Department 2012)
Oil-palm plantations	The Sabah Development Corridor Project states that up to 2.1 million ha of land in Sabah could be converted to agriculture (Sabah Economic Development and Investment Authority 2008)	Media report (“Oil palm acreage target achievable” 2012)	Media report (Bahroeny 2009; Gilbert 2012)	None
Industrial timber plantations	Sabah’s proportion of Malaysia's target of 375,000 ha by 2020 (Malaysian Timber Industry Board 2009)	Sarawak’s proportion of Malaysia's target of 375,000 ha by 2020 (Malaysian Timber Industry Board 2009)	Kalimantan’s proportion of Indonesia's target of 3.6 million new hectares (Obidzinski & Dermawan 2010)	None

Table A.5 | The contribution of each land-use zone towards each target. CL and RIL refer to conventional logging and reduced impact logging respectively. ITP refers to industrial timber plantations.

Target / *Zone*	*Orangutan*	*Elephant*	*Forest cover*	*Protected Area*	*ITP*	*Oil-palm*
Protected	1	1	1	1	0	0
RIL	0.8	0.8	1	0	0	0
CL	0.7	0.7	1	0	0	0
ITP	0	0	0	0	1	0
Oil-palm	0	0	0	0	0	1
Other	0	0	0	0	0	0

Table A.6 | Oil-palm suitability and net present value (NPV). Oil-palm suitability (a) was estimated by classifying a variety of biophysical properties of land units into suitability classes for oil-palm production. The net present value of oil-palm (b) is separated by state and yield (MPOB 2009, 2012; Direktorat Jenderal Perkebunan 2012). Whilst Brunei has the biophysical capacity for oil-palm, it does not currently have an oil-palm industry, so the NPVs from neighbouring Sarawak were applied. Figures are in 2009US\$ $ha^{-1}yr^{-1}$. Characteristics were quantified using the sources (Applied Agricultural Resources Sdn Bhd 2012; FAO/IIASA/ISRIC/ISSCAS/JRC 2012), unless otherwise stated.

a

Characteristic	***1: Desirable***	***2: Minor limitations***	***3: Serious limitations***	***4: Very serious limitations***	***No data: Not at all suitable***	***Sources***
Slope (degree)	0-12	12-16	16-24	24-45	>45	(Carlson et al. 2012)
Topsoil gravel content (%)	0-5	5-20	20-40	>40	-	
Texture (USDA texture class)	1-8	9-11	12	13	-	
Drainage (class)	4-5	3	2,6,7	1	-	
Rivers (100m buffer)	-	-	-	-	all	(Gingold et al. 2012)
Elevation (m)	< 400	400 - 500	500 - 600	600 - 1000	> 1000 or < 0	(Mantel et al. 2007)
Rainfall (mm/yr)	1,750–6,000	1,250–1,750		> 6,000; <1,250		(Gingold et al. 2012)

b

Suitability class	***Yield***	***Sabah***	***Sarawak***	***Kalimantan***
1	Full yield	25,450	17,038	14,960
2	25% less	17,037	10,547	8,972
3	50% less	8,398	4,245	3,174

Table A.7 | Review of estimated yields, costs, revenues and profits from logging in dipterocarp forests in Borneo. This was estimated for methods of clear-felling (CF), conventional logging (CL) or reduced impact logging (RIL). All values refer to harvested hectares, which excludes the hectares that are not harvested due to slope thresholds and RIL criteria (i.e. within a certain distance of water bodies). The cost estimates include post-landing costs and taxes. Figures are in 2009US$.

		Yield (m^3 ha^{-1})		*Cost m^{-3}*			*Revenue m^{-3}*		*Profit ha^{-1} (Intact)*			*Profit ha^{-1} (Logged)*		
Location	***Year published***	***CL***	***RIL***	***CL***	***RIL***	***Source***		***Source***	***CL***	***RIL***	***CF***	***CL***	***RIL***	***CF***
Sabah	*Mean*	127.75	106.00	60.36	64.20		153.00		11,835	9,230	18,635	3,503	2,732	5,263
Danum – Yayasan, Sabah (Tay et al. 2002)	2002	136.00	106.00	60.36	64.20	(Tay et al. 2002; Fisher et al. 2011a)	153.00	(Fisher et al. 2011a)	12,599	9,413	-	-	-	-
Danum, Sabah (Marsh & Greer 1992; Edwards et al. 2011)	1992	120.00	-	60.36	-	(Fisher et al. 2011a)	153.00	(Fisher et al. 2011a)	11,117	-	-	-	-	-
Sabah (Nicholson 1958; Edwards et al. 2011)	1958	117.00	-	60.36	-	(Fisher et al. 2011a)	153.00	(Fisher et al. 2011a)	10,839	-	-	-	-	-
Sabah (Sim & Nykvist 1991; Edwards et al. 2011)	1991	138.00	-	60.36	-	(Fisher et al. 2011a)	153.00	(Fisher et al. 2011a)	12,784	-	-	-	-	-
Sarawak	*Mean*	43.70	27.80	60.03	63.73		153.00		4,063	2,484	7,782	2,194	1,341	3,718
Upper Baram, Sarawak (Richter 2002; Edwards et al. 2011)	2002	44.50	27.80	58.69	63.73	(Richter 2002; Fisher et al. 2011a)	153.00	(Fisher et al. 2011a)	4,197	2,484	-	-	-	-
Sarawak (Grieser-Johns 1996; Edwards et al. 2011)	1996	90.00	-	60.36	-	(Fisher et al. 2011a)	153.00	(Fisher et al. 2011a)	8,338	-	-	-	-	-
Sarawak (Hutchinson 1987; Edwards et al. 2011)	1987	30.00	-	60.36	-	(Fisher et al. 2011a)	153.00	(Fisher et al. 2011a)	2,779	-	-	-	-	-
Sarawak (Lee 1982; Edwards et al. 2011)	1982	25.00	-	60.36	-	(Fisher et al. 2011a)	153.00	(Fisher et al. 2011a)	2,316	-	-	-	-	-
Sarawak (Mattsson-Marn 1982; Edwards et al. 2011)	1982	29.00	-	60.36	-	(Fisher et al. 2011a)	153.00	(Fisher et al. 2011a)	2,687	-	-	-	-	-
Kalimantan	*Mean*	43.61	47.83	73.87	66.20		122.00		2,100	2,679	4,033	1,134	1,447	1,927
Malinau, East Kalimantan (Dwiprabowo et al. 2002)	2002	52.80	60.90	60.57	59.81	(Dwiprabowo et al. 2002; Ruslandi et al. 2011)	122.00	(Ruslandi et al. 2011)	3,244	3,787	-	-	-	-
P.T. Limbang Ganeca, East Kalimantan (Hinrichs et al. 2002)	2002	48.00	48.00	70.75	72.58	(Hinrichs et al. 2002; Ruslandi et al. 2011)	122.00	(Ruslandi et al. 2011)	2,460	2,372	-	-	-	-
Ketapang, West Kalimantan (Elias 2006)	2006	31.40	34.60	65.66	66.20	(Dwiprabowo et al. 2002; Hinrichs et al. 2002; Ruslandi et al. 2011)	122.00	(Ruslandi et al. 2011)	1,769	1,931	-	-	-	-
East Kalimantan (Muladi 1996)	1996	55.00	-	80.02	-	(Ruslandi et al. 2011)	122.00	(Ruslandi et al. 2011)	2,309	-	-	-	-	-
Central Kalimantan - 3 concessions (Ruslandi et al. 2011)	2011	51.50	-	80.02	-	(Ruslandi et al. 2011)	122.00	(Ruslandi et al. 2011)	2,162	-	-	-	-	-
West Kalimantan - Suka Jaya Makmur (Ruslandi et al. 2011)	2011	31.00	-	80.02	-	(Ruslandi et al. 2011)	122.00	(Ruslandi et al. 2011)	1,301	-	-	-	-	-
East Kalimantan - Balikpapan Forest Industries (Ruslandi et al. 2011)	2011	35.60	-	80.02	-	(Ruslandi et al. 2011)	122.00	(Ruslandi et al. 2011)	1,494	-	-	-	-	-

Table A.8 | Details of which parameters were varied to determine the impact on results. CL and RIL refer to conventional logging and reduced impact logging respectively.

Variation	*Discount rate*	*Oil-palm profit*	*ITP profit*	*CL profit*	*RIL profit*	*Forest cover target*	*Orangutan target*
Original	10%	-	-	-	-	Broad	All
Forest moderate	10%	-	-	-	-	Moderate	All
Forest strict	10%	-	-	-	-	Strict	All
Viable orangutan	10%	-	-	-	-	Broad	Viable
Low profit	10%	-50%	-50%	-50%	-50%	Broad	All
High profit	10%	+55%	+50%	+50%	+50%	Broad	All
Low discount rate	5%	-	-	-	-	Broad	All
High discount rate	15%	-	-	-	-	Broad	All
Oil-palm match Sabah	10%	Sabah	-	-	-	Broad	All
High oil-palm profit	10%	+55%	-	-	-	Broad	All
High timber profit	10%	-	+50%	-	-	Broad	All
High CL profit	10%	-	-	+50%	-	Broad	All
High RIL profit	10%	-	-	-	+50%	Broad	All
Low oil-palm profit	10%	-50%	-	-	-	Broad	All
Low timber profit	10%	-	-50%	-	-	Broad	All
Low CL profit	10%	-	-	-50%	-	Broad	All
Low RIL profit	10%	-	-	-	-50%	Broad	All
Low profit, low discount rate	5%	-50%	-50%	-50%	-50%	Broad	All
Low profit, high discount rate	15%	-50%	-50%	-50%	-50%	Broad	All
High profit, low discount rate	5%	+55%	+50%	+50%	+50%	Broad	All
High profit, high discount rate	15%	+55%	+50%	+50%	+50%	Broad	All

Table A.9 | How the variation in input parameters changed the rankings of scenarios. Scenario are ranked by opportunity cost (1 = lowest opportunity cost). Alternative interpretations of public policy targets were not used for scenario 4b, as this scenario had already altered the public policy targets for conservation. CL and RIL refer to conventional logging and reduced impact logging respectively.

	Scenario 1	***Scenario 2***	***Scenario 3***	***Scenario 4a***	***Scenario 4b***
Original	4	2	3	1	5
Forest moderate	4	2	3	1	-
Forest strict	3	2	4	1	-
Viable orangutan	4	2	3	1	-
Low profit	4	2	3	1	5
High profit	4	2	3	1	5
Low discount rate	5	2	4	1	3
High discount rate	4	2	3	1	5
Oil-palm match Sabah	5	2	3	1	4
High oil-palm profit	5	2	3	1	4
High timber profit	4	2	3	1	5
High CL profit	4	2	3	1	5
High RIL profit	5	2	4	1	3
Low oil-palm profit	5	2	4	1	3
Low timber profit	4	2	3	1	5
Low CL profit	5	2	3	1	4
Low RIL profit	4	2	3	1	5
Low profit, low discount rate	4	2	3	1	5
Low profit, high discount rate	4	2	3	1	5
High profit, low discount rate	5	2	3	1	4
High profit, high discount rate	4	2	3	1	5

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Appendix B: Supplementary Information for Chapter 2

Statistical Analysis

Cumulative logit mixed models (Agresti 2010, Christensen 2015) were used to model the relationship between the ordinal categorical impacts of climate change on ecosystem services, and the ecosystem service categories, methods used, the type of ecosystem, the spatial scale of the study, and the climate change attributes. To ensure our response categories were ordinal, we removed all records with a ‘mixed’ response, as these could not be meaningfully ordered among ‘negative’, ‘neutral’, and ‘positive’ categories. This removed 161 (24%) records, leaving a total of 510 records. We do not believe this would unduly affect our results given that ‘mixed’ responses are neutral with respect to increases or decreases. The number of records was larger than the number of studies (117) as each study could include multiple services and attributes of climate change. Since multiple records could come from the same study, the assumption of independence among observations was not satisfied. To account for this, we included a random-effect on the intercept for the study ID. We also tested for collinearity among explanatory variables using Cramer’s V (as all our explanatory variables were categorical), which showed low (<0.3) to moderate (0.3 – 0.5) associations among all variables prior to analysis (Table B.1). Consequently we determined that collinearity was sufficiently low.

Table B.1 | Correlations (Cramer’s V) among categorical explanatory variables used in the cumulative logit mixed model.

	Method used	*Climate change attribute*	*Ecosystem service category*	*Scale of study*	*Ecosystem type*
Method used	-	0.244	0.214	0.479	0.381
Climate change attribute	0.244	-	0.098	0.191	0.329
Ecosystem service category	0.214	0.098	-	0.151	0.168
Scale of study	0.479	0.191	0.151	-	0.386
Ecosystem type	0.381	0.329	0.168	0.386	-

We then used stepwise procedures (both forward and backward), based on likelihood ratio tests (Hilborn and Mangel 1997) at $p < 0.05$ to identify the significant explanatory variables. The ecosystem service categories (i.e., provisioning, regulating, or cultural) and the methods used (i.e., process-based, statistical, empirical, expert, or other) were the variables selected in the final model (Table B.2). We also fitted a saturated model to the data, which included the ecosystem service categories, methods used, the type of ecosystem, the spatial scale of the study, and the climate change attributes (Table B.3). Broad ecosystem service categories (i.e., provisioning, regulating, and cultural) were used instead of the 15 TEEB ecosystem service types as the sample size was not large enough across all of the individual ecosystem services (e.g., local climate regulation and medicinal resources had 9 records each). All explanatory variables were nominal, except for the spatial scale of the study, which was ordinal (6 levels) and modelled using orthogonal polynomial contrasts (e.g., linear, quadratic, cubic) to take into account different shapes of the effect over the range of ordered levels. Using either model (i.e., saturated or not) did not change the significance levels of coefficient estimates of the included variables. This analysis was conducted using the "clmm" function from the "ordinal" R Package (Christensen 2015) in R version 3.2.2 (R Core Team 2015).

Table B.2 | Regression coefficients and *p*-values from the cumulative logit mixed model with only the ecosystem service category and methods used as the explanatory variables. * Indicates *p*-values < 0.05 (no other *p*-values were significant).

	Coefficient	***Std. Error***	***z value***	***Pr(>|z|)***	
Ecosystem service category (nominal) | reference = Provisioning services					
Regulating services	-0.3823	0.3580	-1.068	0.28553	
Cultural services	-1.9017	0.6008	-3.165	0.00155	*
Methods used to assess impacts (nominal) | reference = Process-based models					
Statistical models	-0.4244	0.5502	-0.771	0.44049	
Empirical	-1.3173	1.1211	-1.175	0.24002	
Expert/stakeholder	-5.1745	1.744	-2.967	0.00301	*
Other methods	0.5589	0.9384	0.596	0.55145	

Table B.3 | Regression coefficients and *p*-values from the saturated cumulative logit mixed model. * Indicates *p*-values < 0.05 (no other *p*-values were significant).

	Coefficient	***Std. Error***	***z value***	***Pr(>|z|)***	
Ecosystem type (nominal) | reference = Terrestrial (only)					
Freshwater (only)	-2.24167	1.38929	-1.614	0.10663	
Marine (only)	-0.45239	1.59636	-0.283	0.77688	
Terrestrial and freshwater	-0.33755	0.87489	-0.386	0.69963	
Terrestrial and marine	1.10765	2.21992	0.499	0.61781	
Scale of study (ordinal)					
Linear trend	0.36095	1.2507	0.289	0.77289	
Quadratic trend	-0.17389	1.12806	-0.154	0.87749	
Cubic trend	1.46737	1.13493	1.293	0.19604	
4^{th} degree polynomial	-0.64901	0.93626	-0.693	0.48819	
5^{th} degree polynomial	-1.07542	0.86204	-1.248	0.2122	
Climate change attribute (nominal) | reference = Temperature increase					
Precipitation decrease	-0.16602	0.45206	-0.367	0.71342	
Precipitation increase	0.12941	0.47637	0.272	0.78589	
Increased precipitation variability	-0.09669	0.58665	-0.165	0.86908	
CO_2 fertilization	0.9606	0.68881	1.395	0.16314	
Sea level rise	-1.82468	1.97129	-0.926	0.35464	
Other climate change effects	-1.28001	1.36095	-0.941	0.34695	
Ecosystem service category (nominal) | reference = Provisioning services					
Regulating services	-0.38372	0.36267	-1.058	0.29004	
Cultural services	-1.93008	0.60835	-3.173	0.00151	*
Methods used to assess impacts (nominal) | reference = Process-based models					
Statistical models	-0.52714	0.55725	-0.946	0.34416	
Empirical	-0.3362	1.53505	-0.219	0.82664	
Expert/stakeholder	-5.49756	1.93807	-2.837	0.00456	*
Other methods	0.85129	0.97945	0.869	0.38476	

Table B.4 | The structured questions used to extract data from the journal articles, with answer categories. All questions had space to justify answers. The roman numerals indicate which component of the conceptual framework the section relates to. Each question relates to one of the aims: (a) identify gaps in the literature relating to the context of the assessments, (b) quantify the impacts of climate change and other drivers on ecosystem services, (c) determine how these impacts were measured or modelled, (d) determine how uncertainty was incorporated in these assessments, and (e) determine the extent to which decision making (actions, policies, or other interventions) was considered.

Category	*No.*	*Aim*	*Question*	*Answers*
Filter	1	-	Is the paper an assessment of ecosystem services?	Yes No, does not consider ecosystem services No, considers supporting/habitat services No, is not an assessment (i.e. a review/conceptual paper) No, other reason (specify below)
	2	-	Does the paper incorporate the impacts of climate change?	Yes No, just mentioned in the abstract (i.e. an assessment of carbon sequestration that mentions climate change mitigation) No, other reason (specify below)
(i) Study area	3	(a)	Spatial scale of assessment	Micro: <1 km^2 Patch: 1 – 100 km^2 Local: 100 – 1,000 km^2 Regional: 1,000 - 100,000 km^2 National: 100,000 - 1,000,000 km^2 Continental: 1,000,000 - 100,000,000 km^2 Global: > 100,000,000 km^2
	4	(a)	Location of assessment	Latitude/longitude Country Description
	5	(a)	Type of ecosystem(s)?	Terrestrial Freshwater Marine
(ii) Ecosystem services	6	(a)	Which ecosystem service(s) were considered? State the indicator used. Categories are based on TEEB (2010)	1. Food, 2. Raw Materials, 3. Fresh Water, 4. Medicinal resources, 5. Local climate and air quality, 6. Carbon sequestration and storage, 7. Moderation of extreme events, 8. Waste-water treatment, 9. Erosion prevention and maintenance, 10. Pollination, 11. Biological control, 14. Recreation and mental and physical health, 15. Tourism, 16. Aesthetic appreciation and inspiration for culture, art and design, 17. Spiritual experience and sense of place, 18. Other___________
	7	(a)	What aspect of each ecosystem service is considered? Definition of supply and delivery based on Tallis et al. (2012)	Supply (potential); Delivery/demand (actual); Monetary value;
	8	(c)	If monetary value was considered, what valuation method was used? Methods and definitions adapted from Christie et al. (2012)	Market methods Travel cost Hedonic methods Production approaches Contingent valuation Replacement cost Avoidance cost Benefit/value transfer Other__________

(iii) Drivers: Climate	9	(b)	What aspect(s) of climate change are considered (IPCC 2014)	1 Warming trend 2 Precipitation increase 3 Precipitation decrease (incl. drought) 4 Increased variability of precipitation 5 Carbon dioxide fertilization 6 Sea level rise 7 Other_________ 8 Other_________ 9 Other_________
	10	(b)	Were these attributes of climate change assessed cumulatively, in isolation from each other, or both?	Isolation Cumulative Both
	11	(b)	What was the impact of climate change on the ecosystem services studied?	Positive (increased the ES) Negative (decreased the ES) Neutral Mixed (increased and decreased)
	12	(b)	Are interactions between services considered (i.e., trade-offs)?	No Yes (summarize)
	13	(c)	What method was used to incorporate climate change and ecosystem services?	Empirical (field based or laboratory study) Statistical model (using field-based data) Statistical model (using estimates) Process-based model (using field based data) Process-based model (using estimates) Expert elicitation Other ____________
	14	(c)	Was the method static, or did it consider changes over time?	Static; Dynamic (list time interval)______
(iv) Drivers: other	15	(b)	Are other drivers considered?	Not considered; Mentioned/discussed; Explicitly modelled or otherwise quantitatively assessed
	16	(b)	If other (non-climate) drivers were incorporated; list the drivers.	
	17	(b)	What was the impact of the non-climate driver on the ecosystem service studied?	Positive (increased the ES) Negative (decreased the ES) Neutral Mixed (increased and decreased)
	18	(c)	How was the impact of the driver(s) assessed?	In isolation from climate change impacts (only) Cumulative impacts with climate change (only) Both cumulative impacts and in isolation
	19	(b)	How did each driver interact with climate change? (Brown *et al* 2013)	Synergistic Antagonistic Additive Unclear
(v) Decision making	20	(e)	Is decision-making considered (i.e., actions, policies, or other interventions)?	Not considered; Mentioned/discussed; Explicitly modelled or otherwise quantitatively assessed
	21	(e)	How many objectives are considered (list all)?	
	22	(e)	What method is used to model or assess the action, policy, or interventions?	

	23	(e)	What category do these actions, policies or other interventions fall into?	Allocating protected areas Allocating a range of land uses (land use zoning) Allocating management actions Specific legislation Payment for ecosystem services schemes Subsidies Levies Reverse auction New markets Awareness raising / education Other________
(vi) Uncertainty	24	(d)	Was uncertainty considered?	Uncertainty not considered; Uncertainty mentioned/discussed; Uncertainty explicitly incorporated;
	25	(d)	What was the source of the uncertainty, and what methods were used to incorporate it in the assessment? Methods were sourced from Polasky et al. (2011), Yousefpour et al. (2011), and Refsgaard et al. (2007). This question is answered in matrix form (i.e., source v methods)	SOURCES: The magnitude of climate change; The magnitude of other drivers; How climate change impacts ecosystem services; How other drivers impact ecosystem services; How any intervention (e.g. management) impacts ecosystem services; How ecosystem services are supplied; How ecosystem services are delivered; Other (specify below); METHODS: Scenario analysis (comparison of different, internally consistent, sets of assumptions about the future); Multiple models (assessment is carried out using different models of the same system); Sensitivity analysis (varying parameters of the analysis); Probabilistic - Monte Carlo analysis (statistical technique for stochastic model calculations); Probabilistic - Bayesian (a graphical model that represents a set of variables and their conditional dependencies); Other;
	26	(d) (e)	If decision making is considered, are the decisions robust to uncertainty?	No, Yes, Unclear If yes or unclear, briefly describe

Table B.5 | The final set of peer reviewed studies included in the analysis.

	Studies included in review
1	Abson DJ, Termansen M, Pascual U, *et al.* 2014. Valuing Climate Change Effects Upon UK Agricultural GHG Emissions: Spatial Analysis of a Regulating Ecosystem Service. *Environ Resour Econ* **57**: 215–31.
2	Altieri AH. 2008. Dead zones enhance key fisheries species by providing predation refuge. *Ecology* **89**: 2808–18.
3	Anastácio PM, Marques B, and Lillebø AI. 2013. Modeling the effect of temperature, solar radiation and salinity on Bolboschoenus maritimus sequestration of mercury. *Ecol Modell* **256**: 31–42.
4	Arias-Hidalgo M, Villa-Cox G, Griensven AV, *et al.* 2013. A decision framework for wetland management in a river basin context: The "Abras de Mantequilla" case study in the Guayas River Basin, Ecuador. *Environ Sci Policy* **34**: 103–14.
5	Bangash RF, Passuello A, Sanchez-Canales M, *et al.* 2013. Ecosystem services in Mediterranean river basin: climate change impact on water provisioning and erosion control. *Sci Total Environ* **458-460**: 246–55.
6	Bartomeus I, Park MG, Gibbs J, *et al.* 2013. Biodiversity ensures plant-pollinator phenological synchrony against climate change. *Ecol Lett* **16**: 1331–8.
7	Bateman IJ, Harwood AR, Mace GM, *et al.* 2013. Bringing ecosystem services into economic decision-making: land use in the United Kingdom. *Science* **341**: 45–50.
8	Bloor JMG and Bardgett RD. 2012. Stability of above-ground and below-ground processes to extreme drought in model grassland ecosystems: Interactions with plant species diversity and soil nitrogen availability. *Perspect Plant Ecol Evol Syst* **14**: 193–204.
9	Bohensky E, Butler JRA, Costanza R, *et al.* 2011. Future makers or future takers? A scenario analysis of climate change and the Great Barrier Reef. *Glob Environ Chang* **21**: 876–93.
10	Boithias L, Acuña V, Vergoñós L, *et al.* 2014. Assessment of the water supply:demand ratios in a Mediterranean basin under different global change scenarios and mitigation alternatives. *Sci Total Environ* **470-471**: 567–77.
11	Briner S, Elkin C, and Huber R. 2013. Evaluating the relative impact of climate and economic changes on forest and agricultural ecosystem services in mountain regions. *J Environ Manage* **129**: 414–22.
12	Briner S, Elkin C, Huber R, and Grêt-Regamey A. 2012. Assessing the impacts of economic and climate changes on land-use in mountain regions: A spatial dynamic modeling approach. *Agric Ecosyst Environ* **149**: 50–63.
13	Brito AC, Newton A, Tett P, and Fernandes TF. 2012. How will shallow coastal lagoons respond to climate change? A modelling investigation. *Estuar Coast Shelf Sci* **112**: 98–104.
14	Brittain C, Kremen C, and Klein A-M. 2013. Biodiversity buffers pollination from changes in environmental conditions. *Glob Chang Biol* **19**: 540–7.
15	Buma B and Wessman CA. 2013. Forest resilience, climate change, and opportunities for adaptation: A specific case of a general problem. *For Ecol Manage* **306**: 216–25.
16	Busch DS, Harvey CJ, and McElhany P. 2013. Potential impacts of ocean acidification on the Puget Sound food web. *ICES J Mar Sci* **70**: 823–33.
17	Butler JRA, Skewes T, Mitchell D, *et al.* 2014. Stakeholder perceptions of ecosystem service declines in Milne Bay, Papua New Guinea: Is human population a more critical driver than climate change? *Mar Policy* **46**: 1–13.
18	Buytaert W and Bièvre B De. 2012. Water for cities: The impact of climate change and demographic growth in the tropical Andes. *Water Resour Res* **48**: W08503.
19	Cavan G, Lindley S, Jalayer F, *et al.* 2014. Urban morphological determinants of temperature regulating ecosystem services in two African cities. *Ecol Indic* **42**: 43–57.
20	Charles H and Dukes JS. 2009. Effects of warming and altered precipitation on plant and nutrient dynamics of a New England salt marsh. *Ecol Appl* **19**: 1758–73.

21 Chown SL, Slabber S, McGeouch M, *et al.* 2007. Phenotypic plasticity mediates climate change responses among invasive and indigenous arthropods. *Proc Biol Sci* **274**: 2531–7.

22 Civantos E, Thuiller W, Maiorano L, *et al.* 2012. Potential Impacts of Climate Change on Ecosystem Services in Europe: The Case of Pest Control by Vertebrates. *Bioscience* **62**: 658–66.

23 Claessens L, Antle JM, Stoorvogel JJ, *et al.* 2012. A method for evaluating climate change adaptation strategies for small-scale farmers using survey, experimental and modeled data. *Agric Syst* **111**: 85–95.

24 Cook GS, Fletcher PJ, and Kelble CR. 2014. Towards marine ecosystem based management in South Florida: Investigating the connections among ecosystem pressures, states, and services in a complex coastal system. *Ecol Indic* **44**: 26–39.

25 Craft C, Clough J, Ehman J, *et al.* 2009. Forecasting the effects of accelerated sea-level rise on tidal marsh ecosystem services. *Front Ecol Environ* **7**: 73–8.

26 Dearing JA, Yang X, Dong X, *et al.* 2012. Extending the timescale and range of ecosystem services through paleoenvironmental analyses, exemplified in the lower Yangtze basin. *Proc Natl Acad Sci U S A* **109**: E1111–20.

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28 de Vries FT, Liiri ME, Bjørnlund L, *et al.* 2012. Land use alters the resistance and resilience of soil food webs to drought. *Nat Clim Chang* **2**: 276–80.

29 Ding H and Nunes PALD. 2014. Modeling the links between biodiversity, ecosystem services and human wellbeing in the context of climate change: Results from an econometric analysis of the European forest ecosystems. *Ecol Econ* **97**: 60–73.

30 Doherty RM, Sitch S, Smith B, *et al.* 2010. Implications of future climate and atmospheric CO 2 content for regional biogeochemistry, biogeography and ecosystem services across East Africa. *Glob Chang Biol* **16**: 617–40.

31 Ekvall MK, la Calle Martin J de, Faassen EJ, *et al.* 2013. Synergistic and species-specific effects of climate change and water colour on cyanobacterial toxicity and bloom formation. *Freshw Biol* **58**: 2414–22.

32 Elkin C, Gutiérrez AG, Leuzinger S, *et al.* 2013. A 2 °C warmer world is not safe for ecosystem services in the European Alps. *Glob Chang Biol* **19**: 1827–40.

33 Feagin R. 2010. Salt marsh zonal migration and ecosystem service change in response to global sea level rise: a case study from an urban region. *Ecol Soc* **15**: 14.

34 Fezzi C, Bateman I, Askew T, *et al.* 2014. Valuing Provisioning Ecosystem Services in Agriculture: The Impact of Climate Change on Food Production in the United Kingdom. *Environ Resour Econ* **57**: 197–214.

35 Ford CR, Laseter SH, Swank WT, and Vose JM. 2011. Can forest management be used to sustain water-based ecosystem services in the face of climate change? *Ecol Appl* **21**: 2049–67.

36 Forsius M, Anttila S, Arvola L, *et al.* 2013. Impacts and adaptation options of climate change on ecosystem services in Finland: a model based study. *Curr Opin Environ Sustain* **5**: 26–40.

37 García de Jalón S, Iglesias A, Cunningham R, and Pérez Díaz JI. 2013. Building resilience to water scarcity in southern Spain: a case study of rice farming in Doñana protected wetlands. *Reg Environ Chang* **14**: 1229–42.

38 Gathenya M, Mwangi H, Coe R, and Sang J. 2011. Climate- and land use-induced risks to watershed services in the Nyando River Basin, Kenya. *Agric Ecosyst Environ* **47**: 339–56.

39 Giannini TC, Acosta AL, Silva CI da, *et al.* 2013. Identifying the areas to preserve passion fruit pollination service in Brazilian Tropical Savannas under climate change. *Agric Ecosyst Environ* **171**: 39–46.

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41 Grêt-Regamey A, Bebi P, Bishop ID, and Schmid WA. 2008. Linking GIS-based models to value ecosystem services in an Alpine region. *J Environ Manage* **89**: 197–208.

42	Grêt-Regamey A, Brunner S, Altwegg J, *et al.* 2013. Integrating Expert Knowledge into Mapping Ecosystem Services Trade-offs for Sustainable Forest Management. *Ecol Soc* **18**: 34.
43	Grossmann M and Dietrich O. 2012. Integrated Economic-Hydrologic Assessment of Water Management Options for Regulated Wetlands Under Conditions of Climate Change: A Case Study from the Spreewald (Germany). *Water Resour Manag* **26**: 2081–108.
44	Hill MJ and Olson R. 2012. Possible future trade-offs between agriculture, energy production, and biodiversity conservation in North Dakota. *Reg Environ Chang* **13**: 311–28.
45	Hill SL, Phillips T, and Atkinson A. 2013. Potential climate change effects on the habitat of antarctic krill in the weddell quadrant of the southern ocean. *PLoS One* **8**: e72246.
46	Hoover SER, Ladley JJ, Shchepetkina AA, *et al.* 2012. Warming, CO2, and nitrogen deposition interactively affect a plant-pollinator mutualism. *Ecol Lett* **15**: 227–34.
47	Hoyer R and Chang H. 2014. Assessment of freshwater ecosystem services in the Tualatin and Yamhill basins under climate change and urbanization. *Appl Geogr* **53**: 402–16.
48	Huang L, Xu X, Shao Q, and Liu J. 2014. Improving Carbon Mitigation Potential through Grassland Ecosystem Restoration under Climatic Change in Northeastern Tibetan Plateau. *Adv Meteorol* **2014**: 379306.
49	Ilukor J, Bagamba F, and Bashaasha B. 2014. Application of the TOA-MD model to assess adoption potential of improved sweet potato technologies by rural poor farm households under climate change: the case of Kabale district in Uganda. *Food Secur* **6**: 359–68.
50	Inauen N, Körner C, and Hiltbrunner E. 2013. Hydrological consequences of declining land use and elevated CO 2 in alpine grassland (R Bardgett, Ed). *J Ecol* **101**: 86–96.
51	Jaramillo J, Setamou M, Muchugu E, *et al.* 2013. Climate change or urbanization? Impacts on a traditional coffee production system in East Africa over the last 80 years. *PLoS One* **8**: e51815.
52	Jentsch A, Kreyling J, Elmer M, *et al.* 2011. Climate extremes initiate ecosystem-regulating functions while maintaining productivity. *J Ecol* **99**: 689–702.
53	Jung IW, Bae DH, and Lee BJ. 2013. Possible change in Korean streamflow seasonality based on multi-model climate projections. *Hydrol Process* **27**: 1033–45.
54	Klein T, Holzkämper A, Calanca P, and Fuhrer J. 2013. Adaptation options under climate change for multifunctional agriculture: a simulation study for western Switzerland. *Reg Environ Chang* **14**: 167–84.
55	Koca D, Smith B, and Sykes MT. 2006. Modelling Regional Climate Change Effects On Potential Natural Ecosystems in Sweden. *Clim Change* **78**: 381–406.
56	Lamarque P, Lavorel S, Mouchet M, and Quétier F. 2014. Plant trait-based models identify direct and indirect effects of climate change on bundles of grassland ecosystem services. *Proc Natl Acad Sci U S A* **111**: 13751–6.
57	Lépy É, Heikkinen HI, Karjalainen TP, *et al.* 2014. Multidisciplinary and Participatory Approach for Assessing Local Vulnerability of Tourism Industry to Climate Change. *Scand J Hosp Tour* **14**: 41–59.
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59	Liersch S, Cools J, Kone B, *et al.* 2013. Vulnerability of rice production in the Inner Niger Delta to water resources management under climate variability and change. *Environ Sci Policy* **34**: 18–33.
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89	Rounsevell MDA, Ewert F, Reginster I, *et al.* 2005. Future scenarios of European agricultural land use. *Agric Ecosyst Environ* **107**: 117–35.
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94	Schröter D, Cramer W, Leemans R, *et al.* 2005. Ecosystem service supply and vulnerability to global change in Europe. *Science* **310**: 1333–7.
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103	Steenberg JWN, Duinker PN, and Bush PG. 2011. Exploring adaptation to climate change in the forests of central Nova Scotia, Canada. *For Ecol Manage* **262**: 2316–27.
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Appendix C: Supplementary Information for Chapter 3

Table C.1 | The change in the provision of wetlands and ecosystem services under sea level rise. 'Protected' refers to the area of wetlands, amount of carbon sequestration, or area of nursery habitat that falls within the current reserve network. 'Total' refers to the sum of the protected and unprotected wetland area or ecosystem service provision. The percentage change from the baseline (no sea level rise) is given in parentheses.

	Wetland area (% change from 0 cm)		*Carbon sequestration (% change from 0 cm)*		*Nursery habitat area (% change from 0 cm)*	
	Total	*Protected*	*Total*	*Protected*	*Total*	*Protected*
0 cm	10,933 ha (-)	5,577 ha (-)	52.1 Mg yr^{-1} (-)	24.6 Mg yr^{-1} (-)	256.6 ha (-)	209.5 ha (-)
28 cm	15,359 ha (+40.5%)	5,299 ha (-4.9%)	62.7 Mg yr^{-1} (+20.3%)	19.9 Mg yr^{-1} (-19.3%)	346.5 ha (+35.0%)	212.0 ha (+1.2%)
55 cm	16,412 ha (+50.1%)	5,158 ha (-7.5%)	64.3 Mg yr^{-1} (+23.4%)	19.4 Mg yr^{-1} (-21.1%)	390.9 ha (+52.3%)	222.7 ha (+6.3%)
98 cm	15,611 ha (+42.8%)	4,144 ha (-25.7%)	58.4 Mg yr^{-1} (+12.1%)	14.4 Mg yr^{-1} (-41.3%)	478.5 ha (+86.5%)	209.0 ha (-0.2%)
128 cm	14,824 ha (+35.6%)	3,830 ha (-31.3%)	55.3 Mg yr^{-1} (+6.1%)	13.1 Mg yr^{-1} (-46.8%)	655.3 ha (+155.4%)	252.5 ha (+20.5%)

Table C.2 | The variation in the potential for payments that reflect the social value of carbon and the total value of nursery habitat to attenuate the costs of preserving wetlands under sea level rise. 'Current' refers to the current extent of wetlands that are protected in the study site (5577 ha). '+50%' refers to a 50% increase in the current extent (8365 ha). 'Total' refers to the overall cost (or profit, if negative) in million 2012 AUD. 'Additional' refers to the additional cost when compared to the baseline of no sea level rise. Values in parenthesis refer to the minimum and maximum values respectively.

	Social Value of Carbon				*Social Value of Carbon & Nursery Habitat Payments*			
	Current		*+50%*		*Current*		*+50%*	
	Total	*Additional*	*Total*	*Additional*	*Total*	*Additional*	*Total*	*Additional*
0 cm	-43.47 (-2.69,-84.94)	- -	-43.47 (-1.99,-84.94)	- -	-50.83 (-7.62,-100.75)	- -	-50.83 (-7.03,-100.75)	- -
28 cm	-37.47 (-2.25,-76.52)	6 (0.44,8.41)	-37.47 (-1.36,-76.52)	6 (0.62,8.41)	-45.89 (-7.80,-95.86)	4.95 (-0.17,4.88)	-45.89 (-6.94,-95.86)	4.95 (0.10,4.88)
55 cm	-37.59 (-2.25,-76.83)	5.88 (0.43,8.11)	-37.59 (-1.34,-76.830)	5.88 (0.65,8.11)	-46.01 (-7.80,-96.33)	4.82 (-0.18,-0.46)	-46.01 (-6.92,-96.33)	4.82 (0.12,-0.46)
98 cm	-37.65 (-2.25,-76.25)	5.82 (0.43,8.69)	-37.58 (1.51,-76.25)	5.89 (3.49,8.69)	-46.17 (-7.82,-96.17)	4.66 (-0.20,0.16)	-46.14 (-4.27,-96.17)	4.69 (2.76,0.16)
128 cm	-37.3 (-2.24,-74.55)	6.17 (0.45,10.39)	-36.42 (4,27,-74.55)	7.05 (6.26,10.39)	-46.18 (-8.35,-95.15)	4.65 (-0.73,1.01)	-45.6 (-2.23,-95.15)	5.24 (4.80,1.01)

Table C.3 | The additional cost from using the strict connectivity requirement when compared to the more flexible connectivity requirement (in $1,000s 2012 AUD). The more flexible connectivity requirement only resulted in a minor cost difference (maximum 5.3% of the total cost).

	Increase in area of reserve network					
	0%	*10%*	*20%*	*30%*	*40%*	*50%*
0 cm	$0	$0	$0	$0	$0	$0
28 cm	$0	$0	$0	$5.13 (1.6%)	$13.08 (2.1%)	$20.00 (1.3%)
55 cm	$0	$0	$0	$0	$20.00 (3.0%)	$20.00 (1.3%)
98 cm	$0	$0	$5.13 (0.8%)	$60.85 (3.7%)	$41.11 (1.4%)	$48.40 (1.0%)
128 cm	$0	$0	$56.20 (3.8%)	$21.79 (0.7%)	$271.50 (5.3%)	$24.15 (0.3%)

Table C.4 | The variation in, and combinations of, ecosystem value estimates and discount rates when capitalizing the value of ecosystem services to 2100. These combinations contain variations of: discount rates (DR), voluntary carbon payments (VC), carbon payments reflecting the social value of carbon (SC), and the method used to calculate nursery habitat payments (NH) (which was either a linear feature [line], a 5 m landward strip [5 m], or a 10 m landward strip [10 m]). All values are in 2012 AUD.

	Carbon price (MgC^{-1})	*Nursery habitat value*	*Discount rate*
Voluntary carbon payments			
VC Main, DR main	$6.11	-	10%
VC low, DR main	$0.124	-	10%
VC high, DR main	$9.63	-	10%
VC main, DR low	$6.11	-	5%
VC low, DR low	$0.124	-	5%
VC high, DR low	$9.63	-	5%
VC main, DR high	$6.11	-	15%
VC low, DR high	$0.124	-	15%
VC high, DR high	$9.63	-	15%
Nursery habitat levy payments			
NH 5 m, DR main	-	$118.7 ha^{-1} yr^{-1}	10%
NH line, DR main	-	$64.9 km^{-1} yr^{-1}	10%
NH 10 m, DR main	-	$60.5 ha^{-1} yr^{-1}	10%
NH 5 m, DR low	-	$118.7 ha^{-1} yr^{-1}	5%
NH line, DR low	-	$64.9 km^{-1} yr^{-1}	5%
NH 10 m, DR low	-	$60.5 ha^{-1} yr^{-1}	5%
NH 5 m, DR high	-	$118.7 ha^{-1} yr^{-1}	15%
NH line, DR high	-	$64.9 km^{-1} yr^{-1}	15%
NH 10 m, DR high	-	$60.5 ha^{-1} yr^{-1}	15%
Voluntary carbon payments & Nursery habitat levy payments			
VC Main, NH 5m, DR main	$6.11	$118.7 ha^{-1} yr^{-1}	10%
VC low, NH 5m, DR main	$0.124	$118.7 ha^{-1} yr^{-1}	10%
VC high, NH 5m, DR main	$9.63	$118.7 ha^{-1} yr^{-1}	10%
VC main, NH 5m, DR low	$6.11	$118.7 ha^{-1} yr^{-1}	5%
VC low, NH 5m, DR low	$0.124	$118.7 ha^{-1} yr^{-1}	5%
VC high, NH 5m, DR low	$9.63	$118.7 ha^{-1} yr^{-1}	5%
VC main, NH 5m, DR high	$6.11	$118.7 ha^{-1} yr^{-1}	15%
VC low, NH 5m, DR high	$0.124	$118.7 ha^{-1} yr^{-1}	15%
VC high, NH 5m, DR high	$9.63	$118.7 ha^{-1} yr^{-1}	15%
VC Main, NH line, DR main	$6.11	$64.9 km^{-1} yr^{-1}	10%
VC low, NH line, DR main	$0.124	$64.9 km^{-1} yr^{-1}	10%
VC high, NH line, DR main	$9.63	$64.9 km^{-1} yr^{-1}	10%
VC main, NH line, DR low	$6.11	$64.9 km^{-1} yr^{-1}	5%
VC low, NH line, DR low	$0.124	$64.9 km^{-1} yr^{-1}	5%

VC high, NH line, DR low	$9.63	$64.9 km^{-1} yr^{-1}	5%
VC main, NH line, DR high	$6.11	$64.9 km^{-1} yr^{-1}	15%
VC low, NH line, DR high	$0.124	$64.9 km^{-1} yr^{-1}	15%
VC high, NH line ,DR high	$9.63	$64.9 km^{-1} yr^{-1}	15%
VC low, NH 10 m, DR main	$0.124	$60.5 ha^{-1} yr^{-1}	10%
VC high, NH 10 m, DR main	$9.63	$60.5 ha^{-1} yr^{-1}	10%
VC main, NH 10 m, DR low	$6.11	$60.5 ha^{-1} yr^{-1}	5%
VC low, NH 10 m , DR low	$0.124	$60.5 ha^{-1} yr^{-1}	5%
VC high, NH 10 m, DR low	$9.63	$60.5 ha^{-1} yr^{-1}	5%
VC main, NH 10 m, DR high	$6.11	$60.5 ha^{-1} yr^{-1}	15%
VC low, NH 10 m, DR high	$0.124	$60.5 ha^{-1} yr^{-1}	15%
VC high, NH 10 m ,DR high	$9.63	$60.5 ha^{-1} yr^{-1}	15%
	Carbon price (MgC^{-1})	*Nursery habitat value*	*Discount rate*
Social carbon payments			
SC high, DR main	$96.94	-	10%
SC low, DR main	$10.94	-	10%
SC high, DR low	$96.94	-	5%
SC low, DR low	$10.94	-	5%
SC high, DR high	$96.94	-	15%
SC low, DR high	$10.94	-	15%
Social carbon & full nursery habitat payments			
SC high, NH 5 m, DR main	$96.94	$2,967.6 ha^{-1} yr^{-1}	10%
SC low, NH 5 m, DR main	$10.94	$2,967.6 ha^{-1} yr^{-1}	10%
SC high, NH line, DR main	$96.94	$1,622.7 km^{-1} yr^{-1}	10%
SC low, NH line, DR main	$10.94	$1,622.7 km^{-1} yr^{-1}	10%
SC high, NH 10 m, DR main	$96.94	$1,511.5 ha^{-1} yr^{-1}	10%
SC low, NH 10 m, DR main	$10.94	$1,511.5 ha^{-1} yr^{-1}	10%
SC high, NH 5 m, DR low	$96.94	$2,967.6 ha^{-1} yr^{-1}	5%
SC low, NH 5 m, DR low	$10.94	$2,967.6 ha^{-1} yr^{-1}	5%
SC high, NH line, DR low	$96.94	$1,622.7 km^{-1} yr^{-1}	5%
SC low, NH line, DR low	$10.94	$1,622.7 km^{-1} yr^{-1}	5%
SC high, NH 10 m, DR low	$96.94	$1,511.5 ha^{-1} yr^{-1}	5%
SC low, NH 10 m, DR low	$10.94	$1,511.5 ha^{-1} yr^{-1}	5%
SC high, NH 5 m, DR high	$96.94	$2,967.6 ha^{-1} yr^{-1}	15%
SC low, NH 5 m, DR high	$10.94	$2,967.6 ha^{-1} yr^{-1}	15%
SC high, NH line, DR high	$96.94	$1,622.7 km^{-1} yr^{-1}	15%
SC low, NH line, DR high	$10.94	$1,622.7 km^{-1} yr^{-1}	15%
SC high, NH 10 m, DR high	$96.94	$1,511.5 ha^{-1} yr^{-1}	15%
SC low, NH 10 m, DR high	$10.94	$1,511.5 ha^{-1} yr^{-1}	15%

Appendix D: Supplementary Information for Chapter 4

Supplementary Methods:

Classification Uncertainty

To visualise the spatial uncertainty in land cover type (dryland, wetlands or water), we calculated the classification uncertainty (adapted from Runting et al. (2015) and Levin et al. (2013)):

$$U_i = \left(1 - \left[\frac{\frac{M_i}{S_i} - \frac{1}{n}}{1 - \frac{1}{n}}\right]\right) \tag{D.1}$$

Where U_i is the classification uncertainty for pixel i; M_i is the greatest number of times a particular land cover type was simulated for pixel i; n is the total number of land cover types (in this case 3); and S_i is the total number of runs (in this case 804). Pixels where each land cover type was simulated to occur an equal number of times would receive a value of 1, whereas pixels where only one land cover type was simulated to occur were given a value of zero. This enabled a spatial depiction of the uncertainty in the land cover types shown in Figure 4.1.

Standardisation of conservation objectives

Each of the 1225 planning units had 804 projections of each of the three conservation objectives in 2100. The values for each conservation objective were standardised by the range of the means across all scenarios for each property:

$$x' = \frac{x - \min(\bar{x})}{\max(\bar{x}) - \min(\bar{x})} \tag{D.2}$$

Where x are the raw values for each objective (a matrix of 804 scenarios by 1225 planning units), $\bar{x}$ are the means for each planning unit across scenarios, and x' are the scaled values. The means of the scaled vales in each property range from 0-1.

Supplementary Figures:

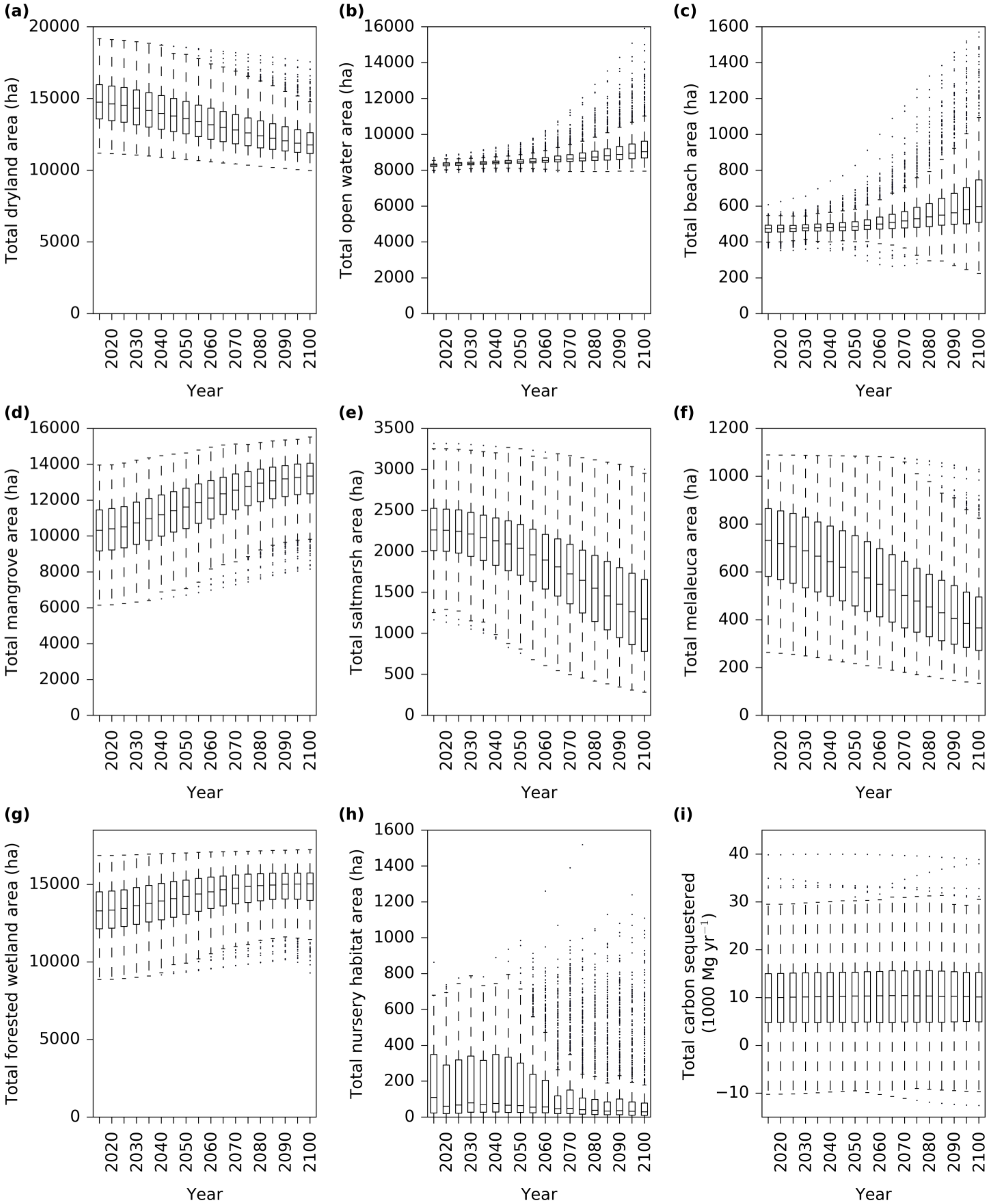

Figure D.1 | The uncertainty and change in wetland types and ecosystem services to 2100. This includes (a) dryland, (b) open water, (c) beaches and tidal flats, (d) mangroves, (e) saltmarsh, (f) melaleuca, (g) total forested wetlands (i.e., excluding beaches and tidal flats), (h) nursery habitat, and (i) carbon sequestration.

Supplementary Tables:

Table D.1 | Parameters (other than future sea level rise and elevation) that were varied within SLAMM. A normal distribution was assumed. The equivalent vegetation type for the study site is given in square brackets where relevant.

Parameter	Units	Mean	s.d.	Justification/reference
Historic trend of sea level rise	mm/yr	1.929	0.4	Based on a linear regression of mean sea level data from 1984-2010 (Lovelock et al 2011).
Mean tide level (MTL) - NAVD88/AHD	m	0.056923	0.079	From Queensland tide tables (Maritime Safety Queensland and Department of Transport and Main Roads 2014). Only locations within study site were used (n = 18).
Tidal Range	m	1.531111	0.314	
Salt Elevation	m above MTL	1.293333	0.246	
Accretion				
Irreg.-Flood Marsh [Claypan, Samphire, Sporobolus grassland]	mm/yr	0.597302	0.983	From Lovelock et al (2014).
Mangrove [Mangrove upper]	mm/yr	2.42	1.21	From Lovelock et al (2014).
Reg Flood Max. [Mangrove lower]	mm/yr	2.42	1.21	Tinchi Tamba Reserve measurements from Lovelock et al (2014).
Reg Flood Min. [Mangrove lower]	mm/yr	0.41	0.57	Halloran Reserve measurements from Lovelock et al (2014).
Reg Flood Elev c coeff. [Mangrove lower]	linear	-1		Ensures mangrove accretion rates are higher at lower elevations.
Tidal-fresh marsh, inland-fresh marsh, swamp, and tidal swamp [Grasslands, Sedgelands, Melaleuca]	mm/yr	0.051917	0.53	No field data on accretion for these vegetation types in Moreton Bay - we anticipate they are not significantly increasing in elevation as they are rarely inundated by tide (no sediment). Therefore, we assume their accretion is similar to Juncus marshes in Lovelock et al (2014).
Beach Sedimentation Rate	mm/yr	0.5	0.2	SLAMM defaults used (USFWS 2012). This value is largely irrelevant as beaches are a small part of the study area and are not the focus of our study.
Erosion				
Marsh	horz. m/yr	2	0.8	SLAMM defaults used (USFWS 2012). These parameters are largely irrelevant to the Moreton Bay study site, as they only apply where wetlands are exposed to open ocean with >9km fetch. In our site the wetlands are sheltered within the bay.
Swamp	horz. m/yr	1	0.4	
Tidal Flat	horz. m/yr	0.2	0.08	
Overwash				
Marsh Percent Loss overwash	%	10	4	Not relevant for study area - no marshes/mangroves are landward of beach so a low value is used (with variation as the SLAMM defaults (USFWS 2012)).
Mangrove Percent Loss overwash	%	10	4	
Frequency of overwash	years	25	5	Not important due to small amount of beach. Estimations based on McCauley and Tomlinson (2006) and Traill et al. (2011).
Beach to Ocean overwash	m	24	6	

Table D.2 | Estimates for soil carbon sequestration. The means and standard deviations are based on field data from Lovelock et al. (2014), and are given in g C m-2 y-1. The South East Queensland (SEQ) Wetland Classes are from Dowling & Stephens (1998). There is substantial variation in the amount of carbon sequestered in salt marsh communities across Moreton Bay, so we separated these communities into 'high' and 'low' carbon sequestration categories. The high and low carbon sequestration saltmarsh communities were categorized in accordance with their SEQ Wetland Class, based on the dominant vegetation reported in Lovelock et al (2014) and field observations.

Wetland type	***Mean***	***s.d.***	***SLAMM codes***	***SEQ Wetland Classes***
Mangroves	64	57	8, 9	Mangroves (class 1A-F)
Salt marsh: high C	253	319	5, 6, 7, 23	Sedgelands (class 6A-D), grasslands (class 4B-D), & casuarina (class 5A-C)
Salt marsh: low C	8	14	20	Claypan (class 2), samphire (class 3A), & sporobolus grassland (4A(i))

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Appendix E: Supplementary Information for Chapter 5

Fire Modelling

Fire hazard

Fire hazard in the north of Australia was modelled using survival analysis in the *R* statistical software environment (R Core Team 2015). Modelling the relationship of both temperature and rainfall to fire events for each location in the study area enabled the simulation of fire hazard to be extended to consider the effects of climate change.

Fire frequency data for Australia from 1988 – 2014 was obtained from WA Firewatch, Landgate (*www.firewatch.landgate.wa.gov.au*). This 1 km spatial resolution data was resampled to 2 km and combined with resampled 3″ANUCLIM outputs of mean annual temperature, mean annual rainfall (Hutchinson *et al* 2008) and resampled 100m NVIS 3.1 vegetation presence (0, 1) (DEWR 2007). This data was loaded into *R*, reformatted into a survival dataset, and parametric frailty modelling (PFM) was undertaken for vegetated locations using the R package *parfm 2.5.15* (Munda *et al* 2012). The *select.parfm* function was used to compute Akaike and Bayesian information criterion (AIC and BIC) values of parametric frailty models with different baseline hazards and different frailty distributions (Table E.1). Although the lognormal and loglogistic distributions performed better, they were not chosen due to potential unreliability, and the Weibull distribution was instead used to represent baseline hazard with a gamma distribution for frailty (Eqn E.1 – R code).

```
parFrail <- parfm(Surv(Time, Status) ~ meanrain + meantemp,   cluster="ID", data=survDS,
        dist="weibull", frailty="gamma", method="Nelder-Mead", maxit=50000,
        showtime=TRUE)                                                        (E.1)
```

Table E.1 | AIC and BIC results.

Baseline hazard distribution	Frailty distribution					
	AIC			BIC		
	gamma	inverse Gaussian	positive stable	gamma	inverse Gaussian	positive stable
exponential	851.907	848.529	873.069	865.625	862.246	886.787
weibull	811.113	811.565	846.897	828.26	828.712	864.044
gompertz	843.624	----	874.806	860.771	----	891.953
loglogistic	760.35	----	790.104	777.497	----	807.251
lognormal	756.629	757.692	----	773.776	774.839	----

Frailty for each vegetated location was then calculated from the PFM output parameters (Table E.2) (Munda *et al* 2012). Results were then imported into a GIS and a mean focal statistics method was used to provide frailty measures for (currently) non-vegetated areas. The frailty was then used in *R* to calculate and export instantaneous hazard (Eqn E.2 – R code) for each year (t) in a 100 year period for each location under mean annual rainfall and temperature:

```
hzrd <- rho * lambda * t^(rho-1)  * frailModXY_full$frailMod * exp(meanraincoeff  *
        dFXYPCs$meanrain + meantempcoeff * dFXYPCs$meantemp)        (E.2)
```

Table E.2 | Parametric frailty modelling results

	Estimate	Standard error	p-value
theta	1.320	0.004	
rho	1.564	0.001	
lambda	7.891316e-07	4.097809e-08	
meanrain	0.002	8.006945e-06	0 ***
meantemp	0.388	0.002	0 ***

Loglikelihood: -3992791.98
Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
Kendall's Tau: 0.398

Changes in rainfall and temperature for 2050, modelled under three climate scenarios (RCP2.6, RCP4.5 and RCP8.5) (Figures E.1 and E.2), were then applied to the mean annual rainfall and temperature and instantaneous hazard for a 100 year period again calculated (Eqn E.3 – R code). Figure E.3 provides examples of instantaneous hazard for three locations.

```
hzrd <- rho * lambda * t^(rho-1) * frailModXY_full$frailMod * exp(meanraincoeff *
        precipDelta + meantempcoeff * tempDelta)                    (E.3)
```

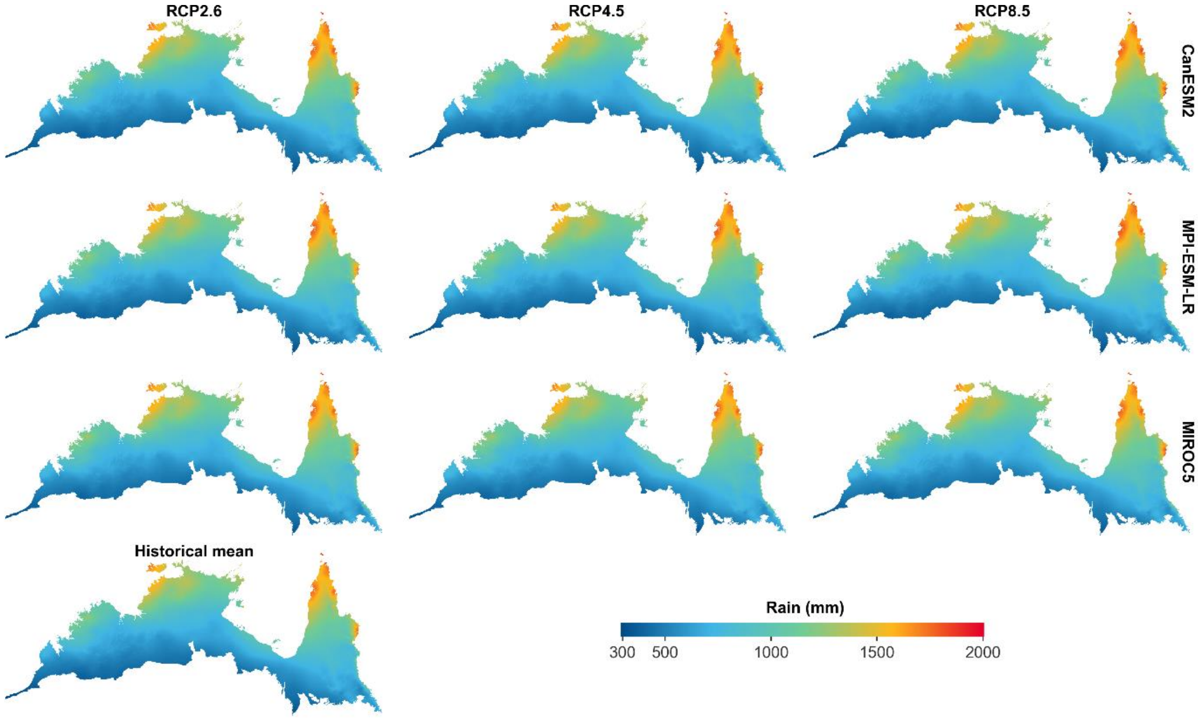

Figure E.1 | Rain in 2050 across scenarios compared with the ANUCLIM historical mean.

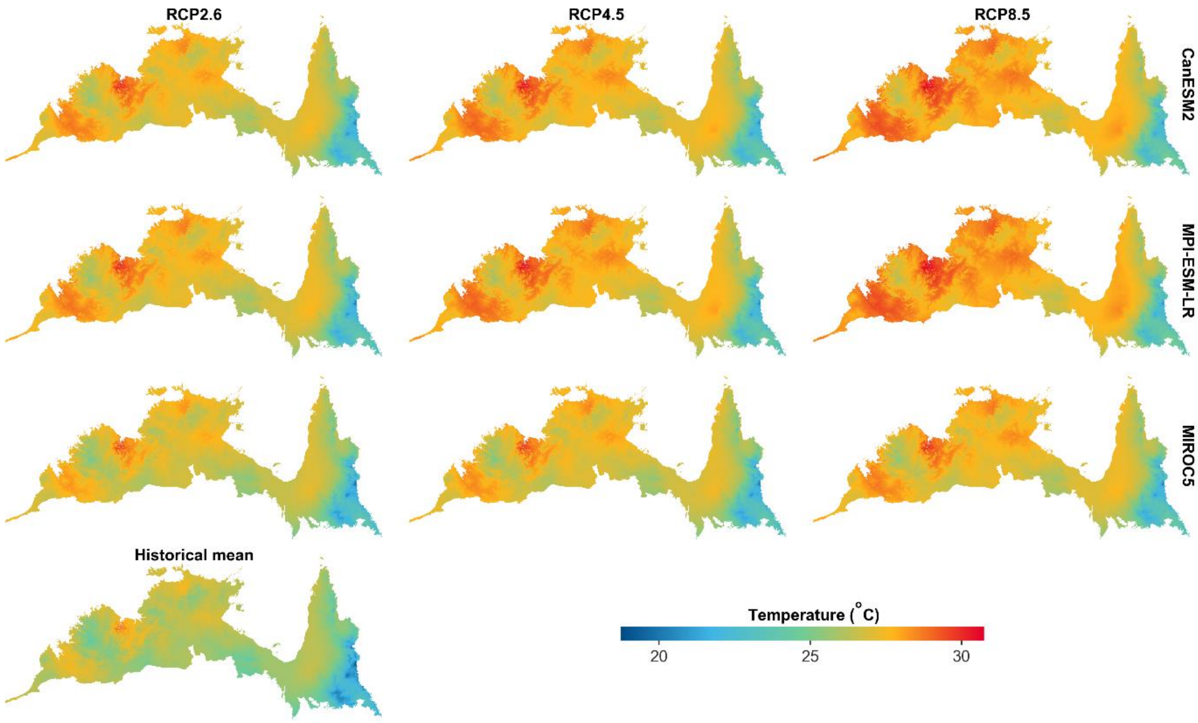

Figure E.2 | Temperature in 2050 across scenarios compared with the ANUCLIM historical mean.

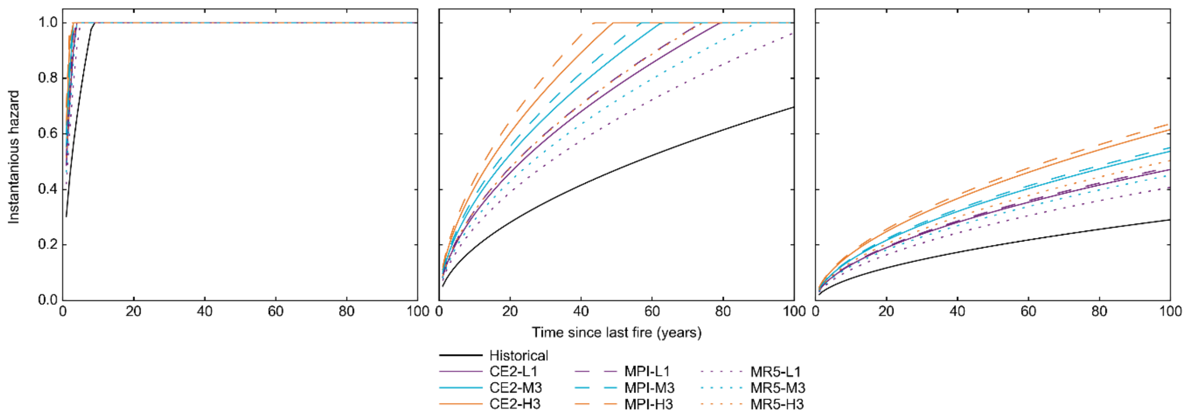

Figure E.3 | Examples of calculated instantaneous hazard. Here, global outlook M3 represents both M3 and M2, as these were both based on RCP 4.5.

Fire severity

Fire severity, as the percentage of biomass lost to fire, was modelled using the MODIS Nadir BRDF-Adjusted Reflectance 16-Day L3 Global 500m data for years from 2002 – 2014 (NASA LP DAAC 2015). The Normalised Burn Ratio (NBR - Eqn E.4) was originally developed with Landsat satellite data using the near infra-red band 4 and mid infra-red band 7 (Lopez Garcia and Caselles 1991).

$$NBR = \frac{iR_n - iR_m}{iR_n + iR_m} \quad \text{(E.4)1}$$

Where iR_n is near infra-red and iR_m is mid infra-red. The differencing of MODIS derived pre-fire NBR and post-fire NBR has been used in burned area mapping (Loboda *et al* 2007). A relative differencing of the NBR (RdNBR - Eqn E.5) using Landsat satellite data has been found to allow a more direct comparison of severity between fires across space and time (Miller and Thode 2007).

$$RdNBR = \frac{NBR_{Pre-fire} - NBR_{Post-fire}}{\sqrt{|NBR_{Pre-fire}|}} \quad \text{(E.5)}$$

MODIS Band 2 (near infra-red) and Band 7 (mid infra-red) were used to calculate the relative differenced normalised burn ratio (RdNBR) for burn areas defined by the Landgate dataset. The 5th, 50th (median) and 95th percentile of RdNBR for Interim Biogeographic Regionalisation for Australia (Australian Government 2012) regions was calculated (Figure E.4).

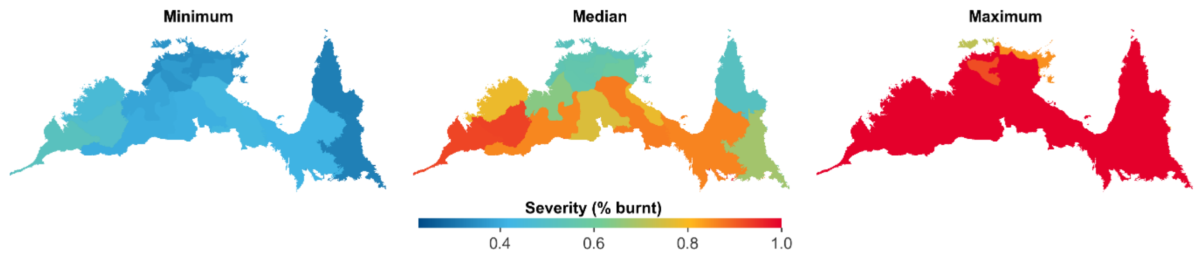

Figure E.4 | Range of severity by IBRA regions.

Fire simulations

The fire simulations were produced using Python (van Rossum and the Python Community 2012) and Numpy (Jones *et al* 2001). For each location, over a one hundred year period, fire events and their severity were simulated under mean conditions and for 2050 under the three climate scenario. The fire simulations modelled at the 2 km spatial resolution was resampled to 0.01 degree spatial resolution for use in the integrated simulation model. Fire events at each location were simulated using a random draw from a binomial distribution determined by the instantaneous hazard with time since last fire event determining the level of hazard. Severity of fire events was drawn from a triangular distribution using the range of RdNBR for each location.

Results

The simulations of fire events under historical mean conditions were used to assess model accuracy. A mean absolute error of 4.07% and a standard error of 5.72% indicates a good fit with mapped historical fire events. A bias, mean difference between historical fire frequency and simulated fire frequency, of -0.34% shows a slight overall over estimate of fire frequency. Figure E.5 provides a comparison of actual versus modelled fire frequency for simulations resampled to 0.01 degree spatial resolution. Although some spatial accuracy is lost in the resampling of results a visual comparison of mapped actual and simulated percentage frequency of fire events at the 0.01 degree resolution shows the overall pattern of fire frequency is reproduced by the simulations (Figure E.6).

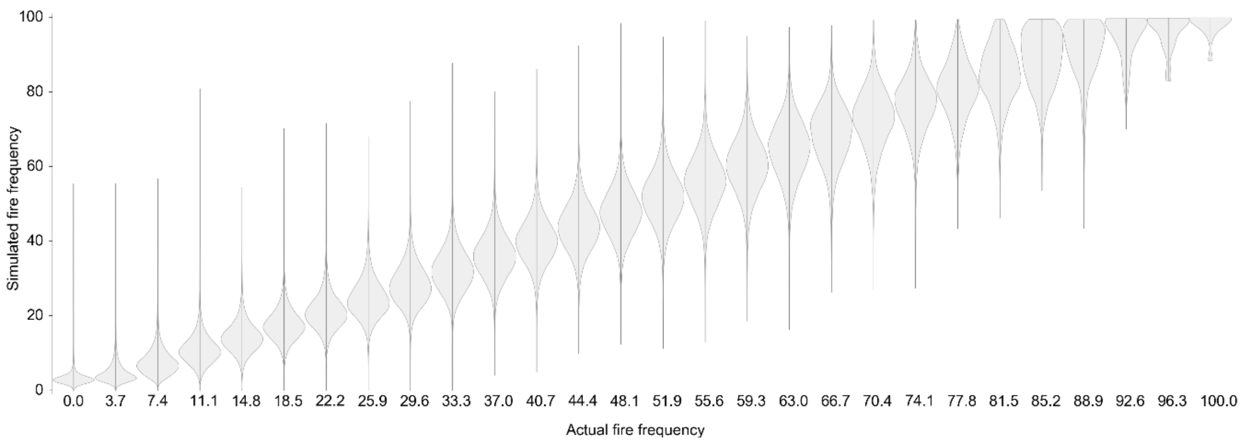

Figure E.5 | Violin plot of actual versus simulated fire frequency. Actual fire frequency was calculated as the number of years burnt within the 27 years of burn area data.

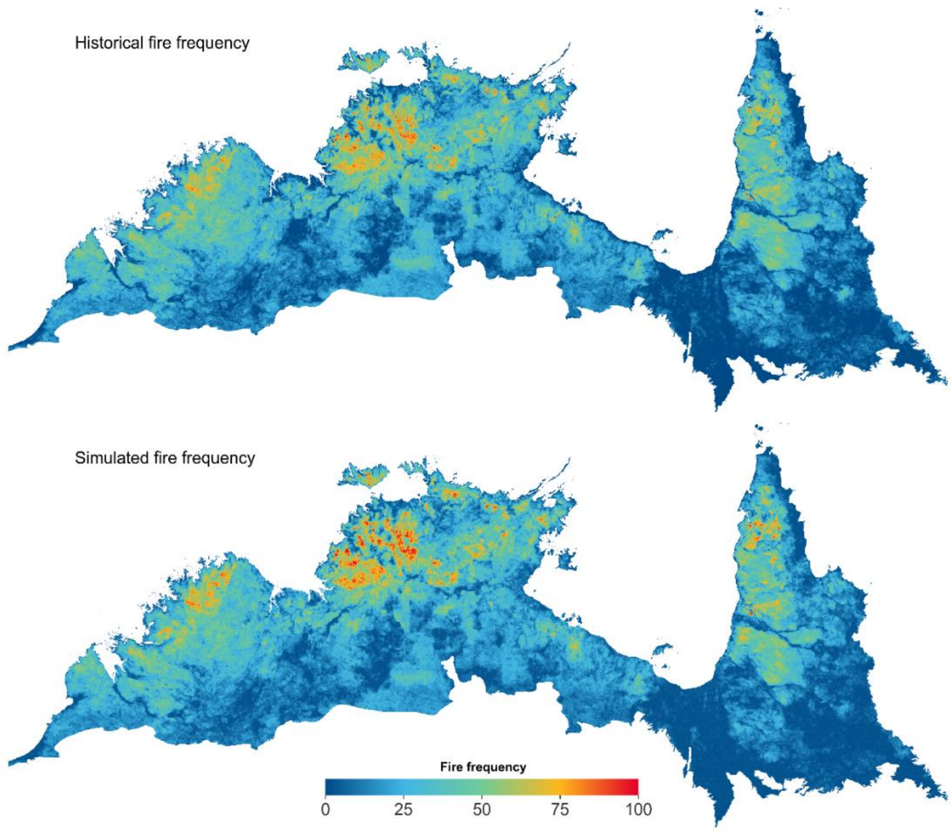

Figure E.6 | Comparison of fire frequency (top) with fire event simulations modelled on historical mean climate (bottom).

Temperature increases vary between all climate scenarios with this variation reflected in the fire event simulations (Figure E.7) as expected with the positive relationship between fire events and temperature indicated by the PFM temperature coefficient. Mean frequency of simulations match actual, and increase with increasing temperature in the 2050 simulations (Table E.3). The MIROC5 global climate modelling having the smallest increase followed by CanESM2 with the MPI-ESM-LR modelling having the highest. Area of low frequency fires reduces and areas of higher frequency fires increases as temperatures increases (Table E.4). The median percentage biomass lost (Figure E.8) increases as with fire events by climate scenario however, the spatial pattern of increase reflects variations in severity by IBRA regions.

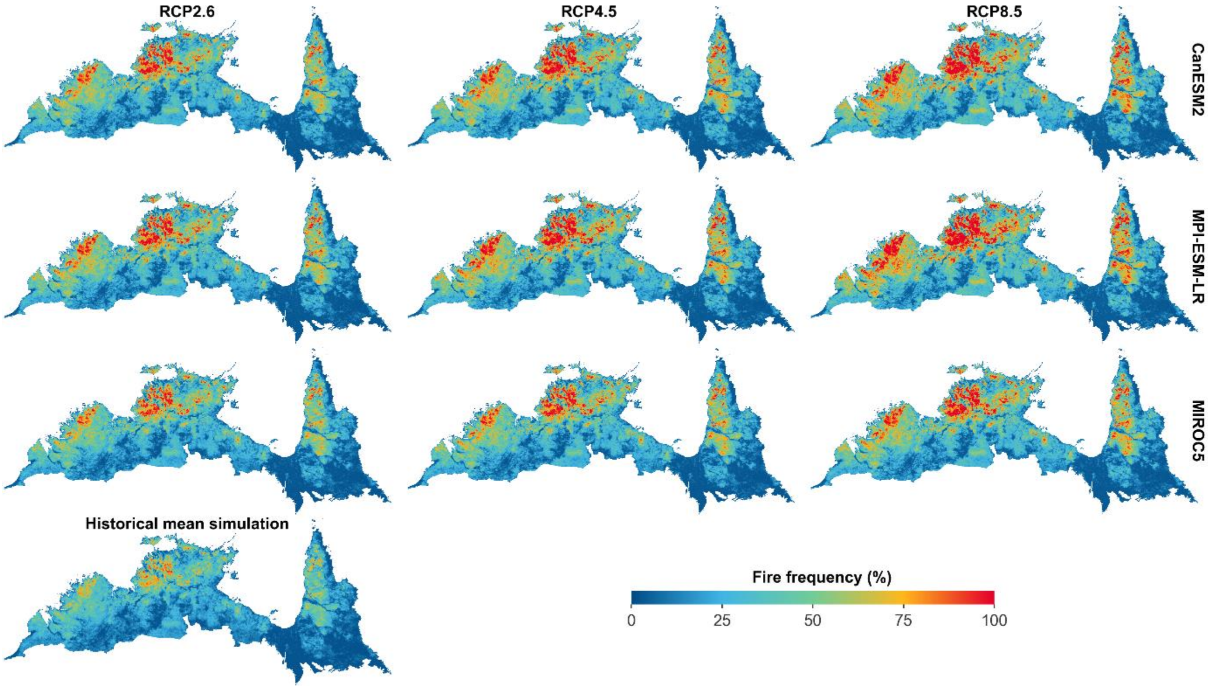

Figure E.7 | Comparison of fire event simulations over three different RCPs and GCMs.

Table E.3 | Historical and simulated fire frequency mean and standard deviation.

Scenario	Mean	STD
Actual 1988-2014	22.31	17.66
Historical mean climate	22.65	17.80
MIROC5 RCP2.6 - 2050	28.06	21.82
MIROC5 RCP4.5 - 2050	29.89	23.09
MIROC5 RCP8.5 - 2050	31.97	24.44
CanESM2 RCP2.6 - 2050	30.36	23.19
CanESM2 RCP4.5 - 2050	32.78	24.68
CanESM2 RCP8.5 - 2050	35.42	26.20
MPI-ESM-LR RCP2.6 - 2050	30.98	23.82
MPI-ESM-LR RCP4.5 - 2050	33.65	25.52
MPI-ESM-LR RCP8.5 - 2050	36.61	27.23

Table E.4 | Areas of fire frequency ranges

Scenario	Area (Mha)			
	0-25	**25-50**	**50-75**	**75-100**
Actual 1988-2014	78.867	42.909	10.049	0.691
Historical mean climate	83.593	37.357	9.988	1.568
MIROC5 RCP2.6 - 2050	70.298	40.211	16.619	5.009
MIROC5 RCP4.5 - 2050	66.382	40.474	18.490	6.437
MIROC5 RCP8.5 - 2050	62.292	40.610	20.147	8.195
CanESM2 RCP2.6 - 2050	64.864	41.332	18.965	6.525
CanESM2 RCP4.5 - 2050	60.160	41.467	20.934	8.491
CanESM2 RCP8.5 - 2050	55.633	41.215	22.301	10.963
MPI-ESM-LR RCP2.6 - 2050	64.167	40.401	19.714	7.245
MPI-ESM-LR RCP4.5 - 2050	59.285	40.359	21.384	9.668
MPI-ESM-LR RCP8.5 - 2050	54.569	39.868	22.377	12.577

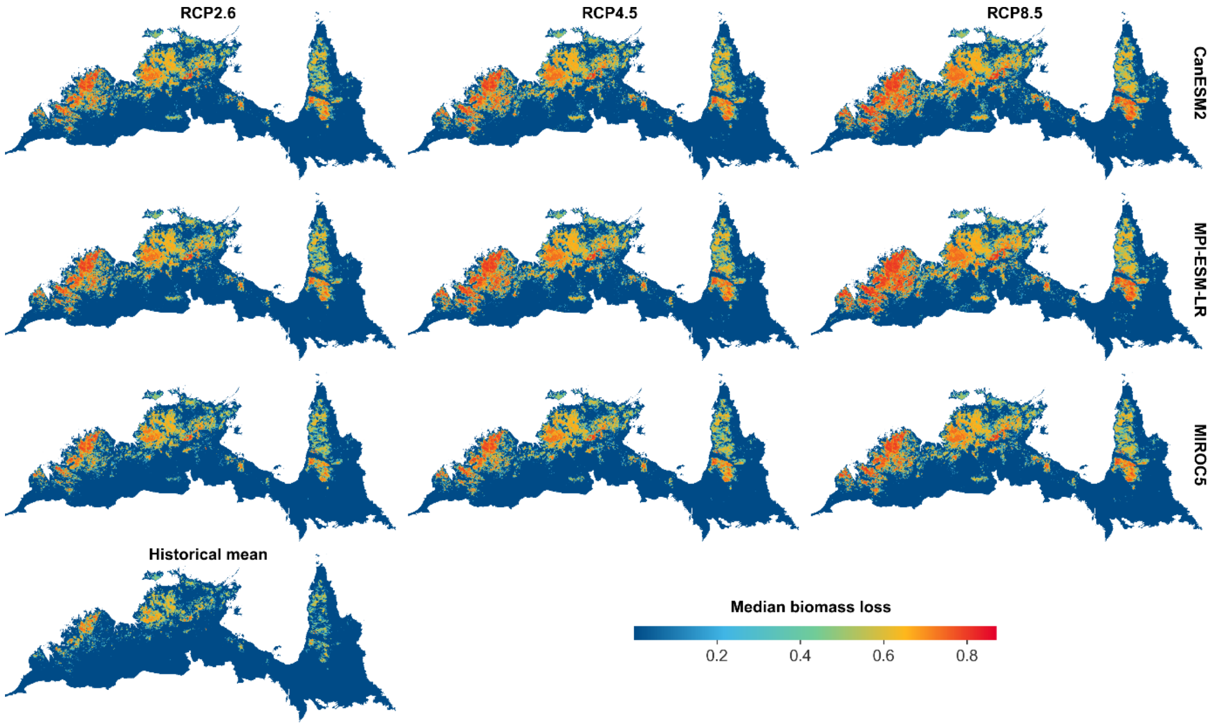

Figure E.8 | Median percentage of biomass lost in 2050 under three different RCPs and GCMs.

Pasture production model

Climate

Historical climate data used in the model was derived from the Bureau of Meteorology's 5 km gridded Australia daily datasets (Jeffrey *et al* 2001) (Figure E.9 and E.10). Daily data was aggregated to monthly, seasonal or annual data for analysis and resampled to 1 km grid cells. Additional summary layers were calculated to use as the historical baseline from which estimates of future climate could be derived. Within the northern Australian study area rainfall across the region is subject to monsoonal patterns of wet and dry with the higher rainfall wet season typically occurring between September and March while the period between April and October is generally dry (Gleeson *et al* 2012).

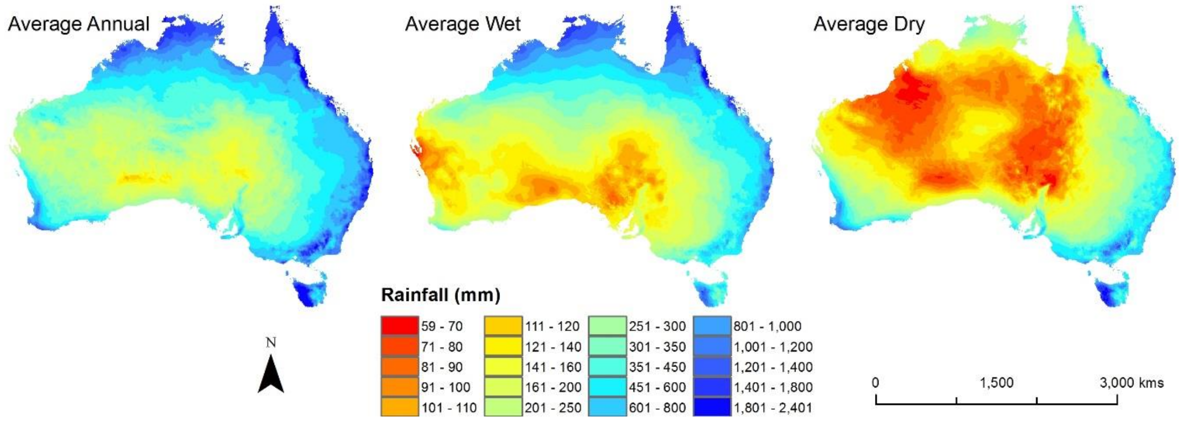

Figure E.9 | Average annual, wet season, and dry season rainfall for Australia (Jeffrey *et al* 2001).

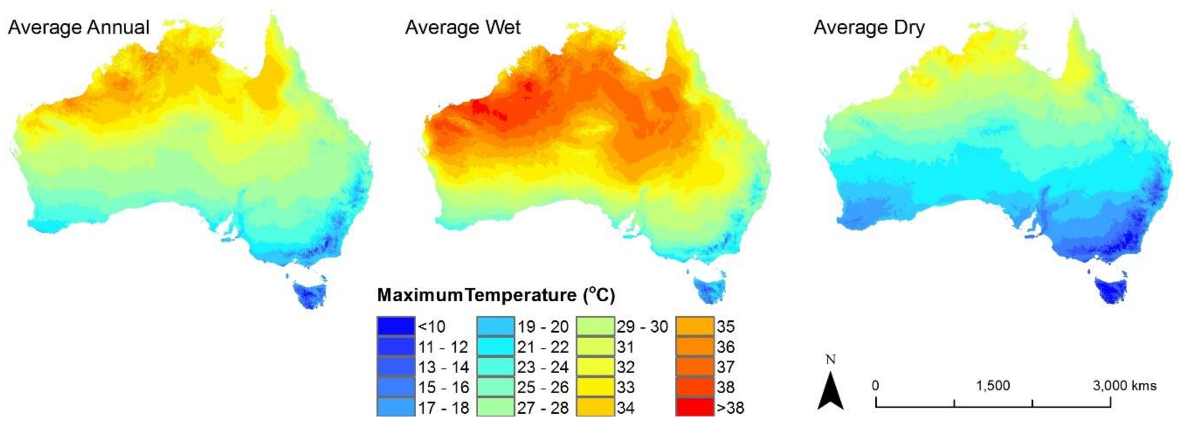

Figure E.10 | Average annual, wet season, and dry season maximum temperature for Australia (Jeffrey *et al* 2001).

Pasture Production Estimation

We used long run data outputs from the AussieGrass pasture production model. This model has been developed by Department of Environment and Resource Management in Queensland and represents the most complete model of pasture production in the Australia. The AussieGrass model is based fundamentally on a point based soil-water balance pasture production model called GRASP. Much like APSIM the GRASP model uses soil and climatic parameters in a plant phenology model to estimate pasture production rates under specified conditions on a daily time step. Within AussieGrass, the GRASP model runs across a 5km by 5km grid covering all of Australia. Outputs are calibrated against values from NOAA's Normalized Difference Vegetation Index (NDVI) and ground-truthed through 600,000 field observations (Stone *et al* 2010). Long run and large scale datasets (as used in this model) are only available at more aggregated sub-IBRA region levels (Australian Government 2012) (Figure E.11).

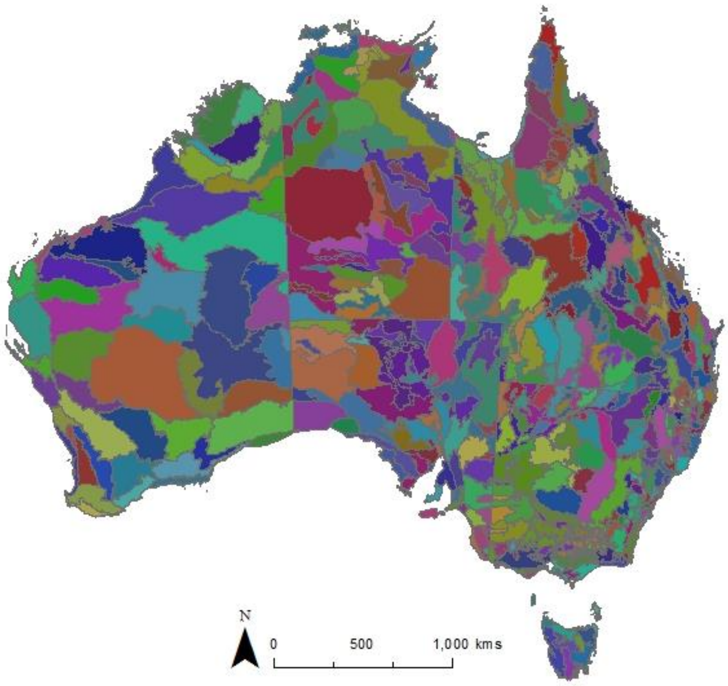

Figure E.11 | Australian IBRA sub-regions (Australian Government 2012).

In total 125 years of monthly pasture growth data based on the historical climate record 1890 to 2015 were obtained and used in the model. AussieGrass model parameters and outputs were provided at the monthly time step and include rainfall, min and max temperatures, evaporation, pasture growth, total standing dry matter, and three safe stocking rates options (% utilization, total cover and eaten) (Table E.5).

Table E.5 | Example data from AussieGrass modelling.

Year	*Month*	*rai*	*max*	*min*	*evap*	*growth*	*tsdm*	*utilization*	*totalcover*	*eaten*
1890	1	267.3	29.6	20.7	5.1	1581.2	4264.9	1.1	89.3	16.7
	2	181.2	30	20.6	5	461.4	4525.8	1.7	91.4	15
	3	367.2	29.9	19.9	4.7	183.3	4481.1	2.2	91.7	13
	4	47.8	27	17.2	4.2	57.2	4308.5	2.9	91.5	14.1
	5	80.7	24.8	14.4	3.3	15	4070.9	3.6	91.4	14.5
	6	27.7	23.3	11.8	2.9	20	3842.5	4	91	9.9
	7	29.1	22.5	9	3.2	5.5	3574.3	4.5	90.8	10.3
	8	4.2	25.1	10	4.1	1.5	3281.6	5	90.5	10.3
	9	56.5	28	13.4	5.5	7.6	2941.9	5.6	90.2	14.5
	10	24.3	31.8	16.8	6.9	27.2	2618.5	92.8	89.5	15
	11	47.2	32	17.6	7.2	90.8	2387.5	43.2	88.8	14.5
	12	75.4	32.6	19.6	6.8	538.6	2592.4	11.9	88.4	18.7
1891	1	288	30.9	21.2	5.4	1526.6	3818.2	3.4	89.2	18.7
	2	223.5	29.2	20	4.7	1165.4	4728.5	2.6	91.4	16.9
…	…	…	…	…	…	…	…	…	…	…
2014	11	5	33.7	20.2	9.3	1.1	802.1	10.1	77.9	19.6
	12	66.4	34.2	21.8	8.3	43.7	678.1	78.7	75.4	23.4

Future climate modelling

Three possible future climate scenarios (RCP 2.6, RCP 4.5, and RCP 8.5) (van Vuuren *et al* 2011, Hatfield-Dodds *et al* 2015) resulting from specified emissions trajectories were modelled through three General Circulation Models (GCM). Each GCM (CanESM2, MPI-ESM, and MIROC5). This produced future climate deltas for rainfall and temperature for each year between 2013 and 2050 at ~1.88^{o} resolution. The mid-points of these data were then interpolated to 1.1 km grid cell resolution using a regularized spline interpolation technique. This approach is an exact interpolator where interpolated values honour the original value at the data point, with a smooth surface in between (continuous first derivative) (Figure E.12). It is important to note that the original climate deltas are an average value for the entire 295km^2 grid cell as modelled in the three climate models. Therefore the interpolation approach has the potential to violate some of the original assumptions/processes used in the climate modelling. However, as high resolution data is necessary to produce a smooth high resolution surface (removing unrealistic sharp spatial edges between very coarse grid cells) the interpolation to climate model error is outweighed by any negative impacts resulting from contravening climate modelling logic.

The historical climate data series carries considerable variability over time and space and while we can generally reproduce the spatial variability there is uncertainty associated with predicting each future year. The climate deltas represent an expected average change for each given location. Future climate prediction in this model assumes average historical climate as a baseline and predicts forwards using the interpolated climate deltas. Each year generates a new mean climate layer for rainfall and temperature to which regression function applied and pasture predicted.

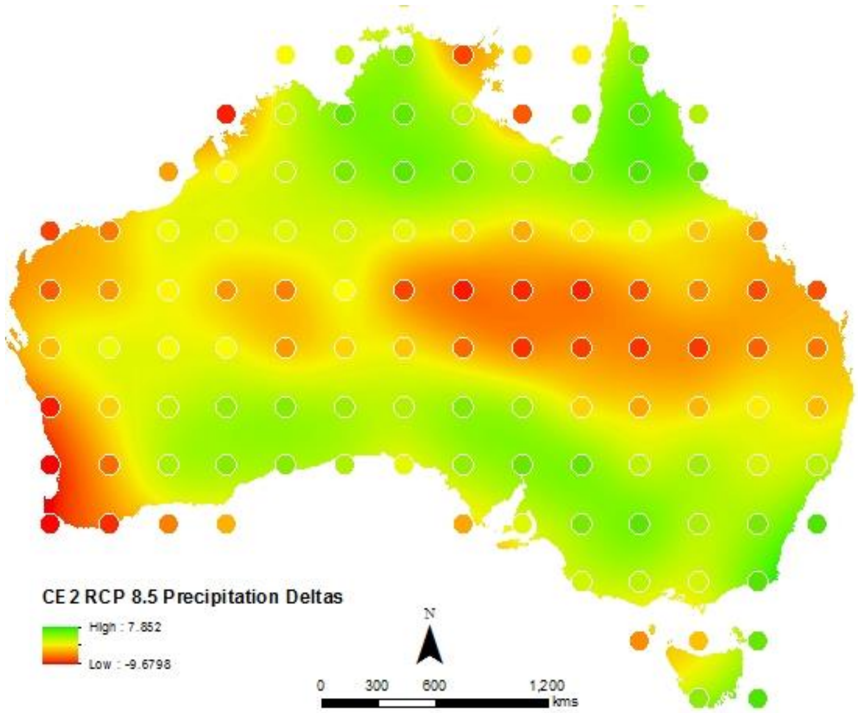

Figure E.12 | An example output of the climate data interpolation technique.

Regression

AussieGrass data from a set of randomly selected locations was examined to explore the relationship between climatic variables and pasture production. The three climate parameters produced in the AussieGrass outputs are rainfall, temperature and evapotranspiration. Scatter plots of model variables for the randomly selected regions provide a first cut indication of any potential correlation between climate parameters and pasture growth (Figure E.13). These scatter plots of indicated a likely relationship between rainfall and pasture and less of a relationship between temperature or evapotranspiration and pasture. In order to identify the drivers of pasture production we tested several regression equations on the sample locations. Three regression approaches (linear, quadratic, General Additive Model) were considered each with a variation of rainfall, temperature and evapotranspiration (Table E.6). Analysis of the regressions returned R-squared values in the range of 0.6 to 0.98 with linear regression exhibiting the best fit using rainfall and maximum temperature as the independent variables (Table E.6). Simulations using this model were closely aligned with actual data (Figure E.14).

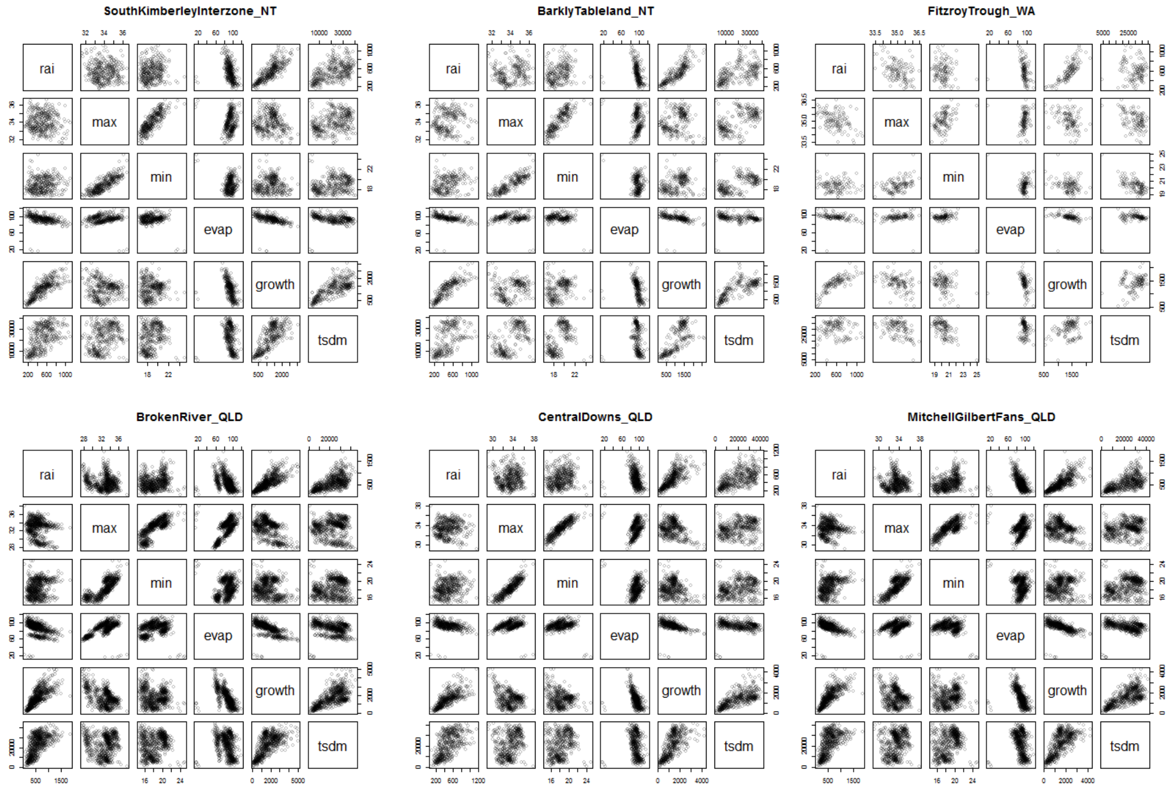

Figure E.13 | Scatter plots of climate and pasture production in six selected sub regions.

Table E.6 | Regression R-squared results for sample locations.

WA	**NT**		**QLD**			
Fitzroy Trough	**Barkly Tableland**	**South Kimberley Interzone**	**Central Downs**	**Mitchell Gilbert Fans**	**Broken River**	**Model**
0.753302	0.766419	0.695247	0.568605	0.676282	0.633264	general additive model of growth and rainfall
0.763583	0.826339	0.695339	0.758941	0.78647	0.809728	general additive model of growth and rainfall + max temp
0.790674	0.793865	0.786059	0.779543	0.796874	0.822701	general additive model of growth and rainfall + evap
0.966094	0.952617	0.944574	0.901814	0.919118	0.906028	linear model of growth and rainfall (intercept removed)
0.98744	0.934126	0.957822	0.94912	0.980617	0.963415	linear model of growth and rainfall + max temp
0.985824	0.952847	0.944576	0.901897	0.921319	0.906029	linear model of growth and rainfall + evap
0.653982	0.716879	0.659891	0.54238	0.623092	0.581039	linear model of growth and rainfall + quadratic rainfall
0.661594	0.729609	0.660056	0.648834	0.67925	0.741891	linear model of growth and rainfall + quadratic max temp
0.654612	0.722806	0.689872	0.604389	0.667123	0.704259	linear model of growth and rainfall + quadratic evap

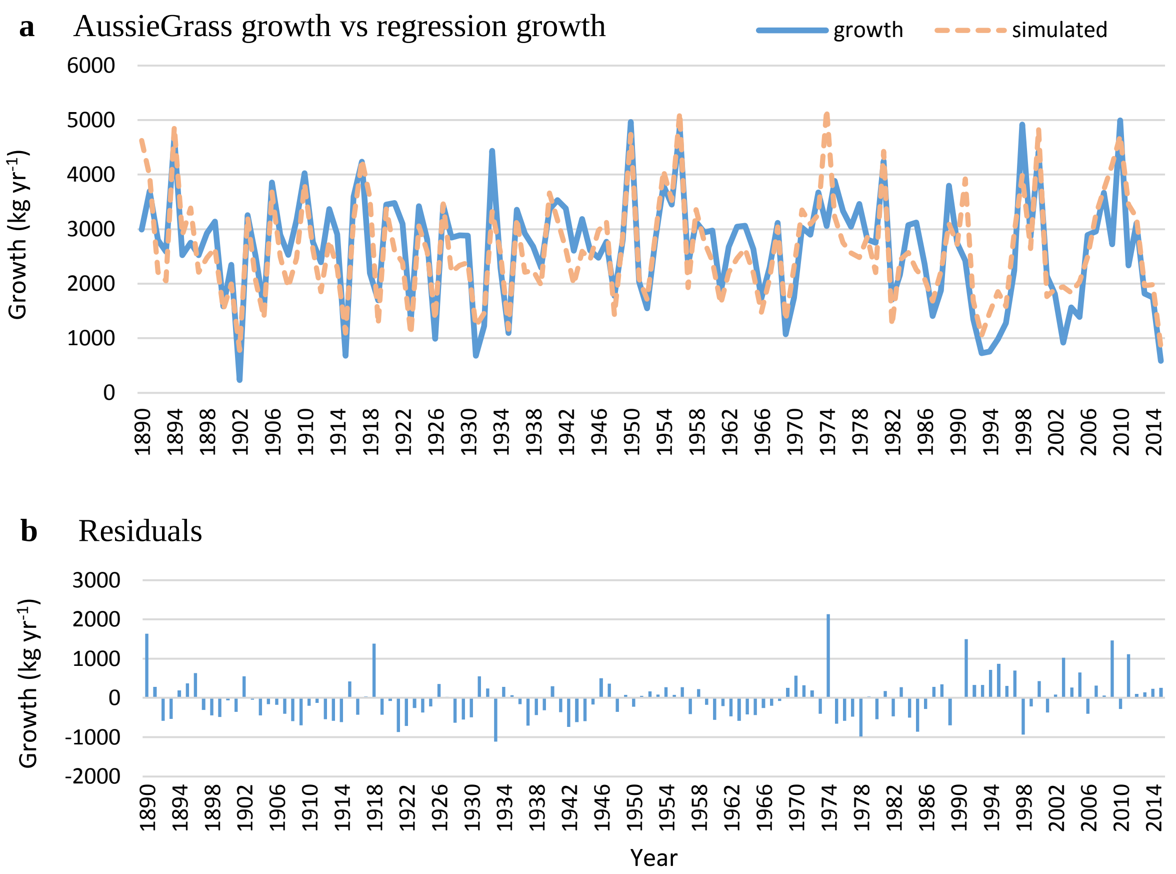

Figure E.14 | Comparison of AussieGrass pasture production data and growth (a) simulated via regression equation with residuals (b) for each year in the Broken Riven sub region.

Results

Simulated pasture production values across the study area ranged from. 0.1 to 4.5 Mg ha^{-1} yr^{-1} although approximately 70% of the area produces between 1.5 and 3 Mg ha^{-1} yr^{-1}. Coastal areas were consistently more productive than inland reflecting the higher rainfall near the coast (Figure E.15 and E.16). Climate change effects on pasture production are negative under all scenarios and GCMs. Mean declines in production included 124 (CE2), 126 (MPI) and 74 (MR5) kg ha^{-1} yr^{-1} for the RCP 2.6 between 2013 and 2050. RCP 4.5 produced reductions of 161 (CE2), 163 (MPI) and 98 (MR5) kg ha^{-1} yr^{-1} while the worst case scenario RCP 8.5 resulted in 193 (CE2), 197 (MPI) and 121 (MR5) kg ha^{-1} yr^{-1} reductions (Figure E.16).

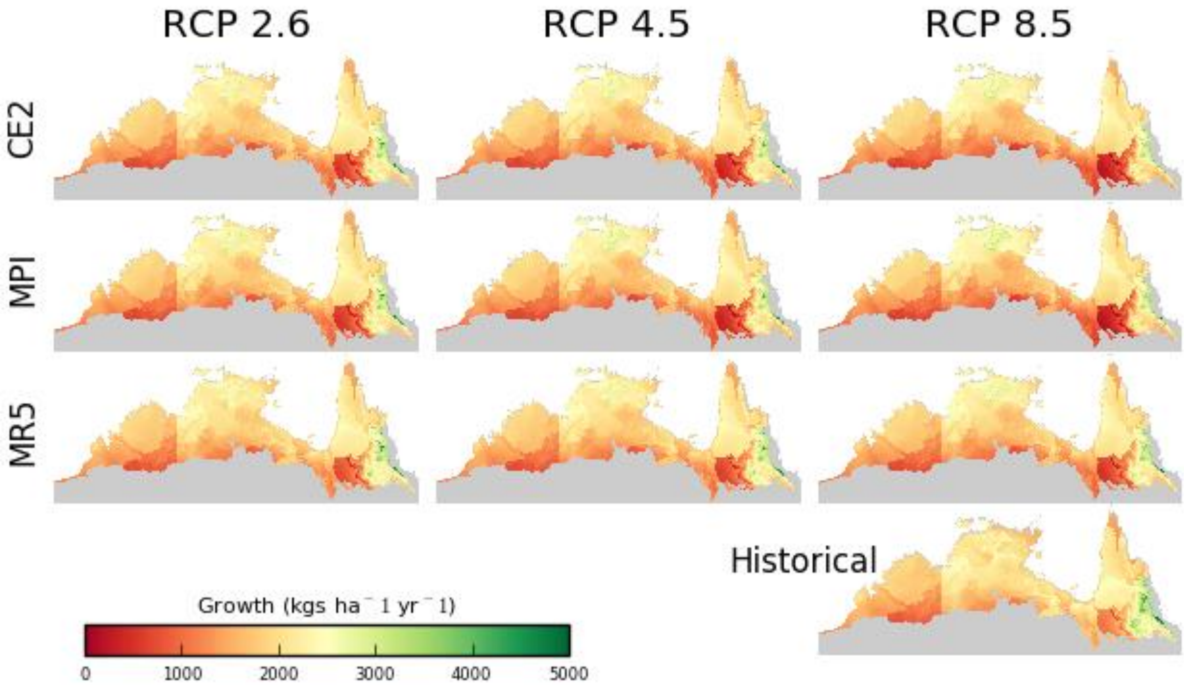

Figure E.15 | Pasture growth in (kg ha^{-1} yr^{-1}) under historical climate and each scenario and GCM in the year 2050.

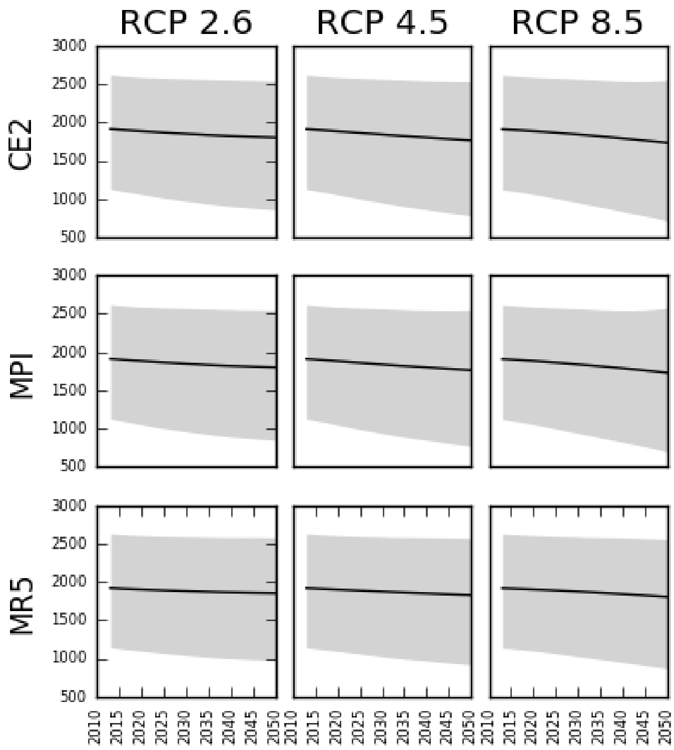

Figure E.16 | Mean pasture production (kg ha^{-1} yr^{-1}) across all locations for each scenario, GCM and future year with 5th and 95th percentile range in grey.

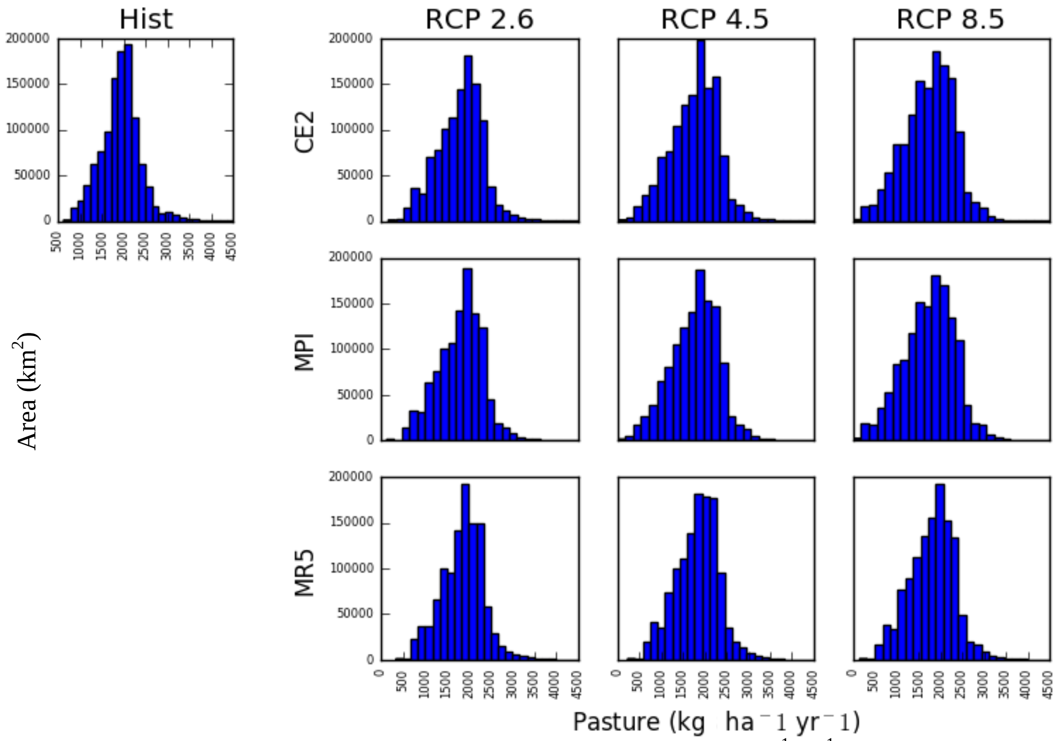

Figure E.17 | Histograms of total area of pasture production rates (kg ha^{-1} yr^{-1}) under historic conditions and for each scenario and GCM at the year 2050.

Supplementary Results

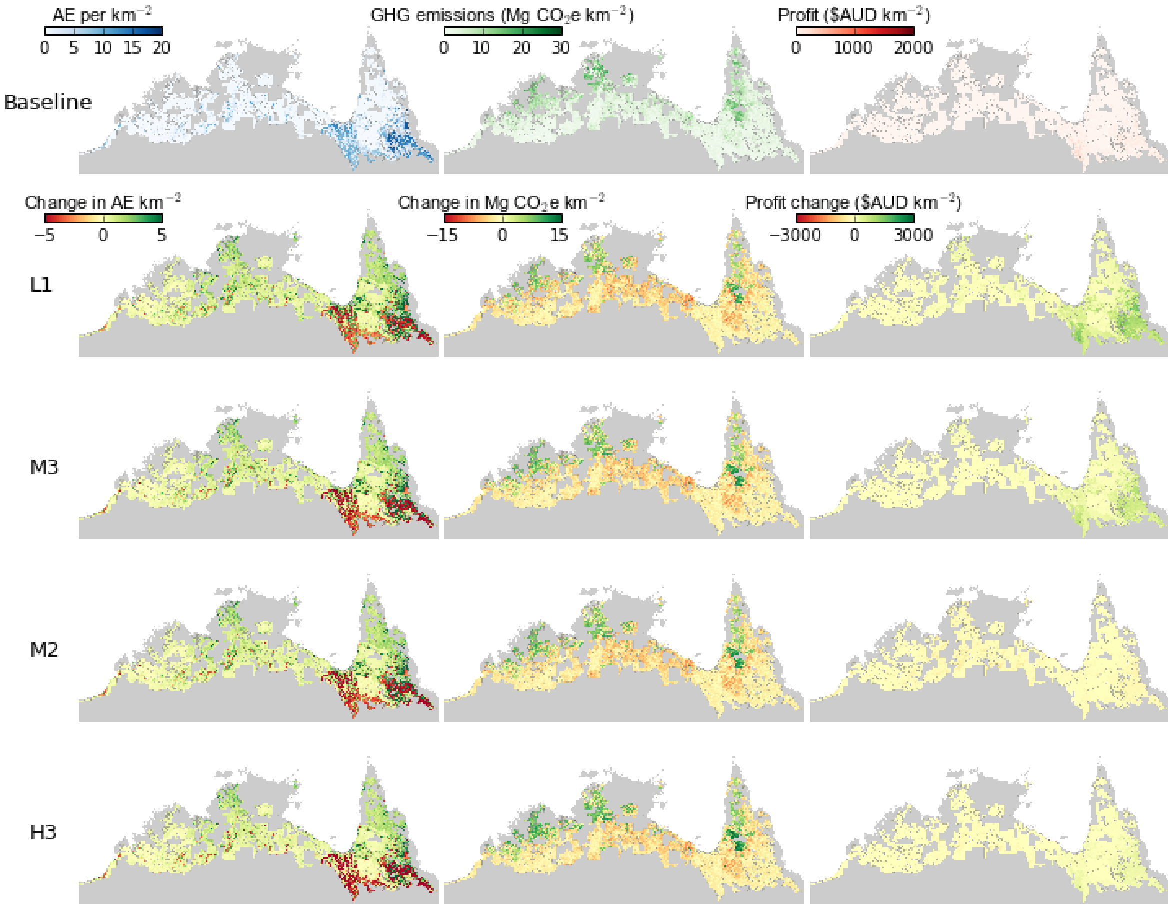

Figure E.18 | The lower bound of outcomes for safe stocking rates under global change scenarios to 2050. The lower estimates for the baselines of livestock, greenhouse gas emissions, and profit are shown in the top row for the 'safe stocking' management action (safe stocking rates without controlled burning). These include the impact of the most severe fire (95th percentile). The remaining rows show the lower bound of change by 2050 in each outcome under the global outlooks. GHG emissions include emissions from both wildfire and livestock as there was no action to control fire.

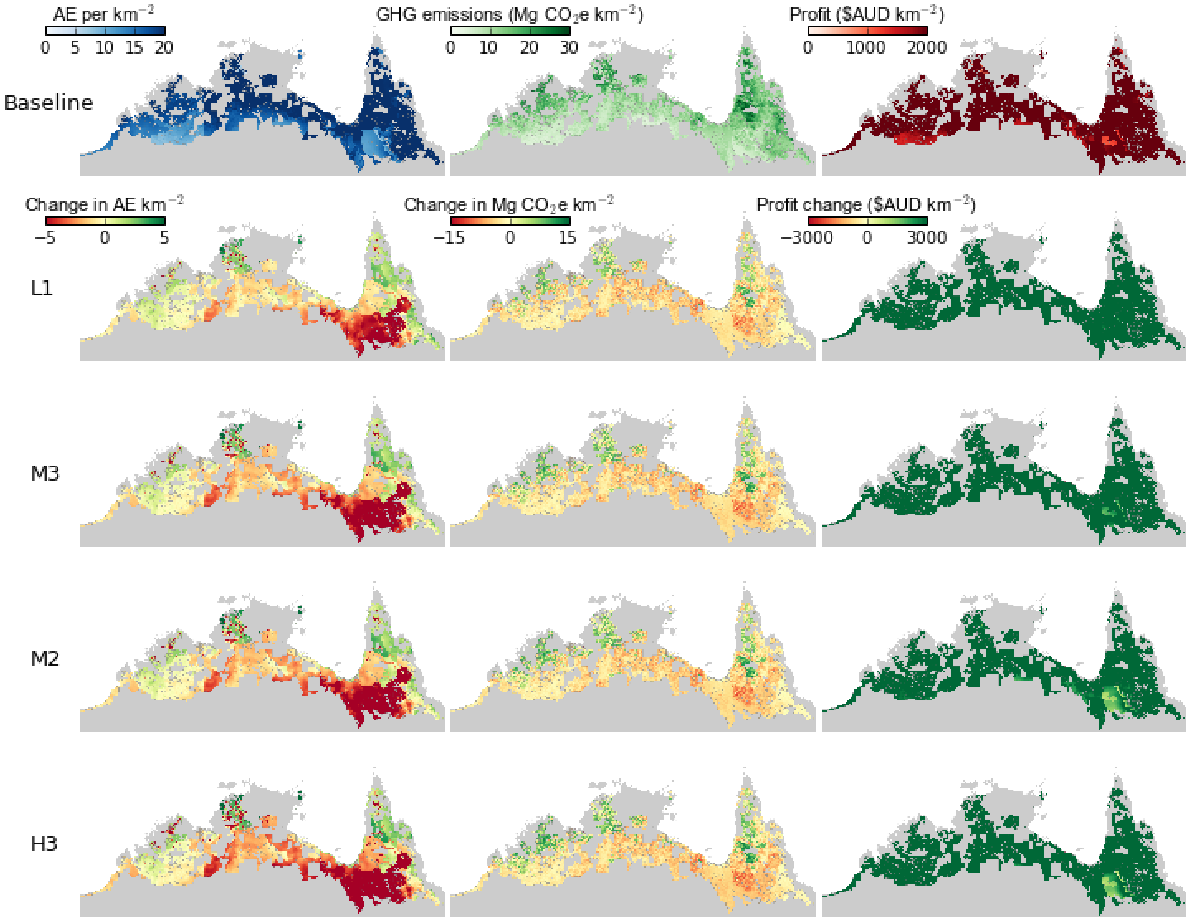

Figure E.19 | The upper bound of outcomes for safe stocking rates under global change scenarios to 2050. The upper estimates for the baselines of livestock, greenhouse gas emissions, and profit are shown in the top row for the 'safe stocking' management action (safe stocking rates without controlled burning). These include the impact of the least severe fire (5th percentile). The remaining rows show the upper level of change by 2050 in each outcome under the global outlooks. GHG emissions include emissions from both wildfire and livestock as there was no action to control fire.

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