THE UNIVERSITY OF QUEENSLAND

AUSTRALIA

Applying ecosystem services thinking to natural resource management and conservation decision making

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BSc, MSc

A thesis submitted for the degree of Doctor of Philosophy at

The University of Queensland in 2016

School of Biological Sciences

Abstract

As global consumption increases, there is a growing emphasis on the production of food and the use of other resources necessary for life. Consequently, many ecosystems are stressed because their ability to produce market goods is favoured over other critical functions and services that ecosystems provide such as energy transfer, water regulation, nutrient filtering, and carbon sequestration. Capturing these benefits using ecosystem services thinking offers decision makers a methodology for considering the multiple benefits that ecosystems provide. However, gaps in our understanding of how we can make the ecosystem services concept operational remain.

Recognising the relationship between natural capital stocks and the provision of ecosystem processes and services is a crucial step in operationalising ecosystem services thinking. I advance this concept by identifying that attributes of natural capital are not uniform in their amenability to change. Hence, the central tenet of this thesis is that management actions targeted at manageable attributes of natural capital stocks is effective for influencing provision of ecosystem services and benefits.

I test how management practices influence natural capital stocks that contribute to the provision of required ecosystem services, using a 'provider group' approach. Provider groups are sets of species which exhibit attributes which contribute to ecosystem services and benefits. The traditionally farmed grassland system in the Southeastern Carpathians, Romania, is a good example of a multifunctional landscape providing provisioning, regulating, and cultural ecosystem services and thus a useful case study to test this approach. I assigned grassland plant species to provider groups (quality fodder, medical and aromatic compounds, honey, pollen, nitrogen fixation, and conservation concern) based on their characteristics and tested the impact of management practices (abandonment of hay meadows, grazing, and mowing) on species diversity and abundance within each group. Over three quarters (77%) of the 210 unique species sampled during this study contributed to at least one provider group and over a third (36%) contributed to more than one group. I found that different management practices favour certain provider groups over others, and thus supply of certain ecosystem services over others. A more nuanced understanding of the influence of management practices on natural capital stocks can better inform agricultural and conservation policies targeted at sustaining multifunctional landscapes.

Incorporation of social data, particularly that describing human behaviour and decision making, is critical to embed the ecosystem services concept into natural resource management policy and

practice. Riparian management is a common policy option for mitigating the externalities of land use. A riparian management programme has been running in Taranaki Region, New Zealand for over 20 years providing a useful case study to elicit farmers perceptions and experiences of the pros and cons of planting riparian margins. I found the views of dairy farmers farming the Taranaki ring plain to be varied. Farmers with planted margins reported experiencing many on and off-farm benefits from multi-tier riparian plantings including production, environmental, and social values. This group of farmers identified 32 aspects of riparian vegetation across nine categories, 65% of which were positive aspects and 35% of which were negative aspects. Farmers who had fenced but not planted their riparian margins also believe benefits for water quality, animal safety, and farm management can be achieved from fenced grass strip riparian margins but were less convinced about additional benefits from planting. This group of farmers identified 15 aspects of riparian vegetation across four categories, all of which (100%) were negative aspects. Recognising that farmers' perceptions and/or experiences vary can help inform how best to structure and deliver policies for sustaining provision of multiple ecosystem services and benefits.

Biodiversity offsetting represents a critical application of the ecosystem services concept as trading biodiversity also inherently trades the associated ecosystem service values. Further, trading biodiversity in an offset exchange embodies the manipulation of natural capital stocks, in both the removal of species and habitats and in their replacement or enhancement elsewhere. Currencies used to evaluate offset proposals can either aggregate (combine measures of biodiversity attributes into a composite unit) or disaggregate (individually account for each measured biodiversity attribute of interest). I developed a disaggregated accounting model that balances like-for-like biodiversity trades using a suite of area by condition currencies to individually calculate the net present biodiversity value (NPBV) by which to evaluate no net loss for each biodiversity attribute inputted into the model. The model improves on more aggregated models by enabling increased transparency of biodiversity offsetting proposals, and thus improved decision making processes.

This thesis provides an increased understanding of the relationship between management actions and ecosystem services and associated benefits at local scales, and a collection of tools and methods to support decision making targeted at sustaining multifunctional landscapes. Overall, this research illustrates that a natural capital focussed ecosystem services approach provides an opportunity to shift land management towards practices that sustain rather than deplete the natural capacity of ecosystems.

Declaration by author

This thesis is composed of my original work, and contains no material previously published or written by another person except where d5ue 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.

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Publications during candidature

Peer-reviewed papers

Maseyk FJF, Mackay AD, Possingham HP, Dominati EJ, Buckley YM 2016. Managing natural capital stocks for the provision of ecosystem services. Conservation Letters doi:10.1111/conl.12242.

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Publications included in this thesis

Maseyk FJF, Mackay AD, Possingham HP, Dominati EJ, Buckley YM 2016. Managing natural capital stocks for the provision of ecosystem services. Conservation Letters. doi:10.1111/conl.12242. — incorporated as Chapter 2.

Contributor	Statement of contribution
Fleur JF Maseyk (Candidate)	Idea generation and development (55%) Mathematical formulation (10%) Wrote and edited the paper (65%) Figure generation (85%)
Alec D Mackay	Idea generation and development (20%) Editorial contributions to the paper (15%) Figure generation (5%)
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Contributions by others to the thesis

The majority of the thesis was conceived, developed, and written by the candidate Fleur Maseyk. Others made contributions as detailed below.

Chapter 2: The conceptual framework central to this chapter was developed by the candidate with substantial contributions by Alec Mackay (supervisor) and Estelle Dominati. Hugh Possingham (supervisor) and Yvonne Buckley (supervisor), with involvement from the candidate, generated the equations. Figures were created by the candidate. The soil related components of Table 2.1 were taken from previous work of Estelle Dominati, the remainder of the table was developed by the candidate with input from Alec Mackay, Estelle Dominati, and Yvonne Buckley. The chapter was written by the candidate with editorial contributions from all involved.

Chapter 3: The candidate, Anna Csergő and Yvonne Buckley developed the chapter idea. Further refinement was lead by the candidate with contributions from all authors. Experimental design and field work was conducted by Anna Csergő and László Demeter with assistance from Alpár Kelemen and Gabriella Péter. Statistical analysis was conducted by Anna Csergő, with input from Yvonne Buckley. Figure 3.2 was created by László Demeter, Anna Csergő and the candidate. Figures 3.3, 3.4, and 3.5 were created by Anna Csergő. The candidate provided input into Figures 3.3–3.5 and created all remaining graphics. The majority of the chapter was written by the candidate with contributions from Anna Csergő, especially the methods and results sections. All authors made editorial contributions.

Chapter 4: The initial idea for this chapter was generated by Alec Mackay but further defined and developed by the candidate, with input from Alec Mackay, Estelle Dominati, Toni White, and Don Shearman. The structure and agenda for the workshops were designed by the candidate with input from all others involved in this chapter. The workshop logistics were organised by Don Shearman with assistance from the candidate. Toni White facilitated the workshops. Personal invitations to farmers to attend the workshops were extended by Land Management Officers of the Taranaki Regional Council. Data analysis was conducted by the candidate. The chapter was written by the candidate with editorial contributions from all others involved. Figure 4.1 was created by Michelle Baker. All other figures were created by the candidate. The workshop summary report distributed to participant farmers was written by the candidate with minor editorial input from Alec Mackay and Don Shearman. An information leaflet outlining the research findings for distribution amongst the wider farming community was written and produced by the candidate.

Chapter 5: The biodiversity accounting model described in this chapter was developed under contract with the New Zealand Department of Conservation (DOC). The basic parameters and requirements for the model and initial idea generation were provided by DOC as a component of the project scope, but further developed by the candidate, Laurence Barea, and Martine Maron. Development of the product was entirely done by the candidate. Review of mathematical formulation was undertaken by Hugh Possingham. Andrew Maseyk provided technical assistance in writing Excel scripts. Guy Dutson and Martine Maron provided review of the model functionality. The writing of the chapter was undertaken by the candidate. Laurence Barea, Martine Maron, and Theo Stephens all made important editorial contributions, with Hugh Possingham and Guy Dutson also providing editorial input. Figures were created by the candidate. This contract also involved the production of a user manual, the majority of the writing of which was undertaken by the candidate. Martine Maron wrote a section and contributed to editing. Richard Seaton and Alistair Beveridge contributed to scenario development. Guy Dutson reviewed the manual and contributed to editing.

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

None.

Acknowledgements

There is a large cast of people to whom I owe my thanks and appreciation for their support and assistance and who collectively made this a better thesis, but firstly I am extremely grateful to my funding sources for keeping me in the black. I received funding from an Australian Postgraduate Award; Australian Research Council Centre of Excellence for Environmental Decisions (CEED)/National Environmental Research Programme (NERP) Top-Up Scholarship; and the New Zealand Ministry of Business, Innovation and Employment funded research partnership project: Building biodiversity into an ecosystem service-based approach for resource management (BEST).

To Alec Mackay, Yvonne Buckley, and Hugh Possingham my sincere thanks for your superb supervisory support which made my PhD journey a thoroughly enjoyable and rewarding one. All three provided that magic combination of autonomy, guidance, and humour in just the right amounts, and I am extremely grateful for the space this created without any of the loneliness. You are an inspirational trio and I have massively benefited from your influence these past three and a half years. My thanks and gratitude to Alec, for your generosity with your time, endless positivity, and for never letting me rest for risk of missing an opportunity to push a bit further; to Yvonne, for your persuasive encouragement to sharpen my thinking and raise the bar and for some of the all time best pieces of life advice; and to Hugh, for welcoming me into the lab despite my limited mathematical and bird watching skills, our chats that were always off-topic but nonetheless left me contemplative, and for creating and leading such an empowering, supportive and dynamic lab environment. Although my thesis headed off in an entirely different direction, I am also grateful to Marit Kragt for early conversations which helped shape my thinking. Thanks also to my readers Kerrie Wilson and John Dwyer for your gentle guidance along the way.

I could not have completed this thesis without the substantial support from Taranaki Regional Council and their generosity in sharing their data. I am extremely grateful to Don Shearman for his initial interest, substantial time investment, and continued enthusiasm for this project, to his team of Land Management Officers for facilitating engagement with farmers, and to Claire Robertson for taking me into the field for the day. Alex Connolly for overseeing my sizable data request and Chris Lambert for generating GIS data. My thanks also to Stephen Hall for supporting this project and to Stephen, Don, and Gary Bedford for their constructive comment on the manuscript drafts. I am extremely grateful to the dairy farmers of Taranaki who generously donated their time to participate in this study, and so openly shared their views. Taranaki Regional Council and AgResearch both provided financial support to host workshops with farmers.

I thank Anna Csergő and László Demeter for sharing their Southeastern Carpathian grassland field data with me and showing me a part of the world so different from my own. To the Department of Conservation for the opportunity to work on the offsets accounting model for them and especially to Laurence Barea for his facilitation of and input into the project. To my non-supervisor co-authors: Estelle Dominati; Anna Csergő; László Demeter; Toni White; Laurence Barea; Theo Stephens; Guy Dutson; and Martine Maron who provided not just editorial contributions but also intellectual challenge and helped shape and improve my thinking.

I was extremely lucky to do my PhD under the umbrella of the Centre for Biodiversity and Conservation Science and the ARC Centre of Excellence for Environmental Decisions which provided such a dynamic, positive, and exciting place to study and I am grateful for the many and varied opportunities being part of CBCS and CEED offered me. Special mention to the post-docs for their general willingness to help PhD students and to the admin team without whom we would be utterly lost. To the occupants of Room 508 past and present — thanks for your camaraderie and the good times.

They may not have realised I had adopted them as my post-doc mentors, but to Danielle Shanahan, Nathalie Butt, and Estelle Dominati who have each ploughed an inspirational path for me to follow — thanks for your wisdom, good humour, and friendship.

I am indebted to Greg Carlyon for his tireless belief in and support of me over the last decade. It is impossible to define the myriad of opportunities Greg creates for the people around him, but that I have personally benefited on numerous occasions is certain. To the rest of The Catalyst Group who are as much friends as colleagues, particularly: Alistair Beveridge, Maree Gurney, Peter Taylor, and Kate McArthur my endless thanks for your support in all its forms. It is both humbling and a privilege to be associated with such a passionate, fearless, and creative team who are utterly committed to fighting the good fight.

I owe a great deal of my success and happiness to my family who have always, and remain, supportive and encouraging of all my endeavours and to Rich Seaton who always has my back and whom I thank for the encouragement to start, the support to finish, and everything in between.

Keywords

Biodiversity offsets, conservation, decision making, ecosystem services, land management, multifunctionality, natural resource management, policy

Australian and New Zealand Standard Research Classifications (ANZSRC)

ANZSRC code: 050209 Natural Resource Management 60%
ANZSRC code: 050202 Conservation & Biodiversity 40%

Fields of Research (FoR) Classification

FoR code: 0502 Environmental Science and Management 100%

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ABBREVIATIONS

ANOVA	Analysis of variance
BBN	Bayesian Belief Network
BBOP	Business and Biodiversity Offsets Programme
BV	Biodiversity value
CAP	Common Agricultural Policy
DEFRA	Department for Environment Food & Rural Affairs (United Kingdom)
DOC	Department of Conservation (New Zealand)
EPBC	Environmental Protection and Biodiversity Conservation Act 1999 (Australia)
EU	European Union
IP	Intellectual property
IUCN	International Union for Conservation of Nature
LMM	Linear mixed effect models
LMO	Land Management Officer
MEA	Millennium Ecosystem Assessment
NPBV	Net present biodiversity value
RFWP	Regional Freshwater Plan
TRC	Taranaki Regional Council
UK	United Kingdom

CHAPTER ONE:

INTRODUCTION

Contextual background

Nature has long been known to provide the goods and services (e.g. nutrition, energy, raw materials, regulation of the physical environment) that humans depend on for survival and wellbeing (Schumacher 1974). As the global population grows, there is an increasing emphasis on the production of food and other resources necessary for life. Increases in per-capita consumption also intensify human impacts on ecosystems (Holdren & Ehrlich 1974). Consequently, biodiversity has been much reduced globally and many ecosystems are being pushed beyond their inherent capacity to provide necessary ecosystem services (MEA (Millennium Ecosystem Assessment) 2005; Rockström *et al.* 2009; Steffen *et al.* 2015). This is occurring because the ability to produce goods is typically favoured over other critical functions that ecosystems provide such as energy transfer, water regulation, nutrient filtering, and carbon sequestration (Seppelt *et al.* 2012; Ausseil *et al.* 2013). This has had long-term consequences, including continued habitat and species loss, global deterioration in ecosystem function, and intensified human pressure on finite natural capital stocks (MEA 2005; Mace *et al.* 2015). Previous policies and actions have clearly fallen short of sustainable resource management and conservation objectives indicating a need for practice-change.

Where benefits are not well defined they are easily undervalued (Figure 1.1) or not accounted for at all (externalised) within decision making processes (Wallace 2007; Helm & Hepburn 2012; Robinson *et al.* 2013). This results in decision making that risks depleting rather than sustaining the elements that provide benefits (e.g. natural capital stocks such as soil, water, vegetation, biodiversity etc.) (Stephens *et al.* 2002; Bristow *et al.* 2010).

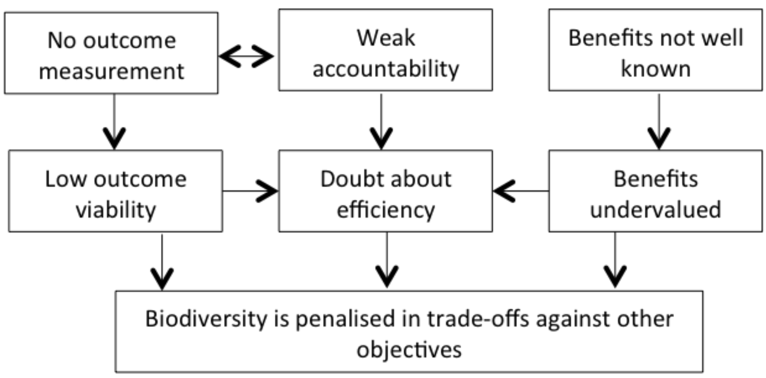


Figure 1.1: Diagrammatic representation of the externalisation of biodiversity in decision making. Adapted from Stephens *et al.* (2002).

The ecosystem services concept is based upon an interdisciplinary view of ecological systems and functions, the social systems and human wellbeings that benefit from nature, and the decision making systems and agencies by which sustainable use of resources can be achieved. Thus the concept is informed by the natural sciences, economics, social sciences, natural resource management concepts, and ethical considerations. By capitalising on this more holistic, interdisciplinary thinking, an ecosystem services approach not only aims to better inform the linkages between science, policy, and resource management practices, but can serve to broaden the conversation by incorporating a wider raft of values into the decision making process.

However, there remains a high degree of ecological and social uncertainty regarding the complex relationships between generation and flow of ecosystem services. Crucially, these complexities have resulted in a confused understanding of ecosystem services which limits the effective integration of the concept into resource management policy and practice (Wallace 2007). For example, in Australia, while 'ecosystem services' appears widely in policy and strategy documents and the concept has been embedded within some collaborative attempts to resolve natural resource management issues, the full suite of ecosystem services and benefits have yet to be fully integrated into natural resource management decision making (Pittock *et al.* 2012). The potential for improved outcomes that an ecosystem services approach may offer risks being lost via misinformed application. Here I argue that the orientation of the ecosystem services concept towards the management of natural capital stocks, rather than ecosystem services *per se*, enables greater potential to effectively influence the sustained provision of ecosystem services, while providing an alternative approach to resource management that sustains the inherent capacity of systems rather than depletes natural capital stocks.

What is natural capital and why is it important?

Capital is "*a stock that yields a flow of valuable goods or services into the future*" (Costanza & Daly 1992). *Natural capital* includes all abiotic and biotic elements of ecosystems including all the physical, biological, and chemical processes (Mace *et al.* 2015), and *natural capital stocks* are the resources from which flow the benefits (natural income) upon which human survival is dependent (Costanza & Daly 1992). Thus, ecosystem services and natural capital are not the same, but rather ecosystem services flow from and are reliant on natural capital. Collectively, natural capital and ecosystem services and the interconnections within and between them can be considered as ecological infrastructure, which like built infrastructure (e.g. power stations, roads, telecommunication cables, health systems etc.) requires investment for maintenance and capacity building (Bristow *et al.* 2010).

It is not uncommon for assessments of ecosystem services to rely on aggregated measures of natural capital or biodiversity, such as land cover, or broad-scale management measures e.g. land use as proxies for ecosystem service provision (Martinez-Harms *et al.* 2015). However, such measures are imperfect surrogates for the assessment of ecosystem services delivery (Lavorel *et al.* 2011). Explicitly recognising how ecosystem services are produced from natural capital is critical in order for the implementation of an ecosystem services approach to be more successful — not doing so risks underestimating their benefits (Figure 1.1). Critically, polices targeted at and reporting solely on ecosystem services (rather than how they are produced) increase the risk that declines in natural capital can go unnoticed (Robinson *et al.* 2013). Neglectful management of natural capital jeopardises our capacity to sustain the provision of ecosystem services into the future.

In this thesis I argue that greater understanding of the relationship between natural capital, other forms of capital (e.g. social and built capital), and ecosystem services will enable management interventions to be targeted where they are likely to have the most impact.

The ecosystem services concept

Ecosystem services were described in the Millennium Ecosystem Assessment (MEA) as the "*the benefits of nature to humans*" (MEA 2005). However, as the concept has evolved so too have the definitions used to reflect varying emphasis on either economics or ecology (Braat & de Groot 2012). Despite the lack of a singular definition, all interpretations capture the essential reliance of human survival and wellbeing on nature as a core principle of the ecosystem services concept.

In this thesis I define ecosystem services as "*the benefits flowing from natural capital stocks consumed or used by humans to sustain or advance wellbeing*" in order to reflect the critical role of natural capital stocks. Here I recognise three categories of ecosystem services: provisioning services (e.g. food supply, raw materials, structural support); regulating services (e.g. mitigated storm impacts, water quality, stable climate); and cultural services (e.g. spiritual, amenity values, recreation, presence of significant species and habitats). Other frameworks and authors include a fourth category of services (e.g. 'supporting services' (MEA 2005; Kumar 2010) or 'intermediate services' (Mace *et al.* 2011)) which I consider to capture ecosystem processes (e.g. pollination, erosion, sediment retention) rather than services. I make this differentiation as ecosystem processes are not services, but are instead the pathways that enable ecosystem functions and which depend on, and build, natural capital stocks. For example, pollinators and floral resources are *natural capital stocks* that enable pollination, an *ecosystem process* that interacts with natural and other forms of capital and ecosystem processes to give rise to food production, a *provisioning ecosystem service*, the *benefit* to humans of which is nourishment and survival. This distinction between ecosystem processes and ecosystem services aligns with several other studies (de Groot *et al.* 2002; Boyd & Banzhaf 2007; Wallace 2007; Dominati *et al.* 2010; Haines-Young & Potschin 2011; Braat & de Groot 2012).

There are numerous in-depth syntheses and reviews of the ecosystem services literature within and spanning the various associated disciplines (e.g. Farber *et al.* 2002; Jack *et al.* 2008; Dick *et al.* 2011; Braat & de Groot 2012; Milcu *et al.* 2013; Howe *et al.* 2014; Martinez-Harms *et al.* 2015). While it is not my intention to replicate such reviews here, it is useful to reflect on the genesis and consequent growth in popularity of the ecosystem services concept, and the opportunities and liabilities it brings with it.

Here I argue that the ecosystem services concept represents an opportunity for a more holistic approach to natural resource management and conservation decision making. A holistic 'systems' approach is not new, nor is the recognition of the social realm in managing natural systems (e.g. Leopold 1949), nor indeed that human societies were reliant on nature (Mooney & Ehrlich 1997). However, the explicit reliance on both natural and social sciences in decision making is an important component of ecosystem services thinking. The term 'ecosystem services' itself first appeared in the early 1980s (Ehrlich & Ehrlich 1981) although prior to and around this time, several authors began to describe ecological concerns with economic terms. Indeed, the modern history of the ecosystem services paradigm grew out of an aim to frame ecosystem function in terms of benefit for humans as an 'economic service' in order to increase public interest in biodiversity conservation (Braat & de Groot 2012) and the need for considered management of our natural assets to sustain life. Building on this early history, the high-profile publication '*The value of the world's ecosystem services and natural capital'* (Costanza *et al.* 1997) and the conceptual frameworks and topology promoted by the Millennium Ecosystem Assessment (MEA 2005) rapidly elevated the ecosystem services concept to the attention of policy-makers internationally.

Numerous organisations and collaborations have emerged since the Millennium Ecosystem Assessment[1] and the ecosystem services concept has growing interest within government agencies (e.g. Cork *et al.* 2007; Roberts *et al.* 2015), and non-governmental organisations globally (e.g. The Natural Conservancy Trust[2] and the Wildlife Conservation Society[3]). Protected area systems globally are diversifying their focus to incorporate a broader range of objectives including ecosystem services (Watson *et al.* 2014). The literature pertaining to ecosystem services across multiple disciplines is substantial (e.g. over 6,500 papers containing the words 'ecosystem services' or 'ecosystem goods' were published between 2003–2013, Martinez-Harms *et al.* 2015) and 'ecosystem services' is now considered a discipline in its own right as reflected in the establishment of a peer reviewed journal pertaining solely to ecosystem services (Braat & de Groot 2012). The concept has been heralded as an evolving paradigm that now *"pervades all current discourse about the environment"* (Robinson *et al.* 2013). The growing interest in ecosystem services includes efforts to better understand how the concept may be used to inform decision making, and attempts to shift natural resource management practices towards those that sustain natural capital stocks and provide for multiple functions and services.

However, there have been no shortage of criticisms of the ecosystem services concept (e.g. McCauley 2006; Redford & Adams 2009; Silvertown 2015). A valuable synthesis of these criticisms and the counter-arguments is provided by Schröter *et al.* (2014) who categorise the debate into three types: ethical considerations (relating to how humans interact with nature); nature conservation and sustainable use of ecosystems (the science-policy interface); and ecosystem services as a scientific approach. Although I will not attempt to rehash what has been succinctly dealt with elsewhere (i.e. Schröter *et al.* 2014), it is useful to highlight some of the main arguments here as they pertain to my thesis. Further, I suggest that the counter-arguments to the critique reinforce the ecosystem services concept as a worthwhile addition to current decision making practices. In this section I have focused on criticisms that fall within the science-policy interface category, specifically: the quantification and valuations of nature and the ethical challenges.

[1] For example: Ecosystem Services Project (an Australian collaborative natural resource management project); The Ecosystem Services Partnership (ESP), a worldwide network to enhance the science and practical application of ecosystem services assessment; Natural Capital Project, an international partnership aligning economic forces with conservation; Project for Ecosystem Services, a Global Environment Fund funded umbrella project aimed at integration of ecosystem services approaches into natural resource management and decision making; Towards a Common International Classification of Ecosystem Services (CICES), inter-governmental agreement under the United Nations Statistics Division; The Economics of Ecosystems and Biodiversity (TEEB), a global initiative focussed on the economic benefits of biodiversity.
[2] http://www.nature.org/science-in-action/ecosystem-services.xml
[3] http://programs.wcs.org/carbon/Ecosystem-Services/Overview.aspx

I address considerations of natural resource management and biodiversity conservation in later sections.

The quantification and valuation of nature

Valuation is the process of expressing value (Farber *et al.* 2002) and while everything has a value not everything has a price. Criticisms that suggest ecosystem services reduce nature to dollar figures are ignoring the many ways that value can be quantified. Here I argue that by recognising the role of natural capital in ecosystem function and provision of a full range of services rather than just its value when extracted, exploited, or brought to market we can in fact broaden the way we value nature rather than narrow it.

Before we can assign value we first need to quantify or describe what it is that is to be valued. Robust, transparent, and repeatable quantification is also important as it allows for detection of change over time and performance monitoring (tracking against policy objectives). Quantification and consequent valuation is easier for some ecosystem services than others. For example, in terms of food supply, production can be easily quantified (e.g. kg milk solids/ha/year) and this quantification can then easily be converted into monetary valuation (e.g. $/kg milk solids). In some cases, quantification is not difficult but valuation can be. For example, the state and trend of water quality parameters can be robustly measured and quantified but the valuation of the condition, and importantly the change in condition, of water quality is much more difficult and contentious to determine and agree on. For less tangible ecosystem services such as cultural services, both quantification and valuation can be challenging. The latter two examples embody public goods for which no market exists and monetary valuation is unsatisfactory at best and inappropriate at worst. Applying ecosystem services thinking to decision making can provide a platform that allows values to be expressed more broadly than monetary valuation.

The valuation of ecosystem services is a key component of the concept, and is intended to describe a wide range of values including market and non-market, use and non-use, and tangible and intangible values across ecological, socio-cultural, and economic domains (MEA 2003; Kumar 2010). Thus, 'value' within the ecosystem services concept is multidimensional and not restricted solely to monetary terms. Indeed the primary use of valuation intended by the Millennium Ecological Assessment (MEA 2003) was as a tool to enable the evaluation of trade-offs between alternative scenarios and consequences of particular land use options. Despite this, there is concern within the literature (Spash 2013; Turnhout *et al.* 2013; Silvertown 2015) and the public debate (e.g. Monbiot 2014) that the valuation of ecosystem services only results in the commodification of nature, relying on markets to account for the value of nature. Given the well-known history of market failures to adequately value nature and public goods (Cork *et al.* 2001; Turner & Daily 2008; Liu *et al.* 2010; Helm & Hepburn 2012; Silvertown 2015) this concern is understandable. Economic methods (market prices, hedonic prices, travel costs, and replacement costs) can be used to indicate the value of change in the provision of ecosystem services but are unlikely to be able to value the actual ecosystem services themselves. Therefore, not only is economics limited in its ability to adequately value the contribution of nature to human wellbeing, it is unhelpful to decision making to do so (Daily *et al.* 2000; Heal 2000). Thus, relying solely on direct economic valuations of nature will indeed have a detrimental impact on its long-term persistence as that which cannot be captured in monetary terms will continue to be underprovided for by the market (Helm & Hepburn 2012).

Recognising the full range of services and values is challenging to implement simply because describing and accounting for non-market, non-use, or intangible values is difficult and economic and ecological values do not always align (Nunes & van den Bergh 2001; Farber *et al.* 2002). Use of a common currency (to compare like with like) for valuing ecosystem services and benefits when assessing trade-offs between values is critical for comparative analysis (Seppelt *et al.* 2012). In this respect, it is tempting to default to monetary valuation to guide decision making due to general familiarity with both valuation techniques (e.g. markets) and outputs (monetary values). Indeed, it is within the cost-benefit analysis (where the consequences of actions and alternative actions are assessed) within the decision making process that economic valuation is necessary and where the common currency is typically monetary (Daily *et al.* 2000; Heal 2000). However, this evaluation should be focused on the benefits the system generates (not the underpinning natural capital stocks) and will always be an underestimation of the total value provided to society (Heal 2000).

I suggest that the persistent and widespread concern over the pitfalls of 'valuing nature' has caused the potential improvements for decision making to be overlooked or underestimated and slowed a more general acceptance of an ecosystem services approach. However, advances in accounting for the benefits we derive from nature are being made. For example, environmental accounting based on indicators and benchmarks for each environmental asset is being progressed. Such an accounting system would introduce a non-monetary common currency by which accounting for the multiple measures of natural capital and ecosystem services could be readily incorporated into decision making (Wentworth Group of Concerned Scientists 2008; McDonald 2014). In recent years frameworks for natural capital accounting have been developed providing a methodology by which to identify “natural capital deficits” providing the potential to trigger necessary policy responses (Guerry *et al.* 2015). Other major attempts to incorporate the value of natural capital and ecosystem services into national economic accounting (e.g. the World Bank’s Wealth Accounting and Valuation of Ecosystem Services initiative), infrastructure investments (e.g. the InterAmerican Development Bank’s Biodiversity and Ecosystem Services Program), and impact assessments on loan applications (e.g. the International Finance Corporation) have been initiated (Guerry *et al.* 2015; Polasky *et al.* 2015).

However, economic valuation is just one component of the decision making process within an ecosystem services context and will always need to be integrated with ecological and wider social considerations that explain the relationship between humans and natural capital and our dependence on it.

Ethical challenges

The ethical quandaries surrounding the ecosystem services concept are a continuation of those surrounding many conservation and environmental issues. Environmental and conservation ethics have been long contested and likely always will be where we need to make difficult choices. For example, applying principles of triage and prioritising species for conservation (planning for extinctions vs. *ad hoc* extinctions) (Bottrill *et al.* 2008; Joseph *et al.* 2009), indigenous peoples’ use rights versus conservation of species, such as the New Zealand endemic kereru (*Hemiphaga novaeseelandiae*), a pigeon of significant cultural importance to Māori which is protected and therefore harvest is prohibited (Wright *et al.* 1995), and more recently biodiversity offsetting (Ives & Bekessy 2015; Maron et al. In press-a; Maron *et al.* 2016a). Ethical considerations form an important part of the public discourse but are fundamentally driven by a personal world-view and can be inflexible. The ecosystem services concept has however provided a mechanism by which both utilitarian and intrinsic values can sit side by side (Schröter *et al.* 2014). Importantly, the ecosystem services concept is not intended to replace other paradigms or arguments but to bring together a wide range of values, broadening the conversation and engaging with a wider range of stakeholders and communities (Braat & de Groot 2012; Schröter *et al.* 2014).

Although challenging, these issues need not be prohibitively obstructive and in this thesis I argue that an ecosystem services approach, while not without uncertainty and limitations, provides the opportunity for decision making that more effectively and explicitly incorporates a wider range of values, more clearly illustrates the consequences of actions, and can be used to make decision making more effective.

The challenges of inconsistent frameworks, terminology, and messaging

Commonly used definitions that distil concepts (e.g. 'the benefits of nature to humans') are effective at communicating a multifaceted concept to a wide audience but obscure the complexity inherent in the provision of ecosystem services (Fu *et al.* 2013). Attempts to provide a more informative definition and classification system for ecosystem services have been made by many authors (de Groot *et al.* 2002; Boyd & Banzhaf 2007; Wallace 2007; Fisher *et al.* 2009; Maynard *et al.* 2010; Haines-Young & Potschin 2011; Bastian *et al.* 2012) and these definitions and classifications continue to evolve as our understanding of ecosystem services increases.

As the interest in the ecosystem services concept continues to grow, the various definitions and classifications also continue to be inconsistently applied across published frameworks (Haines-Young & Potschin 2011). This inconsistency is in part reflective of the evolution of both the concept and its application, and in part the interdisciplinary nature of an ecosystem services approach which lends itself to a myriad of terms and applications. As many services and benefits are demanded and described by communities (beneficiaries) there is no 'one size fits all' application of the ecosystem services concept. Local level adoption invariably leads to adaption of broader definitions. Thus, a definitive list of ecosystem services is difficult as one classification system will not fit all applications (Fisher *et al.* 2009). In this sense, inconsistency in terminology and definitions is not entirely negative and can make the concept more accessible across disciplines where strict use of jargon might otherwise limit this (Haines-Young & Potschin 2011). I suggest that as the concept evolves and understanding deepens it is not only appropriate but necessary to move away from initial 'starting points' such as that provided in the Millennium Ecosystem Assessment (MEA 2005).

Rather than focus on a definitive list or classification system of ecosystem services, I consider it to be of more use to increase clarity regarding how ecosystem services are derived and how management actions can best influence their provision. If fundamental misunderstandings or confusion regarding the concept prevail, opportunities to truly effect change on the ground will be lost. For example, terminology and classifications that obscure the key components and relationships that contribute to the provision of ecosystem services are critical impediments to implementing ecosystem service policies. It is difficult to manage ecosystem services without understanding the distinctions and links between stocks, processes, and services (Fu *et al.* 2013). Without this clarification, an ecosystem services approach cannot be readily operationalised as it is not obvious where to target actions that would best influence positive change in the ecosystem service(s) of interest.

This thesis addresses this central issue by clearly distinguishing between natural capital stocks, ecological processes, and ecosystem services. Natural capital focussed approaches to ecosystem service provision and natural resource management decisions, including risks to service provision from natural capital degradation, are becoming more prevalent in the literature (e.g. Bristow *et al.* 2010; Dasgupta 2010; Dominati *et al.* 2010; Bateman *et al.* 2011; Daily *et al.* 2011; Mace *et al.* 2011; Robinson *et al.* 2013; Barbier 2014; Mace *et al.* 2015; Tälle *et al.* 2016) and in high profile collaborations and projects (e.g. the Natural Capital Project[4]) and governmental advisory groups (e.g. the United Kingdom's Natural Capital Committee[5]). However, substantial gaps in our understanding of how management interventions can contribute to improvements in ecosystem services provision remain. I clarify that focusing on how land management impacts on natural capital stocks provides a way to recognise critical connectivity between natural capital stocks and ecosystem services and assists in developing an approach that will see ecosystem services thinking operationalised.

Natural resource management and the ecosystem services concept

Natural resource management is the management of natural resources (e.g. soil, water, biodiversity — i.e. natural capital stocks) with an emphasis on how this management impacts on the capacity of resources to continue to provide for human wellbeing now and into the future. Thus, natural resource management encapsulates principles of sustainable development. The World Commission on Environment and Development report 'Our Common Future' ('the Brundtland Report') defined 'sustainable development' as meeting: *"the needs of the present without compromising the ability of future generations to meet their own needs"* (Brundtland *et al.* 1987). The vagueness inherent in this definition was born of political necessity for universal agreement (Daly 1990; Giddings *et al.* 2002) and left the door open for multiple interpretations to arise.

Daly (1990) suggests long-term sustainable development requires the use of resources at rates that maintain all forms of capital at the "*optimal level"*. For natural capital resources, operating at an

[4] www.naturalcapitalproject.org
[5] www.gov.uk/government/groups/natural-capital-committee

'optimal level' requires the acknowledgement of environmental limits and understanding safe operating thresholds (Rockström *et al.* 2009; Mace *et al.* 2015; Steffen *et al.* 2015). It also requires the environmental, social, and economic dimensions be given consideration in decision making. The representation of the three dimensions of natural resource management has traditionally been as separate, but interconnected rings of equal size (Figure 1.2a). However, this conceptual model also assumes separation as much as it does interconnectedness. This allows for substitution between the dimensions, the rings to be 're-sized' (one given priority over another), or trade-offs to occur in the absence of accounting for consequences (Giddings *et al.* 2002). A reorientation of the conceptual model of sustainable development so that the rings are nested (Figure 1.2b) more accurately reflects the dependency of the economy on society and the dependency of both the economy and society on the environment (Giddings *et al.* 2002).

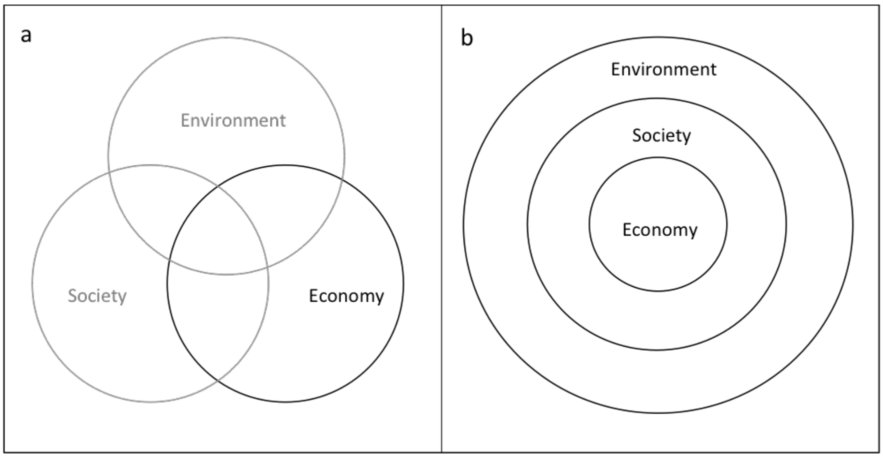


Figure 1.2: Conceptual sustainable development models. **a**. Common model where the three dimensions are compartmentalised and represented by three interconnected but separate rings. This model allows one dimension to be prioritised (e.g. the economy as shown in black) at the cost of the other dimensions (as shown in grey). **b.** A more accurate conceptual model whereby the nesting of the three dimensions represents their total integration and the dependence of the economy on society and the dependence of both the economy and society on the environment. Adapted from Giddings *et al.* (2002).

Despite the inherent protectionist element, natural resource management is not synonymous with preservationist conservation (such as the establishment of 'no-go' reserve networks), in that it recognises and provides for the use and development of resources, as well as the protection of these resources. However, natural resource management can include planning for conservation alongside, or as a critical component of, resource use. For example, regulating land use to prevent the further loss of resources (e.g. controlling vegetation clearance) or embedding restrictions based on environmental flows into water allocation frameworks. Importantly, there is a common call for natural resource management to be focussed on the sustainable management of resources in order to sustain life supporting capacity and provide for intergenerational equity.

However, current realities do not necessarily reflect ideal definitions of natural resource management, and typically view the environment, society, and the economy as separate dimensions allowing greater priority to be given to one over the others (Figure 1.2a). Although there will always be trade-offs across the three dimensions, short-term economic growth (which largely externalises social and environmental costs) is typically favoured in decision making challenging, and in some cases undermining, sustainable management principles (Turner & Daily 2008; Helm & Hepburn 2012). History has shown the consequences of depleting rather than sustaining natural capital to be dire, severely reducing human wellbeing and resulting in societal collapse (Diamond 2005; Montgomery 2012). Despite these lessons, protecting the life-supporting capacity of ecosystems, or recognising the values flowing from natural capital remains poorly integrated in decision making (Wallace 2007; Hails & Ormerod 2013; Robinson *et al.* 2013), as is reflected in the continued global trend of biodiversity loss (Butchart *et al.* 2010; Cardinale *et al.* 2012), and the pushing of systems to the edge and beyond of their inherent capacity (Rockström *et al.* 2009; Mace *et al.* 2015). Natural resource management based on economically weighted decision making is in effect replacing the underlying principles of sustainable 'development' with 'growth' (Daly 1990). However, as the human economy sits *within* the global ecosystem not outside of it (Figure 1.2b), and the global ecosystem and natural resources are not boundless, there are limits to 'growth' (Daly 1990; Rockström *et al.* 2009). Economists would also agree that the costs of any outcome pursued "*at any cost*" is too high (Pannell 2004). Thus, the benefits of unfettered economic development are costly and short-sighted and another model is required.

Attempts to mitigate or reverse environmental degradation caused by non-sustainable land use practices incurs large economic costs and puts pressure on public funding (Holl & Howarth 2000). Further, biodiversity restoration and environmental clean up programmes can rarely occur at rates or spatial scales adequate to mitigate the ecological infrastructure that has been lost. Such programmes are easily undermined, or further compromised, by continued degradation of the very natural capital that recovery projects aimed to reinstate. The following examples of agricultural intensification in New Zealand and vegetation management in Queensland, Australia eloquently illustrate both the huge public costs associated with addressing degradation to natural capital, and the futility of these programmes when large-scale loss of natural capital continues.

The '20 Million Trees Programme' (Australian Government 2016) is an Australian Government funded initiative that aims to re-establish green corridors and urban forests lost by previous land clearing by planting 20 million trees and associated understorey vegetation by 2020. The Tree

Programme has a budget of AUS$50 million to achieve this goal. However, policy changes enacted by the Queensland Government in 2012 have resulted in land clearance rates increasing by 73% from 153 638 ha/yr in 2011–2012 to 266 191 ha/yr in 2012–13, with a further 11% increase to 296 324 h/yr in 2013–2014 (Queensland Department of Science Information Technology and Innovation 2015). The 2013–2014 clearance rate is the highest rate recorded since the end of broad scale clearing permits in 2006 (Queensland Department of Science Information Technology and Innovation 2015) and this loss alone far exceeds what will be replaced by the entire 20 Million Trees Programme (Maron *et al.* 2016b). Changes to the vegetation clearance policies were made in the expectation that increased clearance would contribute to regional economic growth. However, the long-term consequences of continued high rates of vegetation clearance in Queensland are far-reaching and costly, jeopardise Australia's ability to meet conservation targets for threatened species, and undermine attempts to reduce carbon emissions and nitrification and sedimentation of the Great Barrier Reef (Maron *et al.* 2016b).

The New Zealand Government's Business Growth Agenda (Ministry of Business Innovation & Employment 2012) includes the goal to double the value of agricultural sector exports by 2025. This ambitious goal includes the expansion of dairy in order to increase the volume of milk solids produced. To support this intensification the New Zealand Government launched the Irrigation Acceleration Fund in 2011 (initially NZ$35 million over five years, with an additional NZ$25 million 'kick-start' in operating funding in 2015) to support the development of rural water infrastructure (Ministry for Primary Industries 2016). At the same time New Zealand is facing considerable land use induced water quality issues, particularly in agricultural landscapes (Scarsbrook 2006; Ballantine & Davies-Colley 2014; Parlimentary Commissioner for the Environment 2015). In response, the Government-funded Fresh Start for Fresh Water Clean Up Fund allocated NZ$14.5 million from 2011–2014 to assist regional councils to address water quality issues within seven priority waterbodies (Ministry for the Environment 2016). The National Government has made a total 'Clean Up' commitment of NZ$264.8 million (New Zealand Government 2011). In addition, Fonterra (New Zealand's largest dairy cooperative) and the Department of Conservation (DOC) have entered into an agreement to share coordination of a NZ$20 million community investment over ten years to protect and enhance five water bodies as part of the 'Living Water' initiative (Fonterra 2016). Despite this expenditure these programmes cannot address the full scale of the water quality issues, particularly in light of continued intensification of agriculture.

Both the Australian and New Zealand examples highlight not just the environmental and social costs of short-term economic growth objectives, but also the substantial and on-going economic costs to community. The long-term consequences and economic costs of not addressing and investing in ecological infrastructure in order to maintain the capacity of natural capital are compounding and intergenerational.

Public agricultural policies channel billions of dollars towards growers globally, although few extend beyond encouraging and enhancing the provision of commodities (Plieninger *et al.* 2012). Agri-environment schemes (such as those within the European Union’s Common Agricultural Policy) have shifted the focus of incentive schemes towards environmental outcomes and multifunctional landscapes. However, Wilson (2008) argues that policy interpretations of multifunctionality are driven by a strong desire to maintain the farm subsidy culture. Incorporating the full range of ecosystem services and values into natural resource management decision making could more appropriately reorientate agri-environment schemes to better invest in and protect ecological infrastructure and encourage multifunctional agriculture.

Reorientating the implementation of sustainable development principles will require a considerable change in the world-view of individuals, organisations, and governments. To date, this shift has been slow in emerging. In the face of the global biodiversity crisis (Cardinale *et al.* 2012), increasing inequality and poverty (Hardoon *et al.* 2016), and concerns over food security (Godfray *et al.* 2010) the urgency for a paradigm shift could not be much greater. An ecosystem services approach that explicitly identifies how ecosystems provide the natural capital that is central to our wellbeing and survival has the potential to engage participants by bringing a broader perspective to resource management and as such has real merit as a catalyst for change.

Considering the above, I suggest that application of ecosystem services thinking to natural resource management decision making provides an opportunity to account for and thus internalise many of the environmental, social, and intergenerational consequences of development that were previously externalised. Trade-offs will still occur, but an ecosystem services approach provides the opportunity for the consequences of these trade-offs to be more explicitly defined and alternative actions compared. This knowledge can inform natural resource management objectives and priorities targeted at sustaining natural capital and counter unmitigated drawdown of natural capital and associated capacity.

Biodiversity conservation and the ecosystem services concept

What is biodiversity?

'Biodiversity' is a contraction of 'biological diversity', which describes the variability that exists amongst the different levels (ecosystem, species, genes) of ecological organisation. The term is also commonly used as an all-encompassing surrogate term for both ecosystems and species *per se* (rather than the descriptor of variance). This latter usage of the term has become entrenched in both academic and common parlance. In the context of conservation, the term 'biodiversity' becomes even more reduced, typically used to refer to a subset of species, habitats, ecosystems, or areas of conservation concern (Mace *et al.* 2012).

Addressing native biodiversity decline

The continued global decline in biodiversity (Cardinale *et al.* 2012) suggests that, on balance, current conservation initiatives are failing and protection alone is not enough to halt biodiversity declines (Clout 2001; Watson *et al.* 2014). However, views are divided as to whether the ecosystem services concept in effect enhances or detracts from biodiversity conservation. A reoccurring criticism of the concept is that it conflicts with biodiversity conservation objectives (Schneiders *et al.* 2012; Schröter *et al.* 2014), although others claim it is possible to manage for both biodiversity conservation and ecosystem services within managed ecosystems, although trade-offs are inevitable (e.g. Chan *et al.* 2006; Goldman *et al.* 2008; Mason *et al.* 2012; Rega & Spaziante 2013; Cordingley *et al.* 2016).

The introduction of any new jargon or the shifting of focus can further dilute conservation messages and confuse stakeholders and decision makers. Biodiversity conservation is highly complex and subject to socio-political influences and economic constraints and there are extremely few 'one size fits all' management options. As we learn and understand more, biodiversity conservation will get more complex not less, management actions will change and priorities shift. These factors combine to make biodiversity conservation an often misunderstood practice, and communication about biodiversity can become confused. The disinterest or disengagement of large portions of society can in a large part be attributed to the myriad of complexities associated with biodiversity and conservation practices (Novacek 2008). While the introduction of the ecosystem services concept has done little to reduce this well entrenched confusion it equally has not been the sole cause of it. There is some merit in the argument that the ecosystem services concept in fact improves community understanding of biodiversity by increasing the day-to-day relevance of nature to individuals and their wellbeing. We don't value what we don't understand, and by illuminating the direct and indirect links between nature and humans the ecosystem services concept can potentially harness wider support for conservation initiatives.

Conservation actions directly contribute to the sustained provision of cultural ecosystem services, such as the maintenance of culturally and spiritually important species, a sense of place, and intergenerational connections to place. It would however, be misguided to consider ecosystem services as the silver bullet to resolve biodiversity conservation resourcing or valuation issues. It is important to remember that many ecosystem services can be as readily (if not equally[6]) provided by exotic as native biodiversity, and thus an ecosystem services approach to resource management will not necessarily always favour native biodiversity. This is more likely where decision making processes do not include consideration of cultural services and is illustrative of potential negative consequences of not including the full range of ecosystem services in assessments. This could be an issue where maintaining or encouraging native dominance in a landscape is important. Clearly stated biodiversity conservation goals and targeted actions to achieve them will continue to be necessary. This is of particular importance in places where a preference for particular species over others exists, there is a need to actively manage and control invasive species, or there are immediate and urgent threatened species challenges that require a specific focus to address.

Native versus exotic biodiversity

In many parts of the Northern Hemisphere whether biodiversity comprises native or exotic (introduced) species is hard to conclusively define or is of less relevance ecologically or socially, with the exception perhaps of species included on the International Union for Conservation of Nature (IUCN) 'Red List' of threatened species (Hilton-Taylor 2000). In contrast, in areas of the world where biophysical boundaries are clearly defined, species arrivals (including human) are well documented, and a high level of endemism of species and ecosystems exists the distinction between native and exotic is of great importance. New Zealand and Australia both provide examples of where a distinction between native and exotic species is very obvious and native species have greater value over exotic species from conservation and indigenous cultural perspectives. In both

[6] For example, both exotic and native species will sequester carbon, or contribute to the retention of soil but different species will not necessarily perform these functions to the same degree. Monoculture stands will also perform differently than mixed-species stands (Reubens *et al.* 2007; Gamfeldt *et al.* 2013).

countries invasive species[7] are major drivers compromising the persistence of native species and ecosystems, and where land conversion no longer occurs, are the primary driver of species extinction (Craig *et al.* 2000; Clout 2001; Australian Bureau of Statistics (ABS) 2010). Thus, considerable conservation effort and expenditure have been invested in the management of invasive species, entrenching concepts of 'good' and 'bad' species. The value-loaded perception of native versus exotic species has been further complicated by the rapid and severe replacement of many native ecosystems with (often monoculture) exotic ecosystems such as pasture and plantation forestry upon which national economies are reliant. In an economic context it is exotic ecosystems, not native, which are favoured. As a consequence, it is not uncommon for native and exotic biodiversity to be addressed in separate resource management policies and programmes. However, contemporary ecosystems include modified versions of previous assemblages that can include both exotic and native species and entirely novel ecosystems (new, non-historical configurations, Hobbs *et al.* 2009). Novel ecosystems can provide ecosystem functions and services and conservation values that restoration and management interventions need to recognise (Hobbs *et al.* 2014). However, care is needed so as ecosystem services objectives do not undermine biodiversity conservation objectives. I suggest that by placing biodiversity (native and exotic) within the ecosystem services approach as natural capital stocks, and given the appropriate weighting, the two branches of biodiversity management can be brought together in a decision making process without losing sight of the need to address the detrimental impact of invasive species on biodiversity and natural capital stocks.

The relegation of native biodiversity entirely to conservation programmes outside of an ecosystem services framework will continue to externalise costs to native biodiversity and underestimate contributions made by native biodiversity to service provision, and thus miss the opportunity for additional protection (Figure 1.1). Despite the potential risks of exchange of native for exotic biodiversity, both should be captured by an ecosystem services approach. To do otherwise, equally risks continued native biodiversity losses.

Thesis structure

Agricultural landscapes cover 38% of land globally, making agriculture the dominant form of land use (Dale & Polasky 2007). Agricultural practices have, particularly since the Green Revolution, been a major driver of detrimental environmental change, compromising or preventing the

[7] Introduced species that drive detrimental impacts on existing populations or transform ecological patterns or processes (Simberloff *et al.* 2013).

provision of other ecosystem services (Swinton *et al.* 2007; Plieninger *et al.* 2012; Balbi *et al.* 2015; Tanentzap *et al.* 2015). Agricultural landscapes also contain a wealth of natural capital and provide many opportunities for adaptive measures to global change such as carbon sequestration (Smith & Sullivan 2014). Thus, how we manage these landscapes globally is critical. The research described in this thesis predominantly concerns agricultural landscapes as it is here that the opportunities and urgency to effect real change exist. Nevertheless, underpinning concepts presented here are equally applicable within urban landscapes or areas under conservation land use.

In this thesis I focus strongly on the ecological components of ecosystem service provision, but place this in the context of decision making at both the governance and individual level. Using an ecological understanding of system function, this research explores the ecological and social consequences of land management choices (Figure 1.3).

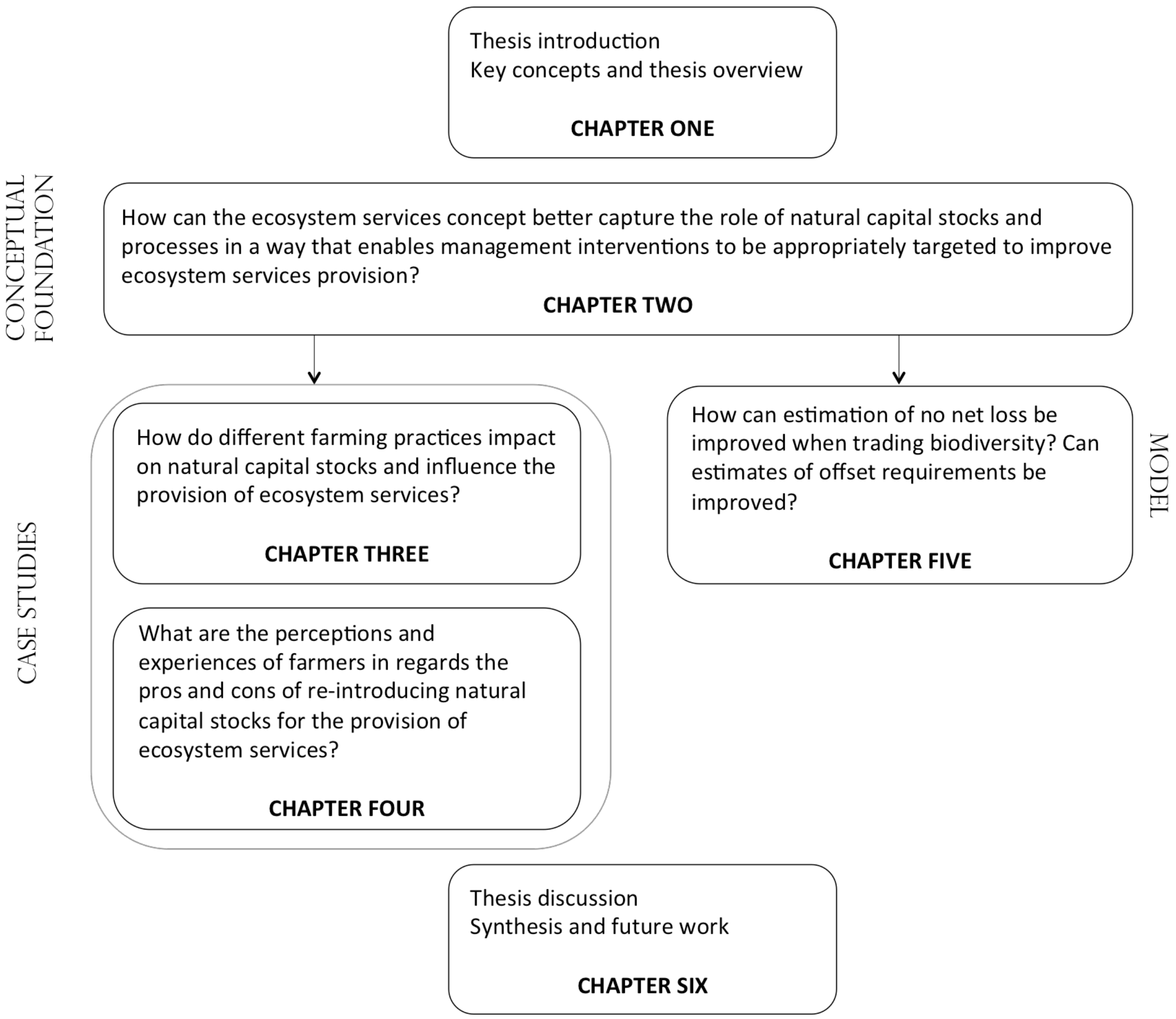


Figure 1.3: Thesis structure and research questions. Arrows indicate linkages between chapters.

Beyond this introduction, this thesis consists of a core of four chapters written in the form of papers, all of which have been submitted to peer reviewed journals. They are reformatted here for consistency and references have been placed together in a section at the end of the thesis. While these chapters reflect stand-alone pieces of work, they combine to address the central aim of narrowing the gaps between science, policy, and practice by developing and testing natural capital-ecosystem services concepts and implementation pathways to support natural resource management and conservation decision making for the purposes of sustaining multifunctional landscapes.

In **Chapter 2** I present a conceptual framework which orientates the ecosystem services concept towards management of natural capital stocks. The framework conceptualises 1) how the stocks and processes that together comprise natural capital contribute to ecosystem services and ultimately to human wellbeing and 2) that management actions targeted at specific attributes of natural capital stocks have greater effectiveness in influencing the provision of ecosystem services. In this chapter I show that attributes of natural capital stocks are variable, and that by differentiating those attributes that are more easily manipulable within policy relevant timeframes from those attributes that are less readily manipulated provides the opportunity to achieve policy objectives. I posit that a structured decision making process can determine the appropriate management of natural capital stocks for achieving specific policy objectives, and provide a process by which to shift an ecosystem services approach from conceptual to operational.

Land management practices directly impact on the condition of natural capital stocks. In **Chapter 3** I demonstrate the impact of different land management regimes on vegetation natural capital stocks as reflected in species diversity and abundance measures. Species richness and abundance drive ecosystem service provision, either through effects of local diversity or through the presence of ecosystem service providing species at particular abundances. However, management practices will differentially affect species providing different ecosystem services. I test the influence of management practice within ‘provider groups’ — sets of plant species which exhibit attributes from which ecosystem services and benefits arise. This chapter relies on empirical data from traditional farmed grasslands in Transylvania (Romania) to demonstrate the concepts presented in Chapter 1.

Policy incentives and environmental enhancement initiatives are typically less successful at bringing about effective behaviour change when the motivations of farmers and social and economic drivers of behaviours are not well understood (Pannell *et al.* 2006). Riparian margin management is a common policy option for mitigating the externalities of land use, and incentive schemes to achieve these policies are equally numerous. **In Chapter 4** I describe the outcomes of workshops with dairy farmers farming the Taranaki ring plain, New Zealand regarding their experiences and perspectives of the pros and cons of planting riparian margins on their farms and place the farmers' perspectives in the context of wider policy and land use pressures. Incorporation of social data, particularly that describing human behaviour and decision making, is critical to embed the ecosystem services concept into natural resource management policy and practice (Asah *et al.* 2014).

Biodiversity offsetting is a mechanism that explicitly combines economic development and conservation objectives by compensating for the residual impacts of development activities on biodiversity with improvements in biodiversity values elsewhere. Policy-driven biodiversity offsetting has a strong focus on conservation priorities (e.g. threatened species or habitats of conservation concern), but can also be applied to a broader range of biodiversity elements including non-threatened species and habitat. Although biodiversity offsetting proposals are typically (but not always) framed in the narrower context of conservation and associated cultural ecosystem services (e.g. priority species and habitats) rather than a wider context of maintaining ecosystem function, trading biodiversity that contributes to ecosystem service provision implicitly trades all the ecosystem services derived from that biodiversity. Therefore, biodiversity offsetting represents a critical and high-stakes application of the ecosystem services concept. A shift towards the more explicit consideration of ecosystem services provision within offset proposals is also occurring (e.g under the United States Clean Water Act, Ruhl *et al.* 2009). Natural capital stocks (species and habitats) are manipulated by either the removal of species and habitats at the development site to provide for provisioning services (e.g. extraction of raw materials or food production) or in their replacement or enhancement at the offset site to deliver cultural services (e.g. maintaining species of conservation concern). Public support for these trade-offs is sought via either the promise of economic development in exchange for conservation, or the promise of environmental gain for the cost of economic development (Tallis *et al.* 2008). Evaluating the biodiversity equivalence of a measurable current state that will be lost to development and an uncertain future state that will be gained by an offset action requires a currency that quantifies both losses and gains and an accounting model to evaluate the exchange objectively. **Chapter 5** outlines a disaggregated accounting model that addresses the limitations of current accounting models in assessing biodiversity offset proposals.

I conclude the thesis in **Chapter 6** with a discussion that ties the key results and conclusions of the preceding chapters together, and identify areas for future work.

CHAPTER TWO:

MANAGING NATURAL CAPITAL STOCKS FOR THE PROVISION OF ECOSYSTEM SERVICES

This chapter provides the conceptual foundation for the thesis and advances existing commonly referenced frameworks by formalising the relationship between natural capital stocks and ecosystem services and thus recognising the manipulation of natural capital stocks provides an effective opportunity by which to influence the provision of ecosystem services. Using theoretical and real examples targeting both vegetation and soil natural capital stocks, I illustrate how the concept can be applied to inform decision making and influence on-the-ground change. This conceptual work recognises critical interactions between aboveground and belowground stocks providing a more realistic picture of ecosystem function.

A version of this chapter has been published in Conservation Letters as:

Maseyk FJF, Mackay AD, Possingham HP, Dominati EJ, Buckley YM 2016. Managing natural capital stocks for the provision of ecosystem services. Conservation Letters doi: 10.1111/conl.12242.

Abstract

Decision makers and land managers are increasingly required to manage landscapes for multiple purposes and benefits. The targeting of interventions towards natural capital stocks that contribute to the supply of ecosystem services can influence the provision of these services. However, despite progress in the development of frameworks linking natural capital to the provision of ecosystem services and human benefits there remain gaps in our understanding as to how management interventions can improve ecosystem service provision. We provide a framework that explicitly links natural capital stocks to ecosystem service provision and identify manageable attributes of natural capital stocks as the critical intervention point. A structured decision making process based on our framing of the ecosystem services concept can facilitate its application on the ground.

Why linking natural capital stocks to ecosystem service provision is crucial

As the global population and consumption increase, there is an increasing emphasis on the production of food and other resources necessary for life. Consequently, many ecosystems are being pushed beyond their inherent capacity (MEA 2005; Rockström *et al.* 2009; Steffen *et al.* 2015) as the ability to produce goods is favoured over other critical functions and services that ecosystems provide such as energy transfer, water regulation, nutrient filtering, and carbon sequestration. The ecosystem services approach (see chapter glossary), and in particular an emphasis on the multifunctionality of ecosystems, allows for decision makers to consider multiple benefits that flow from ecosystems. Although ecosystem services frameworks continue to evolve, several commonly replicated and referenced frameworks (MEA 2005; Kumar 2010) lack explicit linkages between ecosystem services and benefits to humans and the natural capital stocks that underpin them. While the importance of natural capital is increasingly being recognised (Bristow *et al.* 2010; Dasgupta 2010; Dominati *et al.* 2010; Bateman *et al.* 2011; Daily *et al.* 2011; Mace *et al.* 2011; Robinson *et al.* 2013; Barbier 2014; Mace *et al.* 2015) and a natural capital framework has been used to assess risks to ecosystem service provision from natural capital degradation (Mace *et al.* 2015) guidance on explicit intervention points for management is still lacking.

Attempts to implement an ecosystem services approach without explicit reference to natural capital stocks are unlikely to be successful. The natural capital and ecosystem services concepts are not identical, and in order to provide ecologically and socially informed management options we need clarity in how natural capital contributes to the provision of ecosystem services and a structured decision making process to identify and evaluate intervention points. While many uncertainties remain within the biophysical realm of ecosystem services provision, orienting the concept of ecosystem service provision to explicitly consider entry points for management will highlight those uncertainties that are critical for effecting change. The flow of ecosystem services to humans also contains uncertainties. Alternative policy interventions to those used to influence the biophysical components of an ecosystem services approach are required to influence the flow between service generation and the demand for those services. Despite the uncertainties surrounding the flow between ecosystem services and humans, we cannot wait until we understand the full complexity of ecosystem service provision before taking action to implement an ecosystem services approach.

We clarify and further formalise the relationship between natural capital and ecosystem services and use this framework to explore how management of natural capital stocks can alter ecosystem service provision. We show that interventions need to be targeted at specific manageable attributes of natural capital stocks to change service and benefit flow. We explicitly link decision making for ecosystem services with interventions focussed on manageable attributes of natural capital stocks and suggest that a structured decision making process provides a logical pathway to implement this approach.

The relationship between natural capital and ecosystem service provision

Capital is defined as "*a stock that yields a flow of valuable goods or services into the future*" (Costanza & Daly 1992). Formalising the functional relationship between natural capital, ecosystem processes, and ecosystem services introduces a mechanistic perspective to ecosystem service delivery and management that provides useful insights for decision making.

The quality and quantity of natural capital stocks is influenced by many factors including environmental and social drivers and pressures (Table 2.1) and other forms of capital. Thus the concept of natural capital should not be considered to be absolute or independent of human influence but rather as a part of the total capital that gives rise to ecosystem services (Arias-Maldonado 2013). Here we explicitly consider where interventions on the non-human biophysical components might have most influence on the provision of ecosystem services.

Table 2.1 (next page): An illustration of manageable and unmanageable attributes using key attributes of soil and vegetation natural capital stocks as examples. Characteristic properties of both soils and vegetation are those attributes that would be expected to be represented in particular combinations in undisturbed temporal and geographical space and attributes that are intrinsically less amenable to manipulation, and thus classified as *unmanageable attributes*. Soil stocks possess a number of inherent properties that generally would require considerable effort and expense to modify. These inherent properties determine the foundational value of soil stocks upon which other stocks (natural, built, or social capital) can be expended to increase usability and productivity and consequently value. Soil stocks also possess a number of attributes, which are more amenable to manipulation and can be managed at the farm or catchment scale. These we call *manageable properties* of soil (follows Dominati *et al.* 2010). Morphological, physiological, and functional traits of vegetative stocks are presented as attributes (as manifested within a particular species) that can be readily manipulated. Thus, we classify both the manageable properties of soil stocks and morphological, physiological, and functional traits of vegetation stocks as *manageable attributes*. This malleability makes certain stock attributes more responsive to management actions. The quality, quantity, and spatial configuration of natural capital stocks are influenced by environmental and social pressures and drivers and sustained over time by ecological processes, traits and functions. These same factors combine with natural capital and other forms of capital to generate ecosystem services.

Natural Capital Stocks			Pressures and Drivers	
	Key attributes		**Environmental**	**Social**
Soils	***Characteristic properties*** Slope Orientation Depth Clay types Texture Size of aggregates (subsoil) Stoniness Strength (subsoil) Subsoil pans Subsoil wetness class	***Manageable properties*** Soluble phosphate Mineral nitrogen Soil organic matter Total carbon Temperature pH Macroporosity Bulk density Strength (topsoil) Size of aggregates (topsoil)	***Drivers*** Climate Landform Soils (for vegetation) Vegetation (for soils) Geology and geomorphology ***Pressures*** Natural hazards (volcanic eruptions, earthquakes, storm events) Invasive species	***Drivers/pressures*** (as influenced by the market and consumer behaviour) Land use Farming practices Technology Public policy Individual decision making Anthropogenic climate change
Vegetation	***Characteristic properties*** Habitat type Vegetation community Species	***Morphological, physiological and functional traits*** Growth form Root architecture Root length Root depth Root tensile strength Canopy architecture Decomposition rate Nutrient uptake rate Carbon sequestration rate Biomass Height Nitrogen fixing ability	***Degradation processes and consequences caused by pressures on both soils and vegetation (interlinked):*** Erosion Sealing Compaction Salinisation Toxification Loss of organic matter Habitat loss Habitat fragmentation Decline in above and below ground biodiversity	

Processes, traits, and functions that sustain natural capital over time (as occur over short and long time-spans)	
Nutrient cycling Water cycling Energy cycling/transfer	supporting processes that flow between above and below ground stocks
Biological activity	above and below ground species interactions and performing of functions
Reproductive strategies Dispersal mechanisms Defence mechanisms Lifespan	species traits that sustain above and below ground biodiversity

The relationship between natural capital stocks and the flow of ecosystem services is supported by ecological theory linking biodiversity (species and ecosystems are stocks of natural capital, see glossary) to ecosystem processes and functions, which are critical to the provision of ecosystem services. The complex relationship between biodiversity and ecosystem processes and services has been well described (e.g. Cardinale *et al.* 2012; Mace *et al.* 2012; Maskell *et al.* 2013; Byrnes *et al.* 2014; Duncan *et al.* 2015). Here we highlight three key concepts to illustrate the use of ecological theory in structuring the problem: the species-area relationship, landscape ecology, and biodiversity-ecosystem function relationships. The species-area relationship (MacArthur & Wilson 1967; Rosenzweig 1995) which links loss of habitat with loss of species has functional consequences as species loss disrupts ecosystem processes (Hector & Bagchi 2007). The amount and configuration of habitat within a landscape influence species diversity (e.g. Fahrig 2013) and biodiversity-ecosystem function theory posits that biodiversity is a critical component of ecosystem function, where productivity is typically used as the focal ecosystem function (e.g. Hautier *et al.* 2014), but which can be extended to multiple functions (Maestre *et al.* 2012). Thus, biodiversity has a central role in the regulation of ecosystem processes (Balvanera *et al.* 2006; Cardinale *et al.* 2012) and land use that alters the composition, abundance or function of biodiversity has been shown to ultimately alter the structure of ecosystems (Newbold *et al.* 2015) and consequently their capacity to sustain functions and services (Nagendra *et al.* 2013).

While current understanding of the direct links between biodiversity and ecosystem service provision is insufficient to completely quantify consequences of losses and gains of biodiversity (Hails & Ormerod 2013; Harrison *et al.* 2014), changes in spatial patterns of and declines in biodiversity do restrict the provision of some ecosystem services in favour of others and this knowledge can be used to progress natural capital management (Cardinale *et al.* 2012). Here, we present a conceptual framework (Figure 2.1) to illustrate that by focussing on natural capital and formalising how it contributes to ecosystem service provision we can identify intervention points for decision making.

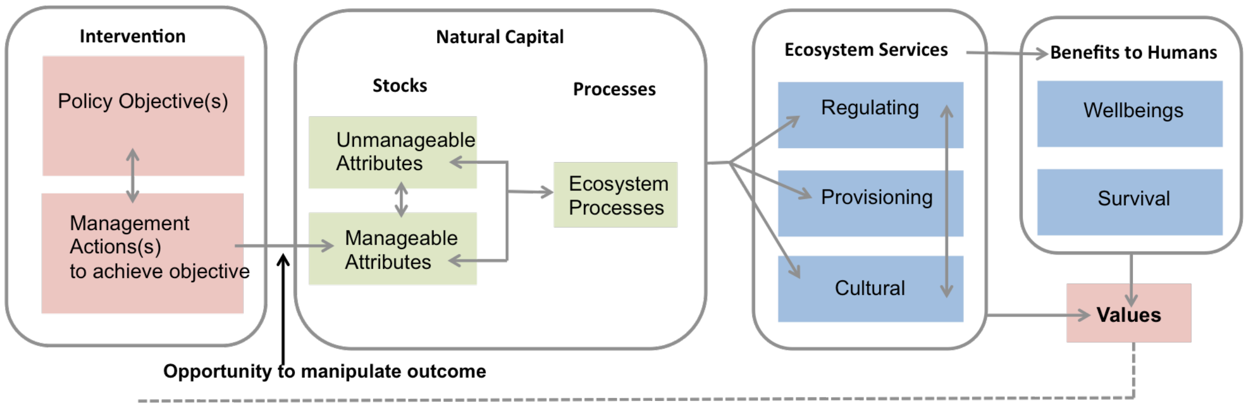


Figure 2.1: The conceptual relationship between interventions, natural capital stocks, ecosystem services and benefits to humans. The attributes of natural capital stocks are not uniform and can be differentiated on the basis of amenability to change, with some attributes less amenable to change on short timescales ('unmanageable attributes'), while others are highly amenable to change ('manageable attributes'). It is the manageable set of attributes to which actions can be specifically targeted. Only internal linkages between framework components are presented in Figure 2.1, although external drivers (e.g. environmental and socio-political) will also be exerting influence at all stages. Other forms of capital (e.g. social, human, and built) also act upon natural capital to bring about ecosystem service provision. These other forms of capital are implicitly captured within management actions and cannot operate in the absence of natural capital. The value assigned to the benefits from ecosystem services received by communities will exert influence through feedback to the size and scale of intervention (dashed line).

A conceptual framework for management of natural capital to enhance the provision of ecosystem services

Interventions

Interventions (policy and management actions) can be targeted on specific natural capital stock attributes for the purpose of influencing ecosystem processes, natural capital building, and ultimately the flow of desired ecosystem services. However, before we choose between actions we need to define broad goals and specific objectives. For example, we might be aiming to increase soil retention to maintain food production potential and reduce sedimentation of waterways to achieve water quality goals. The planting of trees might be an appropriate action for both these objectives, but how many trees, of what species, over how extensive an area, and where on the landscape to plant in order to obtain the required return on investment are critical management decisions. By focussing on changes to natural capital stocks, consequences for ecosystem services can be inferred and effectiveness of policies assessed. This provides the basis for adaptive decision making.

Natural capital

While the concept of natural capital has been recognised by economists for over twenty years as a useful model by which to consider sustainability (e.g. Costanza & Daly 1992; Arias-Maldonado 2013), the concept has more recently spread across other disciplines to where consideration of its importance is rapidly gaining traction. Natural capital comprises all abiotic and biotic elements of ecosystems (as well as ecosystems themselves) including the natural resources (e.g. water, soil, vegetation, species, air, which we refer to as 'stocks') and all physical, biological, and chemical processes (Mace *et al.* 2015). The concept of natural capital in a broad sense also captures stocks of natural elements that have been influenced or modified by social or political factors (Arias-Maldonado 2013). Thus, natural capital is broader than biodiversity and not synonymous with ecosystem services. We differentiate between stocks of natural resources and ecosystem processes here as stocks can be directly responsive to interventions while processes can be difficult to directly manage independently of stocks.

Stocks

Stocks can be measured in terms of their quantity, quality, and spatial and temporal dynamics across the landscape. The attributes of natural capital stocks are not uniform and can be differentiated by amenability to change, with some attributes less amenable, or more costly, to change on short timescales ('unmanageable attributes'), while others are highly amenable to change ('manageable attributes') (Table 2.1). Unmanageable attributes are inherent properties that determine the foundational value of stocks. Foundational stocks might interact with and add value to other natural, built, or social capital. Manageable attributes of natural capital that contribute to the provision of ecosystem services can be specifically targeted with actions to effect change in the provision of those services. Recognising that natural capital stocks are more or less amenable to management provides the basis for assessing the influence that actions have in altering the flow of ecosystem services.

Processes

Ecosystem processes are the pathways that link natural capital stocks and ecosystem services; they enable ecosystem functions and depend on, and build, natural capital stocks. Ecosystem processes include all ecological connections, networks, biogeochemical cycles and feedback loops, such as nutrient cycling, pollination, and energy transfer between trophic levels. Ecosystem processes are many, complex, and not fully understood. They are difficult to measure and monitor and, unlike stocks of natural capital, cannot be directly manipulated.

Ecosystem services and benefits to humans

Ecosystem services flow from natural capital stocks and processes and translate to benefits that give rise to the things that societies value. We recognise three groupings of ecosystem service: regulating (e.g. mitigated storm impacts, water quality, stable climate); provisioning (e.g. sustained food supply, raw materials, structural support); and cultural (e.g. presence of significant species, habitats, and landscapes). ‘Supporting’ (MEA 2005; Kumar 2010) or ‘intermediate’ (Mace *et al.* 2011) services (e.g. nutrient cycling, erosion control, energy transfer, and pollination) are ecosystem processes and not ecosystem services. Our distinction between ecosystem processes and ecosystem services aligns with several other studies (de Groot *et al.* 2002; Boyd & Banzhaf 2007; Wallace 2007; Dominati *et al.* 2010; Haines-Young & Potschin 2011; Braat & de Groot 2012).

The natural capital management framework for ecosystem service provision

Here we describe how ecosystem services are provided as a function of manageable and unmanageable attributes of natural capital stocks.

$$E_{ij} = f(\boldsymbol{x}_i, \boldsymbol{z}_i), \tag{1}$$

where E_{ij} refers to an ecosystem service, i is an index tracking a spatial management unit (i.e. paddock or farm property) (i = 1…n), j is the index that tracks ecosystem services (j = 1…m), $\boldsymbol{x}_i$ represents natural capital stock attributes that can be manipulated (‘manageable’ attributes) in a management unit i (x_{1i}, x_{2i}, x_{3i}...) and $\boldsymbol{z}_i$ are less easily managed natural capital stock attributes in a management unit i (z_{1i}, z_{2i}, z_{3i}…) (‘unmanageable’ attributes). The vectors $\boldsymbol{x}$ and $\boldsymbol{z}$ are stocks of each attribute (measured by the metrics quality, quantity, spatial configuration). The function f turns the stock attributes at a location (both $\boldsymbol{x}_i$ and $\boldsymbol{z}_i$) into an ecosystem service quantity and f consequently represents ecosystem processes.

This formalisation explicitly describes how land use and management actions can change the manageable stock attributes $\boldsymbol{x}_i$, and how the functional relationship between natural capital stocks and ecosystem processes acts upon both manageable $\boldsymbol{x}_i$ and unmanageable $\boldsymbol{z}_i$ stocks to produce ecosystem services.

To illustrate, we use an example of revegetation of the Loess Plateau of China. Large reforestation projects on the Loess Plateau have resulted in an increase in vegetation coverage from 6.5% in the 1970s to 51.13% in 2010 (Fu *et al.* 2013). The planting of erodible slope lands ($\boldsymbol{z}_i$) with tree, shrub, and grass species with desired root tensile strength attributes ($\boldsymbol{x}_i$) shifted what was a annual arable land cover to a mixed perennial vegetation cover and reduced sediment and water runoff via above-below ground interactions between $\boldsymbol{x}_i$ and $\boldsymbol{z}_i$ leading to increased sediment and water retention (f). These processes result in reduced accelerated erosion, slippage, and overland water flow, and increased intactness of the soil profile, thus increasing the provision of flood mitigation and water quality ecosystem services (E_{ij}) in the Yellow River.

This is a simplified illustration of our conceptual framework; ecosystem processes are interrelated and the relationship between stocks, processes and services is generally neither one-to-one nor linear. While equation (1) adequately captures our framework in the broadest sense it hides much of the complexity within the function f. Equation (1) also does not quantify the flow of ecosystem service nor account for different types of value of ecosystem services. This level of detail is highly contextual and gains relevance when linked to specific locations and well-defined outcomes. However, by use of an example we can disentangle some complexity inherent in the function f.

Within a single-use landscape focussed on growing apple (*Malus* species) for food provision (the desired ecosystem service), the ecosystem process function (f) can be further unravelled. We take pollination to illustrate this. Let the rate of pollination (P) be a function (m) of several stocks:

$$P = m(p_a, p_{d,} p_e, f, h_q, h_Q, h_s, a). \tag{2}$$

where p_a is pollinator abundance, p_d is pollinator diversity, p_e is pollinator efficiency, f is floral resource availability, h_q, is pollinator habitat quality, h_Q is pollinator habitat quantity, h_s is spatial configuration in relation to the apple orchard, and a is pollinator predator abundance.

To facilitate or boost pollination, managers can target interventions on any of the ecological components within equation (2), all of which are manageable natural capital stocks. In targeting interventions, managers need to be mindful of the critical interactions between stocks. For example, there would be little to gain from increasing pollinator abundance (p_a) if the quantity of pollinator habitat (h_q) was inadequate to support the enhanced pollinator population. Equation (2) is, again, a simplistic representation of underlying complexity. Pollination is but one ecosystem process that contributes to crop yield, others include nutrient cycling, climate regulation, and soil formation.

The ecological components expressed in equation (2) interact with other natural capital stocks with manageable and unmanageable attributes (e.g. soil and water which support landcover and land use and provide pollinator habitat), and are also influenced by the use of social and built capital. Social and built capital inputs (e.g. labour, fertilisation, irrigation, pest control) combine with natural capital to affect crop yield. Our simplified formalisation can be extrapolated to multifunctional landscapes, where several objectives spanning multiple ecosystem services are sought.

Managing natural capital stock attributes for the provision of ecosystem services

The presence and inherent capacity of natural capital stocks is variable through space and time. This variation is heavily influenced by the attributes stocks possess, and the characteristics of condition (quality, quantity, and spatial and temporal configuration) of these attributes (Yapp *et al.* 2010; Mace *et al.* 2015). Recognising that there is variability in natural capital stock attributes is an important step in implementing an ecosystem services approach. To do this effectively it is crucial to differentiate between 1) manageable stock attributes that contribute to ecosystem functions and processes (e.g. soluble phosphate in soils, or the introduction of plant species with desirable root strength), 2) unmanageable stock properties and/or life history traits that ensure long-term persistence of stocks (e.g. dispersal mechanisms), 3) characteristics that influence the condition of stocks in the context of ecosystem services provision (e.g. adequate quantity and quality of required stocks), and 4) drivers that exert influence on all these elements (e.g. environmental drivers or the influence of the market).

The spatial configuration of natural capital stocks and the extent to which this can be manipulated is an important attribute when managing ecosystem services. For example, investment in ecological infrastructure can be targeted at specific locations in a landscape to ensure the required ecosystem processes and associated services are available in the quantities required. Life history traits (Table 2.1) can be used when choosing species to increase natural capital stocks within landscapes (e.g. tree species with rapid growth, a long life span, or that are unpalatable to browsing animals). The condition of stocks can also be improved by *in situ* management. This can be a more cost effective approach than remediating the problems created by depletion of natural capital or reinstating natural capital in the future. The ability to manipulate natural capital stocks is particularly important for landscapes where stocks have lesser inherent capacity (quality or quantity) to provide desired functionality or where the stocks are under pressure or depleted.

Our conceptual framework illustrates the importance of managing natural capital stock attributes to sustain flow of ecosystem services, but by itself does not address the disjoint between the concept and its application. The complex interaction between stocks of natural capital and influences of other factors in the provision of ecosystem services requires decisions about management interventions to be made in a considered manner. Management decisions made within a well-defined process that forces objectives to be clearly defined and consequences of alternative actions to be weighed against each other have been shown to consistently deliver better outcomes. Structured decision making (Possingham *et al.* 2001; Gregory *et al.* 2012) provides a logical process to implement an ecosystem services approach in alignment with our framework identifying manageable attributes and intervention points.

Structured decision making compels the decision maker to make considered and informed choices via seven sequential steps: 1) Identify and define the problem; 2) Define objectives and performance measures; 3) Define actions and alternatives; 4) Estimate consequences; 5) Evaluate trade-offs; 6) Account for uncertainties; 7) Make decision (Gregory et al. 2012). Structured decision making is increasingly utilised to address conservation and environmental challenges (Guisan *et al.* 2013) and could be used for integrating an ecosystem service approach into applied resource management. We provide a generic example, the concept of which can be applied within any policy framework for the management of any natural capital stocks (Figure 2.2).

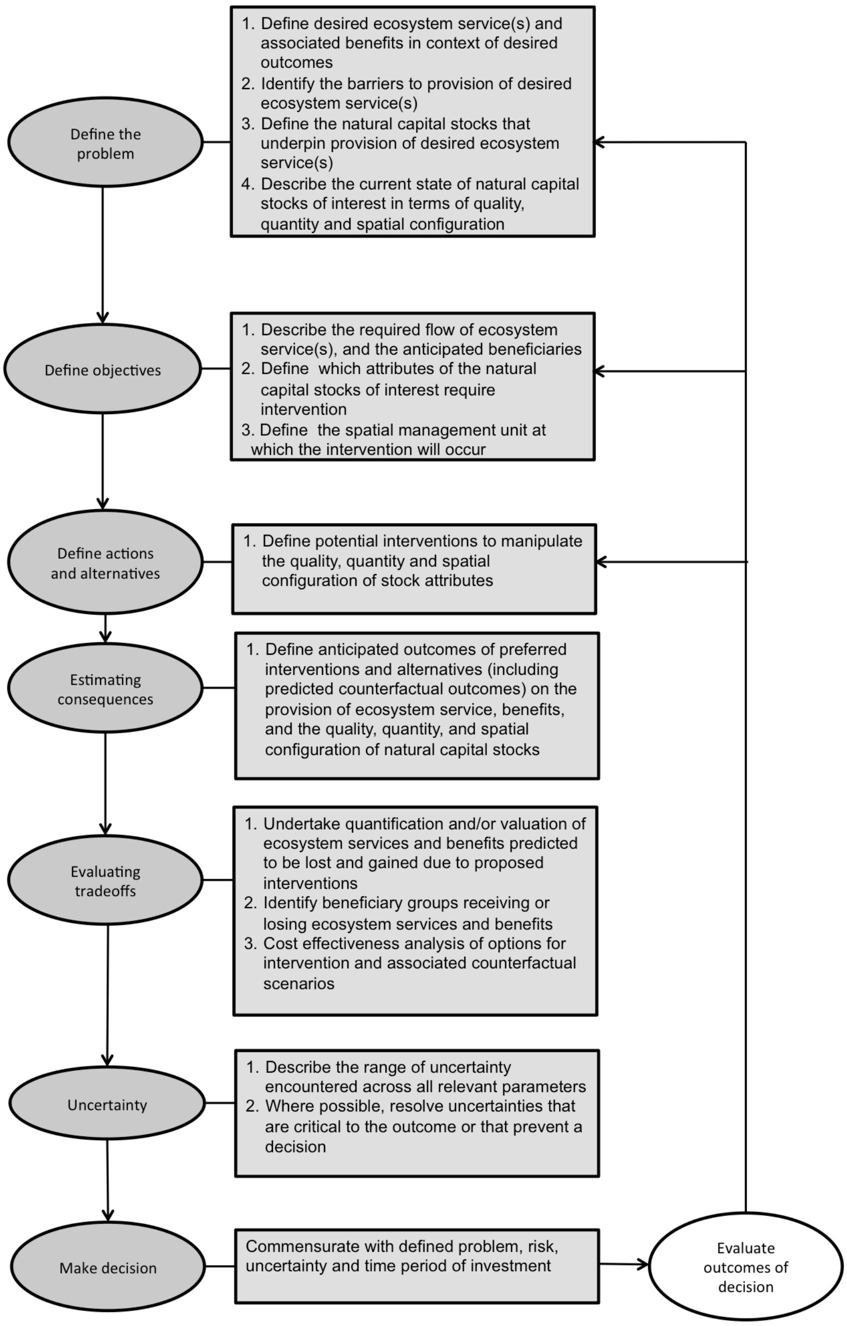


Figure 2.2: Accounting for ecosystem services in a structured decision making process for natural resource management decision making. The dark grey ovals represent the sequential steps in a structured decision making process, the light grey rectangles illustrate how ecosystem services can be incorporated into decision making. Implementation of actions, should not be the endpoint of the decision making process, and we emphasise the importance of monitoring and evaluating outcomes within the context of the objectives defined at the outset. Thus, we add an additional step in the structured decision making process 'evaluate outcomes of the decision' (white oval) which introduces feedback loops, informs further decisions, and is central to the process of adaptive management (Rumpff *et al.* 2011). Anticipated outcomes, consequences and trade-offs due to proposed interventions (actions) and counterfactual scenarios can be estimated using existing data and knowledge or evaluated using existing decision making support tools. For example, InVEST (© Natural Capital Project, www.naturalcaptialproject.org/InVEST.html, (Daily *et al.* 2009) which directly incorporates ecosystem services, or the Investment Framework for Environmental Resources (INFFER, www.inffer.org). Consequences (degradation or enhancement) for natural capital stocks of the proposed interventions can be described in terms of the quality, quantity, and spatial configuration of their attributes and evaluated against asset-benefit relationships (Mace *et al.* 2015).

This concept is illustrated by the implementation of management actions for the conservation of soil natural capital on hill country in the east coast, New Zealand as follows: *Problem definition:* loss of soil natural capital stocks due to accelerated hill country erosion, damage on-farm due to erosion and downstream due to flooding; *Objectives:* reduce risk of soil erosion in hill country, reduce downstream costs associated with sediment loadings in waterways, reduce damage to production farmland and settlements through siltation; *Actions and alternatives:* do nothing, spaced planting of trees in pasture to provide root reinforcement, land use change from grazing to commercial timber trees, retire pasture from grazing, raise stopbanks; *Estimated consequences:* change natural capital and thereby ecosystem services under spaced trees, economic and financial analysis under three scenarios of actions, soil conservation benefits; *Evaluation of trade-offs:* valuation of ecosystem services lost and gained with and without action options, benefit-cost analysis of four options of investment in soil conservation using spaced trees, accounting for and not accounting for ecosystem services; *Defining uncertainty:* described via sensitivity analysis; *Decision:* plant spaced trees in pasture (Dominati *et al.* 2014).

This example illustrates how by simultaneously considering individual stocks (soil and vegetation stocks) and the interactions between them, a wider range of ecosystem services and benefits can be targeted than if considering stocks independently. Decision making that ignores the full range of ecosystem services allows for trade-offs between services and benefits to be implicit, silent, and unaccounted for. In contrast, the explicit consideration of consequences of land use and management practice choices on natural capital stocks enables an ecosystem services approach to not only inform but drive natural resource management decision making.

Conclusions

Recognising the role that natural capital plays in the provision of ecosystem services is a necessary step towards managing for multifunctional landscapes. Ecological theory provides the relationship between stocks and processes and supports our premise that the manipulation of key stock attributes changes ecosystem function and service provision.

The conceptual ecosystem services framework presented here can apply to any natural capital stocks. The framework could also be extended to incorporate costs, benefits, and efficiencies of various actions. This would allow for increased transparency of decision making by accounting for trade-offs.

The conceptual framework presented here combined with a step-wise decision making process provides a transparent and adaptive structure for managing landscapes to gain sustained benefit across multiple ecosystem services. This is particularly relevant for regulating services, which are frequently undermined in single use landscapes but critical for sustaining long-term capacity. The more apparent the wider consequences of public and private choices are, the less likely they are to be ignored in the decision making process.

Glossary

Biodiversity: The term biodiversity is commonly used to refer to the diversity among living organisms across all organisational levels (genes, species, ecosystems), and is also commonly used as a collective noun to refer to species or ecosystems. Here we use the term in both these forms.

Ecosystem services: We define ecosystem services as the benefits flowing from natural capital stocks consumed or used by humans to sustain or advance wellbeing. This includes the goods generated by ecosystems that people value.

Ecosystem services approach: Is an approach to natural resource management that recognises that nature is critical to human wellbeing, and that this needs to be accounted for in the management of landscapes and systems in order to sustain the simultaneous provision of multiple ecosystem services. An ecosystem services approach is underpinned by the concept that ecosystem services flow from natural capital stocks.

Manageable attributes of natural capital stocks: Are those attributes that are responsive to manipulation. For example, organic matter or mineral nitrogen of soil stocks, and the morphological, physiological, and functional traits (e.g. growth form, root length, biomass, nitrogen fixing ability etc.) of plant species stocks are attributes that can be readily manipulated or selected for. The stock condition metrics of quality, quantity and spatial and temporal configuration can be improved via targeted management actions. Management of natural capital stocks can build the capacity of stocks and/or enhance interactions between stocks (e.g. above-below ground interactions) within short, policy-relevant, timescales.

Natural capital: Is the abiotic and biotic elements of nature, including all natural resources (such as soil, water, vegetation, species) and physical, biological, and chemical processes (Mace *et al.* 2015). Natural capital stocks are subject to many influences including environmental and social factors and other forms of capital, the combination of which gives rise to ecosystem services and benefits to humans. We differentiate between the natural capital stocks represented by resources (such as soil, vegetation, water, species) and ecosystem processes (such as pollination, nutrient cycling, energy transfer) (Figure 2.1) as this enables targeted intervention towards the attributes of natural capital stocks that can be manipulated to effect change in the provision of ecosystem services. Where the term biodiversity is used in reference to species and ecosystems, biodiversity is considered a stock of natural capital.

Unmanageable attributes of natural capital stocks: Are those attributes of stocks that are less amenable to management, and therefore less responsive to management induced manipulation within desired (policy-relevant) timescales. These attributes can be considered as fundamental to the nature of the stock (e.g. slope and orientation of landform, depth and texture of soils, vegetation community type etc.). Although some characteristic properties of stock attributes might be able to be manipulated, the substantial effort, cost, or time-period required to achieve change renders these attributes essentially unmanageable.

CHAPTER THREE:

EFFECT OF MANAGEMENT ON NATURAL CAPITAL STOCKS UNDERLYING ECOSYSTEM SERVICE PROVISION: A 'PROVIDER GROUP' APPROACH

As the ecosystem services concept gains traction, there is a need to increase understanding of how the provision of ecosystem services can be influenced at local and regional scales. Land use (e.g. farming) and management practices (e.g. grazing regimes) influence which ecosystem services flow from which landscapes. However, the nature of this relationship is currently not well understood and this knowledge gap prevents more appropriately targeted land use policies from being developed, and reduces effectiveness of current policies aimed at maintaining ecosystem services. In this chapter I test the impact of three management practices on species diversity and abundance using a 'provider group' approach, showing how management that targets specific attributes of species can influence the provision of ecosystem services and providing an empirical illustration of the conceptual framework presented in Chapter 2.

A version of this chapter has been submitted to Biodiversity and Conservation as:

Maseyk FJF, Demeter L, Csergő AM, Buckley YM. Effect of management on natural capital stocks underlying ecosystem service provision: A 'provider group' approach.

Abstract

Land management practices directly impact on the occurrence and condition of plant natural capital stocks as represented by species diversity and abundance measures. Species identity and quantity drive ecosystem service supply, either through effects of local diversity and/or through the presence of service providing species. However, the influence of management practices on the provision of ecosystem services at local and regional scales is not adequately understood. We grouped species into six sets of grassland plant species with desirable attributes which we recognise as ecosystem service ‘provider groups’, and tested the influence of three land management practices (abandonment of mowing, grazing, and mowing) on diversity and abundance within these groups in upland temperate grasslands of Transylvania (Romania) at local and regional scales. The response of diversity and abundance to management varied among provider groups, among diversity or abundance metrics, and with location. The three management practices, combined with landscape heterogeneity, influence species composition and favour certain provider groups, and hence the supply of certain ecosystem services, over others. The services required from landscapes are determined by societal needs and priorities. A greater understanding of how local-scale management practices impact on the provision of ecosystem services can inform the development of agri-environment schemes and conservation policies. The provider group approach is more useful than overall species diversity metrics for assessing the multiple social and environmental characteristics of traditional landscapes and the diverse needs of stakeholders. Applying a provider group approach presents an opportunity to get additional value from commonly collected biodiversity data to inform on ecosystem service provision and response to land management.

Introduction

Diversity metrics are commonly used as a proxy for ecosystem function and ecosystem service provision, particularly when assessing multifunctionality. Multifunctionality is achieved when landscapes simultaneously provide multiple ecosystem functions or services (Byrnes *et al.* 2014; Wagg *et al.* 2014). More diverse systems are thought to provide greater temporal stability of ecosystem functions and services than less diverse systems by providing both greater functional redundancy (the number of species contributing similarly to the same function) and response diversity (the variance in response to disturbance or changed environment conditions between functionally similar species) (Díaz & Cabido 2001). Thus, more diverse systems are more likely to contain species possessing traits that contribute to a given function year to year (Allan *et al.* 2011) and more resilience (the simultaneous resistance to and recovery from disturbance, Hodgson *et al.*

2015) thus providing 'functional insurance' (Díaz & Cabido 2001; Laliberte *et al.* 2010; Isbell *et al.* 2011; Hautier *et al.* 2014) and maintaining multifunctionality. While the importance of biodiversity for multifunctionality is recognized, specific understanding of the mechanistic relationship between biodiversity and ecosystem function and service provision continues to be elusive and simple generalizations are unlikely to be supported (Naeem & Wright 2003; Balvanera *et al.* 2006; Maskell *et al.* 2013; Duncan *et al.* 2015).

Multifunctional landscapes provide a wide range of ecosystem services, with considerable variation in how stakeholders perceive benefits arising from those services. Therefore the effects of changes in diversity on the benefits provided will depend both on which services are provided and the benefits sought by stakeholders. There are several ways in which the effects of management on ecosystem service provision can be explored: 1) assessing the effects of management on species diversity metrics in general, assuming that higher diversity generally will have a positive outcome on ecosystem service provision; 2) assessing the effects of management on individual species known to directly provide particular ecosystem services; or 3) assessing the effects of management directly on ecosystem service provision. However, these approaches do not describe the relationship between natural capital stocks and the provision of ecosystem services.

Assessing management effects on species diversity metrics in general can miss impacts of management on particular species that may be important contributors to the required ecosystem service(s). Management that maximises diversity may not always equate to maximising benefit, particularly where human preference for service provision varies across a landscape. In contrast to the very general approach of assessing effects of management on diversity of whole assemblages, assessing the effects of management on individual species that contribute to ecosystem service provision has more merit on local scales. However, multiple species often contribute to a single ecosystem service at local and larger scales and it is difficult to generalise effects where the species providing that service changes across a landscape or region. Assessing the effects of management on ecosystem service provision is problematic as it ignores the natural capital stocks and ecosystem processes that give rise to those services. Thus, while a change in ecosystem service provision may be detected, the underlying drivers remain unclear. This ambiguity also makes it difficult to generalise results across regions.

Here we test an alternative approach by which to assess the impact of management practice on the provision of ecosystem services. In order to better understand the mechanics of the relationship between biodiversity and ecosystem function and service provision, we need to move beyond overall diversity metrics and consider functional diversity and composition (e.g. Díaz & Cabido 2001; Naeem & Wright 2003; Cadotte *et al.* 2011; Lavorel *et al.* 2011; Mace *et al.* 2012; Duncan *et al.* 2015). Taking this more functional approach when considering diversity can improve management decisions aimed at sustaining ecosystem service provision. We extend the ecological functionality approach by considering attributes of natural capital stocks that have utilitarian function or benefit. For example, the attribute of producing nectar can serve as a proxy for both honey provision (direct human benefit) and pollination (ecological functionality). Species with similar attributes can be grouped into 'sets' of species sharing those attributes, similar in concept to groupings of species based on ecological function. The advantages of focusing on functional groups rather than species richness *per se* are recognized elsewhere (e.g. Naeem & Wright 2003; Laliberté *et al.* 2010; Allan *et al.* 2011). Luck *et al.* (2009) built on earlier work to develop the 'service provider' concept, which recognises "the quantification of organism, community, or habitat characteristics required to provide an ecosystem service in light of beneficiary demands and ecosystem dynamics". The service provider concept provides an expanded conceptual framework for the study of the relationship between biodiversity and ecosystem service provision (Luck *et al.* 2009). Here, we build on these existing functional and service provider approaches by recognising sets of species ('provider groups') that posses particular attributes (natural capital stocks) that contribute to the provision of ecosystem services and benefits to humans (either directly or indirectly). We use the provider group approach to test the impacts of management practice on the natural capital stocks that supply ecosystem services.

Ecosystem services of interest to our case study community include the supply of food, raw materials, medicinal resources, nutrient regulation, species of conservation concern, and sense of place. Thus, we identified six species attributes by which to categorise provider groups that contribute to the provision of these ecosystem services: 1) palatability and nutritional value; 2) medicinal or aromatic compounds; 3) nectar production; 4) pollen production; 5) nitrogen fixation; and 6) endemic and red listed species. For example, nectar producing species were grouped together in the 'honey provider group' and considered to contribute to the provision of both food and medicinal resources. Table 3.1 lists the six provider groups and illustrates the link between these groups and the supply of ecosystem services, showing where this link is direct or indirect.

Table 3.1: Species attributes used in this study to assign species into 'provider groups' for analysis and used as proxies to indicate the direct or indirect supply of ecosystem services. Interactions with other species (e.g. honey bees extracting nectar from nectar producing species and pollinators visiting pollen producing species), other forms of capital (e.g. financial, built, and social), and socio-political drivers is also required for the provision of ecosystem services.

Species attributes	**Provider group**	**Direct or indirect supply of ecosystem service**[1]	**Ecosystem service** P = Provisioning services R = Regulating services C = Cultural services
Pollen production	Pollen	Direct Indirect	Medicinal resources (P) Food (P)
Palatability and nutritional value	Quality fodder	Indirect	Food (P) Raw materials (P)
Medicinal or aromatic compounds	Medical and aromatic compound	Direct	Medicinal resources (P) Raw materials (P)
Nectar production	Honey	Direct	Food (P) Medicinal resources (P)
Nitrogen fixation	Nitrogen fixation	Direct Indirect	Nutrient regulation (R) Food (P)
Endemism or threatened with extinction	Conservation concern	Direct	Maintenance of species of conservation concern (C) Sense of place (C)

[1]Indirect supply requires an intermediate link between the species (within the provider group) and the supply of the service. Ecosystem processes provide this link e.g. via the process of pollination (pollen provider group), energy transfer/food webs (quality fodder provider group), or nutrient cycling necessary for maintenance of system function and support of fodder species (nitrogen fixation provider group).

We show that application of the provider group concept enables rapid assessment of how the natural capital stocks which contribute to a particular ecosystem service respond to different management regimes and across a landscape. This response may disproportionately affect particular provider groups and therefore particular services and benefits. Measuring species diversity within provider groups may serve as a useful proxy by which to evaluate the supply of target ecosystem services under various land management regimes. For example, by implementing management practices that are favourable to groups of plant species with both high nutritional value and which are of conservation concern, the production of stock fodder (allowing for food production which is a provisioning ecosystem service) can be sustained alongside the conservation of endemic and/or red listed species (a cultural ecosystem service) at the local scale. Alternatively, trade-offs can be recognized and different management practices can be implemented at different sites to optimise service provision at the local scale or regional scales and/or to deliver multifunctionality at the landscape scale.

We illustrate the provider group approach by testing the effects of different land management regimes on multiple ecosystem service provider groups, and consequently the provision of multiple ecosystem services, using a case study from the Southeastern Carpathian Mountains in Romania.

The land management practices of grazing and mowing (hay making) are known to influence species diversity and functional composition of plant communities by altering the balance of competition-colonization processes (Grime 1979; Tilman 1988), niche overlap (Mason et al 2011) and abiotic conditions (Köhler *et al.* 2001). However, we know less about how different subsets of species with attributes that contribute to ecosystem processes and provision of ecosystem services are affected by these management practices. For example, honey provider species may respond differently than other groups of species, such as those that are palatable to livestock.

We tested how different sets of grassland plant species with attributes desirable to local communities (provider groups) are impacted by three different land management practices: abandonment of hay meadows, grazing, and mowing (hay making) within the Southeastern Carpathian mountain hay meadows. We then contrasted the provider group approach with assessment of management effects using overall diversity and abundance measures. We hypothesized the following: 1) effects of management on species richness, evenness, diversity, and abundance will vary between provider groups and 2) landscape-level environmental variability and land use history will also influence species richness, evenness, diversity, and abundance within different provider groups. Measurement of the response to management regime at the provider group level enables the rapid assessment of how land use or management practice can impact on consequent ecosystem service and benefit provision. This framework can be generally applied to any system using existing species diversity data with the addition of data on provision of ecosystem services, as relevant to a local or regional setting. The results can be incorporated into structured decision making processes enabling stakeholders to assess effects of management on multiple services across a landscape against their stated objectives and develop landscape scale management plans.

Methods

The Southeastern Carpathian grasslands

The Southeastern Carpathian grasslands are part of a rural landscape where local communities have continued traditional semi-subsistence farming since the Middle Ages, actively maintaining forest-free habitats and shaping the landscape structure to provide summer grazing and winter fodder (hay) for livestock (sheep, cows, goats and horses) (Knowles 2011; Babai & Molnár 2014; Babai *et al.* 2015). As a consequence, large grassland areas were formed following the almost complete clearance of the deciduous forest land cover below 900 masl, and the partial deforestation of spruce and mixed broadleaf-coniferous forest cover at elevations above 900 masl (Csűrös *et al.* 1980). These historically human-induced, extensively managed grasslands represent areas of high biological diversity and habitats of important cultural and social value in Europe and worldwide (Knowles 2011; Wilson *et al.* 2012). Mountain hay meadows in particular are protected within the Natura 2000 framework of the European Union (European Commission 1992). However, species rich mountain hay meadows cannot be sustained without traditional agricultural practices, currently under threat from globalization, agriculture intensification and inadequate public policies (Knowles 2011).

In the Southeastern Carpathians, a 'two round fallow' agricultural land use practice remained essentially unchanged from medieval times until the mid 20th Century (Kozán 1978). In the 1960s and 1970s, rural populations declined sharply as industrialization caused the mass migration from rural settlements to urban centres. In parallel, agriculture intensified and stock numbers reached their peak. To provide adequate year-round fodder for this increased stocking rate, all available grassland habitat in the mountains was either grazed or mown for hay. During this period, agricultural land was used under a combination of family farms, collective farms, and state farms. With the fall of communism in 1989, the practice of collective farming ceased and state farms began to disappear. Family farms were re-established and their numbers increased again although total stock numbers, and thus the demand for hay, declined. Consequently, farmers began to abandon hay meadows. Abandonment occurred in a non-random pattern, with the most remote hay meadows being abandoned first (Demeter & Kelemen 2012). 'Partial abandonment' also occurred whereby good quality meadows were grazed by sheep, but not mown.

Romania joined the EU in 2007 leading to further changes in the landscape. The European Union's (EU) Common Agricultural Policy (CAP, European Commission 2012) relies on economic agri-environment schemes to support agriculture and sustain biodiversity within the EU. Natura 2000, the largest network of protected areas in the world and a cornerstone policy for biodiversity conservation within the EU also allows for agricultural practices (European Commission 1992, 2009). While both the CAP and Natura 2000 encourage the maintenance of traditional land use practices, neither has been adequate to support traditional small-scale farming specific to East-Central European cultural landscapes (Babai *et al.* 2015; Sutcliffe *et al.* 2015). For example, under the 2007–2013 format, the CAP agri-environment scheme did not differentiate between mown and grazed grassland systems, instead offering the same level of payment to both practices. As grazing is a less labour intensive and less costly practice, the scheme indirectly favoured grazing over mowing, causing loss in species diversity (Baur *et al.* 2007; Demeter & Kelemen 2012). As a result of this environmental history, a mosaic of three dominant land cover types exists across the Southeastern Carpathians: pasture, hay meadows and regenerating secondary forest transitioning towards mixed Norway spruce (*Picea abies* (L.) H. Karst.)—European beech (*Fagus sylvatica* L.) forests typical of the area (Figure. 3.1).

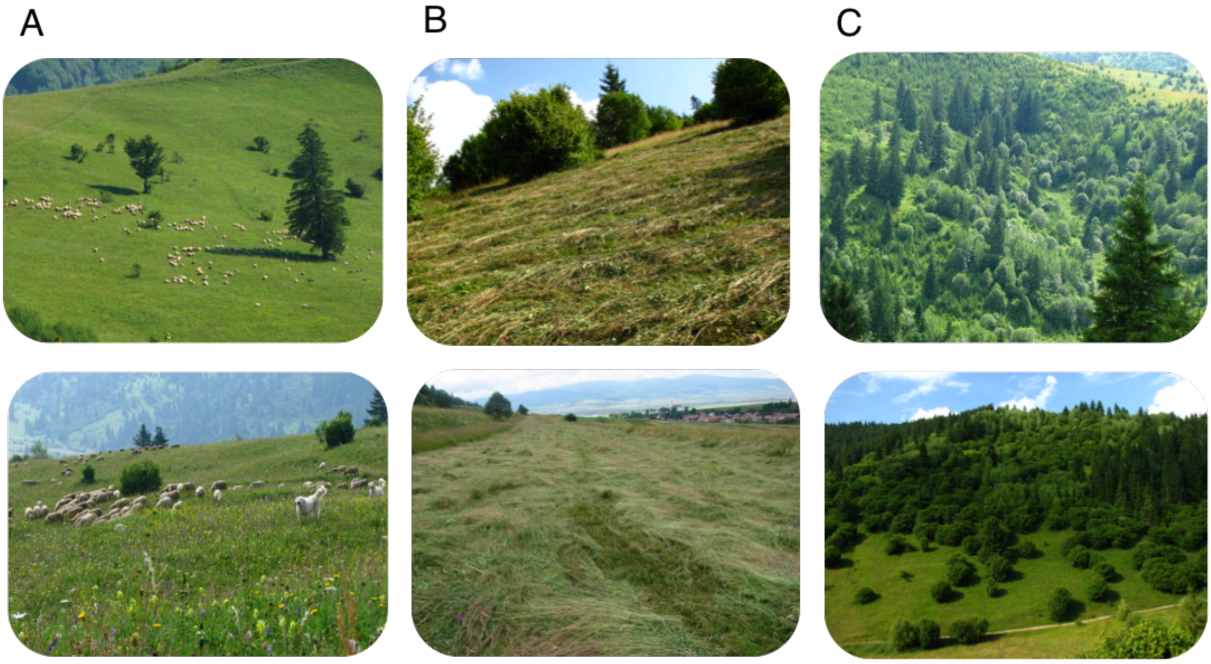


Figure 3.1: The dominant land cover types within the study area associated with the three management practices gazing, mowing, abandonment of hay meadows: A) pasture (grazing); B) hay meadows (mowing); and C) abandoned hay meadows (abandonment) and regenerating secondary forest.

Site description

The study was conducted in two grassland systems within the Southeastern Carpathians, Romania (46°41'N, 25°94'E). The first sampling site was located at Somlyó Valley (Somlyó) within the Csík Basin (Depresiunea Ciucului) and the second at Kolos, within the Csík Mountains (Munții Ciucului). The sampling localities are subject to a boreal-mountainous climate with an average annual precipitation of 580 mm in the Csík Basin increasing with elevation to a maximum precipitation of 1000–1200 mm/year. The geological substrate is Mesosoic flysch (sedimentary rocks) (Ielenicz & Pătru 2005).

The Kolos site covered 1.8 km^2 at an elevation range of 942–1292 masl and was relatively remote (15 km) from the nearest settlement. In contrast, the relatively lower elevation (764–802 masl) site Somlyó covered 1.3 km^2 and was located within 2 km of the nearest settlement. Hay making frequency varied between the two sites, with meadows being mown only once a year in Kolos and once or twice (but most often twice) a year at Somlyó. Temporary grazing of sheep during autumn may occur within hay meadows at both sites and lower elevation meadows may also be fertilized with manure. These are typical management regimes for hay meadows in both remote and less remote areas.

Grassland communities at the Kolos site were co-dominated by common bent and Chewings fescue *Festuca nigrescens* Lam., creating a *Scorzonero roseae-Festucetum nigricantis* community (Puşcaru et al. 1956) Coldea 1987). At the Somlyó site common bent *Agrostis capillaris* L. and red fescue *Festuca rubra* L. co-dominated creating an *Agrostio-Festucetum rubrae* (Horvat (1951) 1952) community, reflecting differences in environmental conditions between the two sites.

Data collection

Within each of the two sampling sites (Kolos and Somlyó), three distinct management practices (abandonment of hay meadows, grazing and mowing (hay making)) were identified (Figure. 3.2).

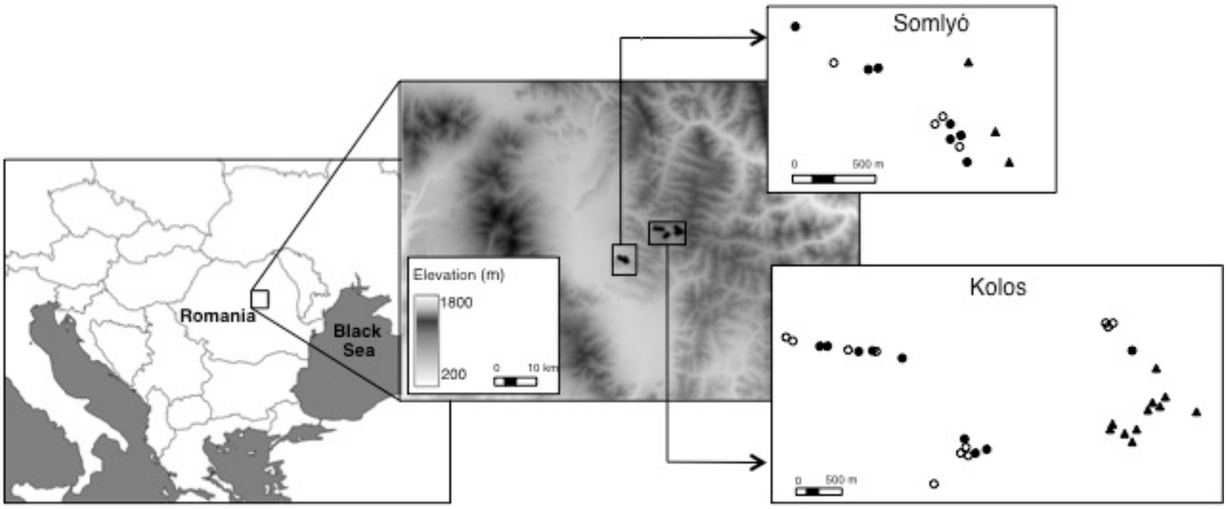


Figure 3.2: Map showing the location of the study area in Romania and the distribution of the 1 m^2 plots at the Kolos (bottom) and Somlyó (top) sites by management practice. Symbols represent management practice: abandonment of hay meadows (hollow circles); grazing (triangles); mowing (solid circles).

Grazing has traditionally been spatially separated from mowing at both sites, but abandonment of hay meadows within a site occurred more or less randomly, depending on individual farmer choice. Within our study, abandoned hay meadows had been unmown for between 3–5 years. To capture environmental heterogeneity, 1 m^2 plots were randomly located within areas under each management practice (abandonment of hay meadows, grazing, and mowing). A total of 31 plots were located at the Kolos site, 11 plots within abandoned hay meadows, 11 within pastures (grazed), and nine within hay meadows (mown). At the Somlyó site four plots were located within abandoned hay meadows, three within pastures, and seven within hay meadows (a total of 14 plots). Thus, there was a total of 45 plots between both sites, randomly located across each of the land cover types representative of the three management practices. The number of plots varied due to the different sizes of sampled land parcels. Abandoned hay meadow, pastures, and hay meadows were therefore sampled with 15, 14, and 16 plots respectively (summed across sites). Within each plot, a list of all taxa present was compiled, and abundance (percent cover) visually estimated for each taxon present and for total vegetation cover.

Data analysis

Classification of species possessing palatability and nutritional value, medicinal or aromatic compounds, nectar production, pollen production, and nitrogen fixation attributes follow Csűrös, Csűrös-Káptalan and Resmeriță (1970). Species listed in this publication under the classifications 'excellent', 'very good' and 'quality' fodder plants were considered to posses palatability and nutritional attributes.. The checklist of endemic and red listed species was extracted from a regional list of endemic and protected plant species (A. Csergő *pers. com*) which was based on the national red list of vascular plants and the national checklist of plant species of Romania (Oltean *et al.* 1994; Oprea 2005) and unpublished data. We applied these characteristics to define our six provider groups: pollen; quality fodder; medicinal and aromatic compound; honey; nitrogen fixation; and conservation concern (Table 3.1).

Sampled species were assigned to the appropriate provider group(s) where they exhibited the described characteristics (attributes). Species were assigned to as many of the six provider groups for which they possessed defining characteristics. We assume that maintaining species diversity and abundance within a given provider group will maintain the flow of services associated with that provider group.

The response variables were 1) species richness (number of species), 2) species evenness (Evar; Smith & Wilson 1996), 3) species diversity (Inverse Simpson Diversity (=1/Simpson's Diversity)), and 4) relative abundance within each 1 m^2 plot for each provider group (vegan package, R, Oksanen *et al.* 2013). We calculated the relative abundance of each species by dividing its abundance by the total vegetation cover of all species within the plot and summed these by provider group. In order to compare our approach to more traditional approaches which use species diversity metrics as a proxy for ecosystem service provision and multifunctionality, we also calculated the same metrics over all species within each plot. We tested for significant differences in our response variables across management practice (abandonment of hay meadows, grazing, and mowing) and sites (high elevation Kolos and lower elevation Somlyó) for each provider group. Pastures were spatially segregated within sites so we analysed the data using a randomized complete block design with subsampling (RCBs) in which the two sites represent blocks, the three treatments (management practices) are spatially segregated within each block and the 1 m^2 plots represent subsamples within each treatment. We fitted linear mixed effects models in which both management practice and sites were introduced as fixed effects and subsamples (specified as interaction of management practice and site) were specified as a random effect. The sub-sampling within management treatment approach allowed for the observed spatial segregation of grazed plots within the sites and enabled testing of the main effects of management and site. This design did not allow for detection of interaction between management and site. Models were fitted in the *nlme* package in R.1.1 (Pinheiro *et al.* 2016), using the '*lme'* command.

Residuals were checked for normality (residuals plots) and heteroscedasticity (Q-Q plots, Bartlett and Levene tests), and data were transformed when needed (Warton & Hui 2011). To estimate effects of management practice and site on provider groups, a robust ANOVA procedure was applied on the models, in which we specified type II errors and we used a heteroscedasticity-corrected coefficient covariance matrix in the event of unequal variances that could not be detected due to low sample size (ANOVA, car package, R.1.1, Fox & Weisberg 2011). This approach computes χ^2 tests for the fixed effects in the mixed effects models. Subsequently we performed pairwise comparisons of the three management categories using post-hoc Tukey-Kramer honest significant difference tests (multcomp package, R.1.1, Hothorn *et al.* 2008), robust against low and unequal sample sizes.

Results

210 taxa were recorded across 45 plots in two sites. Of these, 127 (60%) species were identified as pollen providers, 43 (20%) species as nectar providers, 36 (17%) species as providers of medicinal or aromatic compounds, 26 (12%) species were endemic or red listed species, 20 (10%) species provided quality fodder and 18 (9%) species were nitrogen fixers. A total of 162 species (77%) exhibited at least one attribute of interest and over a third (76 species, 36%) of these species

contributed to more than one provider group. All provider groups except for species of conservation concern were represented by at least one species within each 1 m^2 plot. We first outline the overall species diversity and abundance differences due to management and site and then present results for individual provider groups.

Effects of management and site on total diversity and abundance of species

Overall species richness was significantly higher in mowed than grazed plots, while species diversity was significantly higher in abandoned plots than in grazed plots. Total vegetation cover (abundance) was significantly higher in mowed and grazed plots compared with abandoned plots (Table 3.2, Figure 3.3). Total species richness and diversity were significantly higher at the high elevation site Kolos compared to the lower elevation site Somlyó (Table 3.2, Figure 3.3). Total vegetation cover and species evenness did not differ significantly between the two sites (Table 3.2, Figure 3.3).

Table 3.2: Summary of effect of management and site on total diversity and abundance of species. Letters *a-b* indicate significant differences between arithmetic mean values. A = abandoned, G = grazed, M = mowed.

Response	Management Action or Site	Effect size (mean ± s.d.)	Tukey test, *p*	Management or Site χ^2(1, N = 45)	Management or Site *p*
Richness	Abandoned^{a,b}	33.3 ± 5.0	0.02(G-M)	8.13	0.02
	Grazed^{a}	32.1 ± 5.1			
	Mowed^{b}	36.4 ± 5.8			
	Kolos^{a}	35.7 ± 5.5	-	14.51	<0.001
	Somlyó^{b}	30.2 ± 3.2			
Evenness	Abandoned	0.2 ± 0.05	>0.78	0.45	0.80
	Grazed	0.2 ± 0.03			
	Mowed	0.2 ± 0.04			
	Kolos	0.2 ± 0.03	-	0.67	0.41
	Somlyó	0.3 ± 0.1			
Diversity	Abandoned^{a}	6.1 ± 1.2	<0.01 (G-A)	6.68	0.04
	Grazed^{b}	5.2 ± 1.4			
	Mowed^{ab}	5.6 ± 1.5			
	Kolos^{a}	4.7 ± 0.4	-	28.99	<0.001
	Somlyó^{b}	4.0 ± 0.4			
Abundance	Abandoned^{a}	88.3 ± 8.8	<0.01 (A-M, A-G)	13.5	<0.01
	Grazed^{b}	95.4 ± 3.7			
	Mowed^{b}	97.0 ± 5.9			
	Kolos	92.4 ± 7.8	-	1.73	0.20
	Somlyó	95.8 ± 5.7			

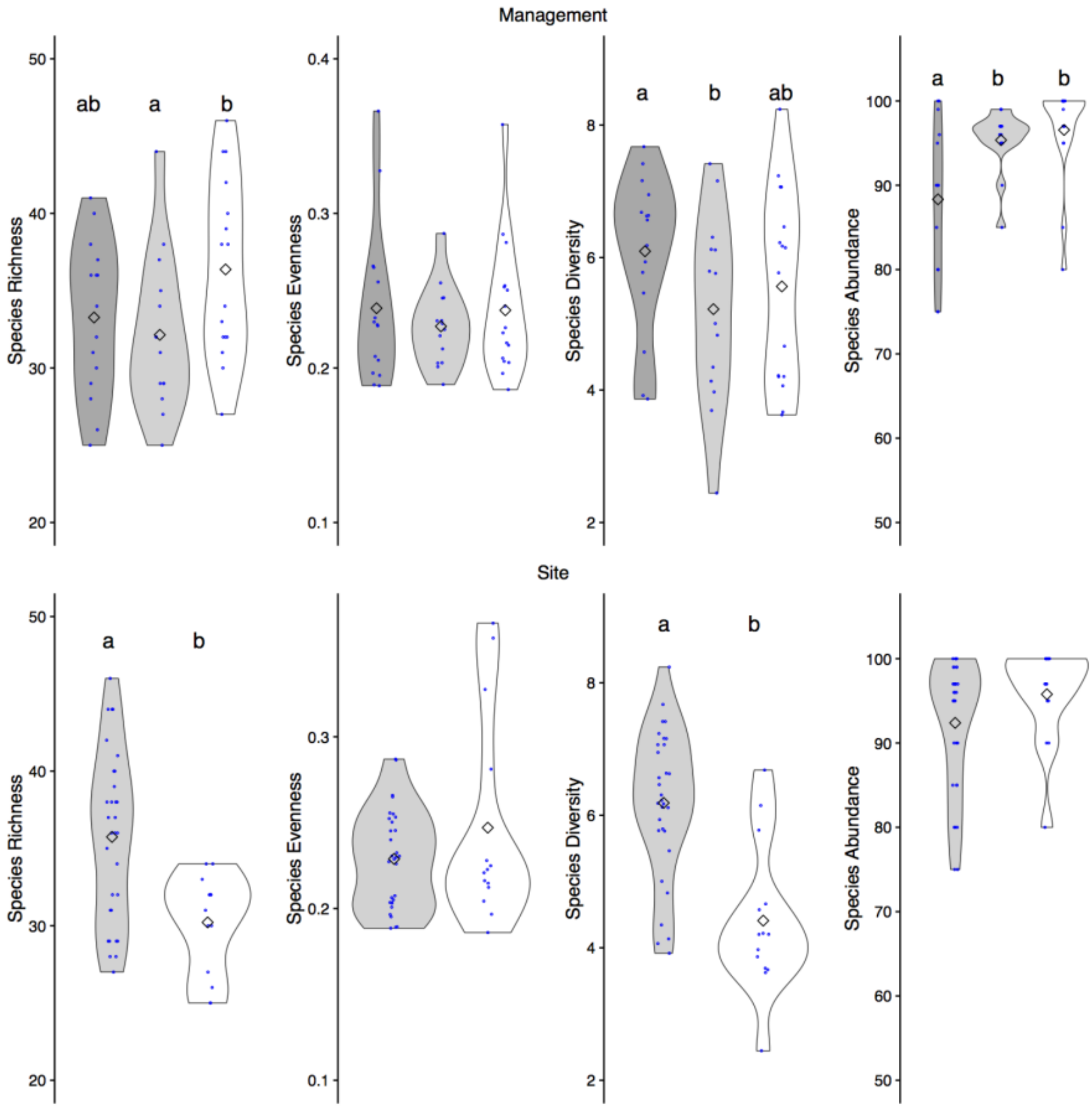


Figure 3.3: Overall species richness, evenness, diversity, and abundance under three management practices: abandoned (dark grey), grazed (light grey), and mowed (white) (top panel), and across two study sites: Kolos, higher elevation (light grey) and Somlyó, lower elevation (white) (bottom panel). Medians are marked with diamonds inside the violin plots, and dots represent raw data. Letters *a-b* indicate significant differences between arithmetic mean values (Linear Mixed Effects models and ANOVA test details are presented in the text).

Effect of management regime on richness, evenness, and diversity within provider groups

The effect of management on species richness, diversity, and evenness varied depending on the provider group (Figures 3.4 & 3.5). Effects of management on species richness were found only for the pollen provider and species of conservation concern groups. Similarly to the overall results, there was significantly higher species richness within the pollen provider group in mowed (25.3 ± 5.3) than in grazed (20.6 ± 4.7) plots but in contrast to the overall results there was also higher pollen provider species richness in mowed than in abandoned (22.2 ± 3.7) plots (Tukey tests $p < 0.05$ Management $\chi^2(2, N = 45) = 16.6$, $p < 0.001$). In accordance with the overall effects there was significantly higher richness of species of conservation concern in mowed (2.6 ± 2) than in grazed (1.8 ± 1.6) plots but contrary to the overall effects there was also a higher species richness in species of conservation concern in abandoned (3.1 ± 1.8) than in grazed plots (Tukey tests $p < 0.05$; Management $\chi^2(2, N = 45) = 10.7$, $p < 0.01$).

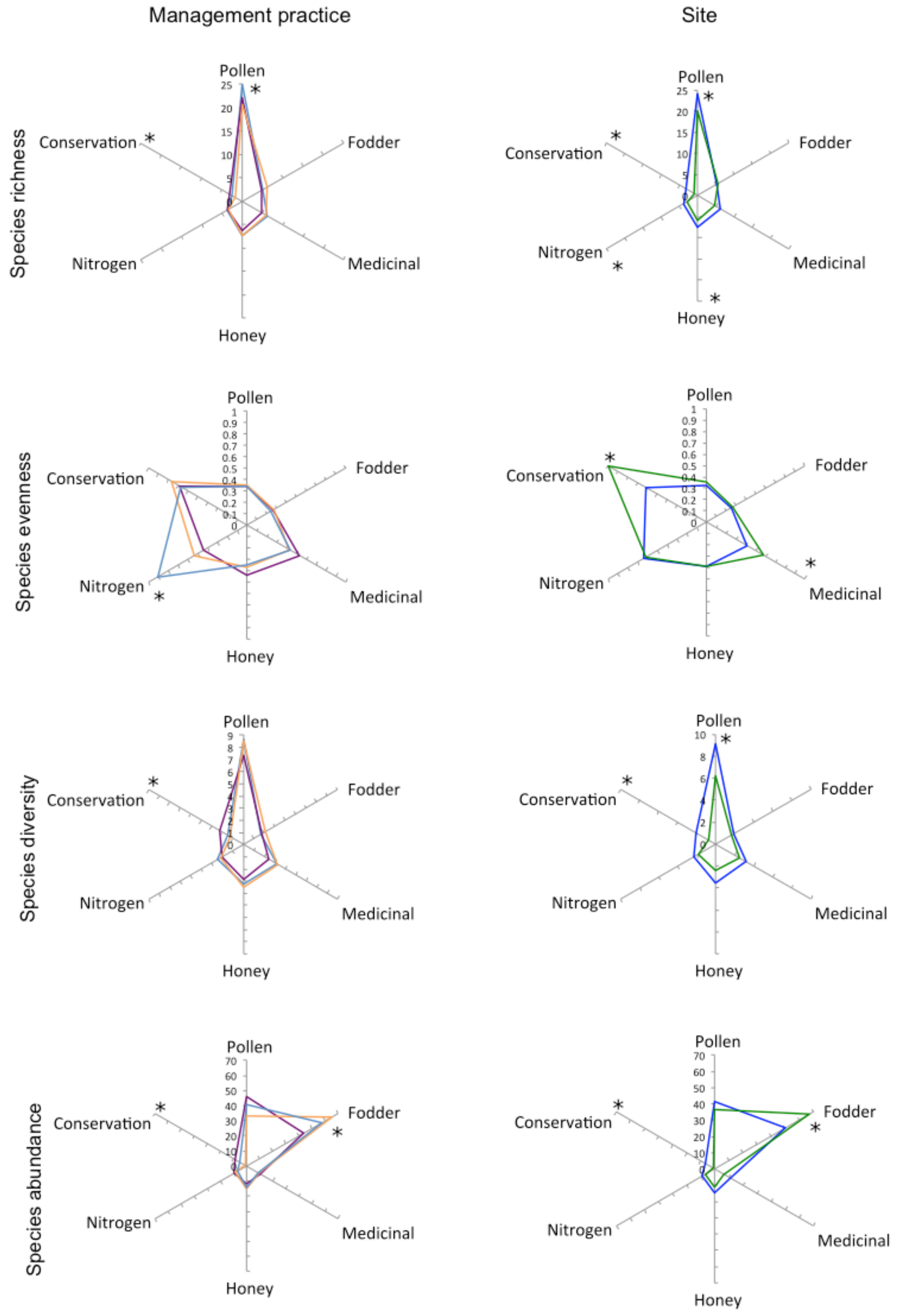


Figure 3.4: Species richness, evenness, diversity, and abundance within each provider group (pollen, quality fodder, medicinal and aromatic compound, honey, nitrogen fixation, and conservation concern) under three management practices: abandonment of hay meadows (purple line), grazing (orange line), and mowing (light blue line); and sites: Kolos, higher elevation (dark blue line) and Somlyó, lower elevation (green line). Stars represent significant management regime and site effects detected with the ANOVA test. Axes represent arithmetic mean of metrics calculated within each 1 m^2 plot for each provider group across the three management categories and the two sites.

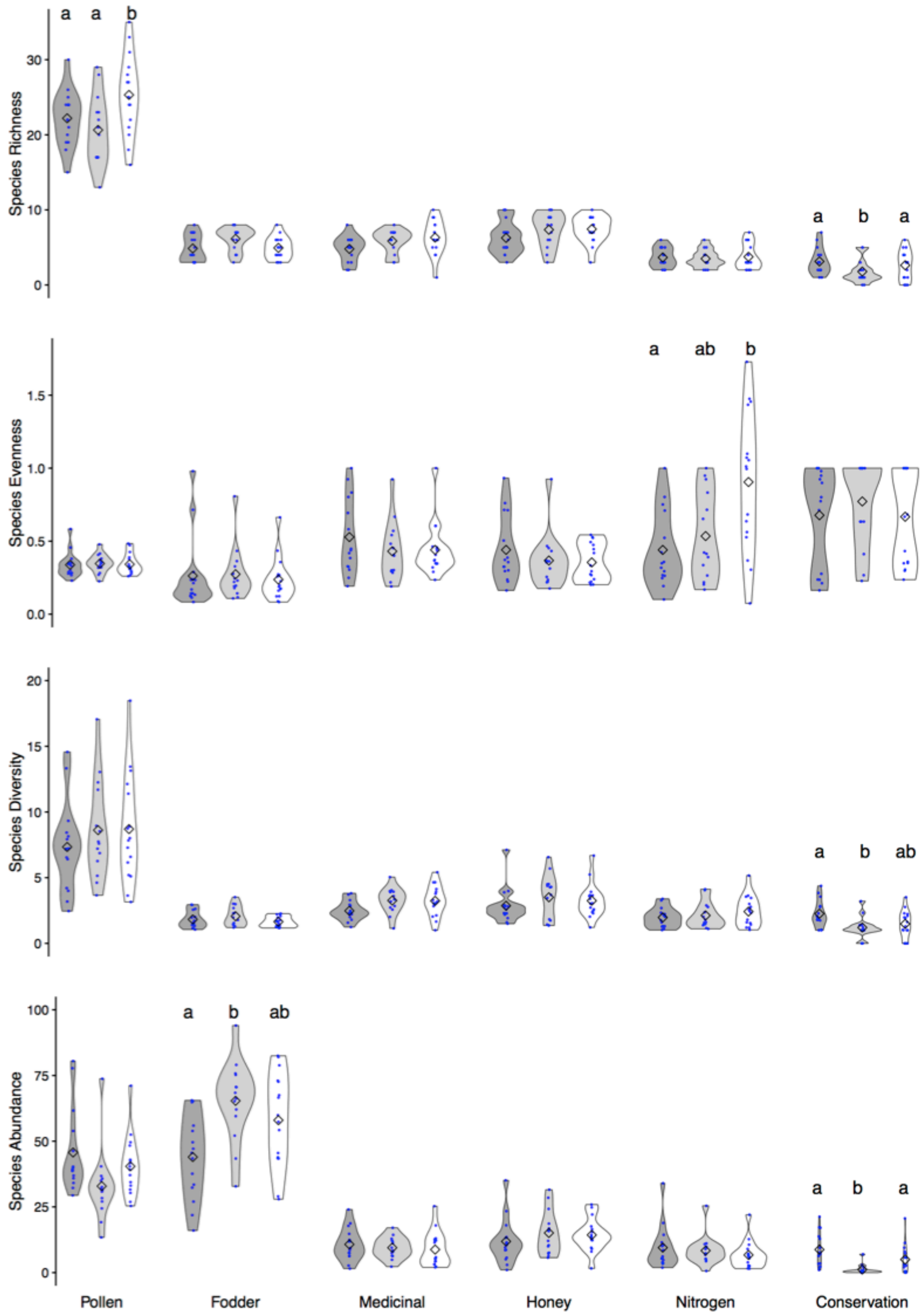


Figure 3.5: Provider group (pollen, quality fodder, medicinal and aromatic compound, honey, nitrogen fixation, and conservation concern) species richness, evenness, diversity, and abundance, across the three management practices: abandoned (dark grey), grazed (light grey), and mowed (white). Medians are marked with diamonds inside the violin plots, and dots represent raw data. Letters *a-b* indicate significant differences between arithmetic mean values (Linear Mixed Effects models and ANOVA test details are presented in the text).

While no effect of management was found on overall evenness there was a significantly higher evenness within the nitrogen fixation group in mowed (0.9 ± 0.5) than in abandoned plots (0.4 ± 0.3) (Tukey test $p < 0.05$; Management $\chi^2(1,2) = 7.9$, $p < 0.05$).

Effects of management on diversity were found only for the species of conservation concern provider group. In accordance with the overall effects there was significantly higher species diversity within the group of species of conservation concern in abandoned (2.3 ± 1.0) than in grazed (1.2 ± 0.8) plots (Tukey test $p < 0.001$; Management $\chi^2(2, N = 45) = 16.6$, $p < 0.001$) and similarly to the overall effects, species diversity within the group of conservation concern did not differ between mowed and grazed plots.

Effect of management regime on relative abundance of provider groups

Differences in relative abundance were only detected for the fodder and conservation concern provider groups, which responded differently to the three management actions (Figures 3.4 & 3.5). In agreement with the overall results, the abundance of species within the quality fodder provider group was significantly higher in grazed (65% ± 15) than in abandoned (44% ± 16) plots (Tukey tests $p < 0.001$; Management $\chi^2(2, N = 45) = 17.9$, $p < 0.001$). However, contrary to the overall results, the abundance of species within the quality fodder group did not differ significantly between abandoned and mowed plots. There was a significantly higher abundance of species of conservation concern in mowed (5% ± 6) and abandoned (9% ± 6) plots than in grazed (0.1% ± 0.2) plots (Tukey tests $p < 0.05$; Management $\chi^2(2, N = 45) = 17.3$, $p < 0.01$), in contrast to the overall results where total vegetation cover was significantly higher in grazed and mowed plots relative to abandoned plots.

Effect of site on richness, evenness, and diversity within provider groups

In agreement with the overall effect, there was significantly higher species richness at the high elevation site (Kolos) than at the lower elevation site (Somlyó) within the pollen (24.1 ± 4.9 vs. 20.0 ± 3.6, $\chi^2(1, N = 45) = 16.0$, $p < 0.001$), honey (7.6 ± 2.0 vs. 5.9 ± 2.0, $\chi^2(1, N = 45) = 8.9$, $p< 0.01$), nitrogen fixation (4.0 ± 1.5 vs. 2.9 ± 0.7, Site $\chi^2(1, N = 45) = 8.0$, $p < 0.01$), and species of conservation concern (3.3 ± 1.7 vs. 0.9 ± 0.9, $\chi^2(1, N = 45) = 32.0$, $p < 0.001$) provider groups (Figures 3.4 & 3.6). In contrast to the overall effect, species richness within the fodder and medicinal compound provider groups did not differ significantly between the two sites.

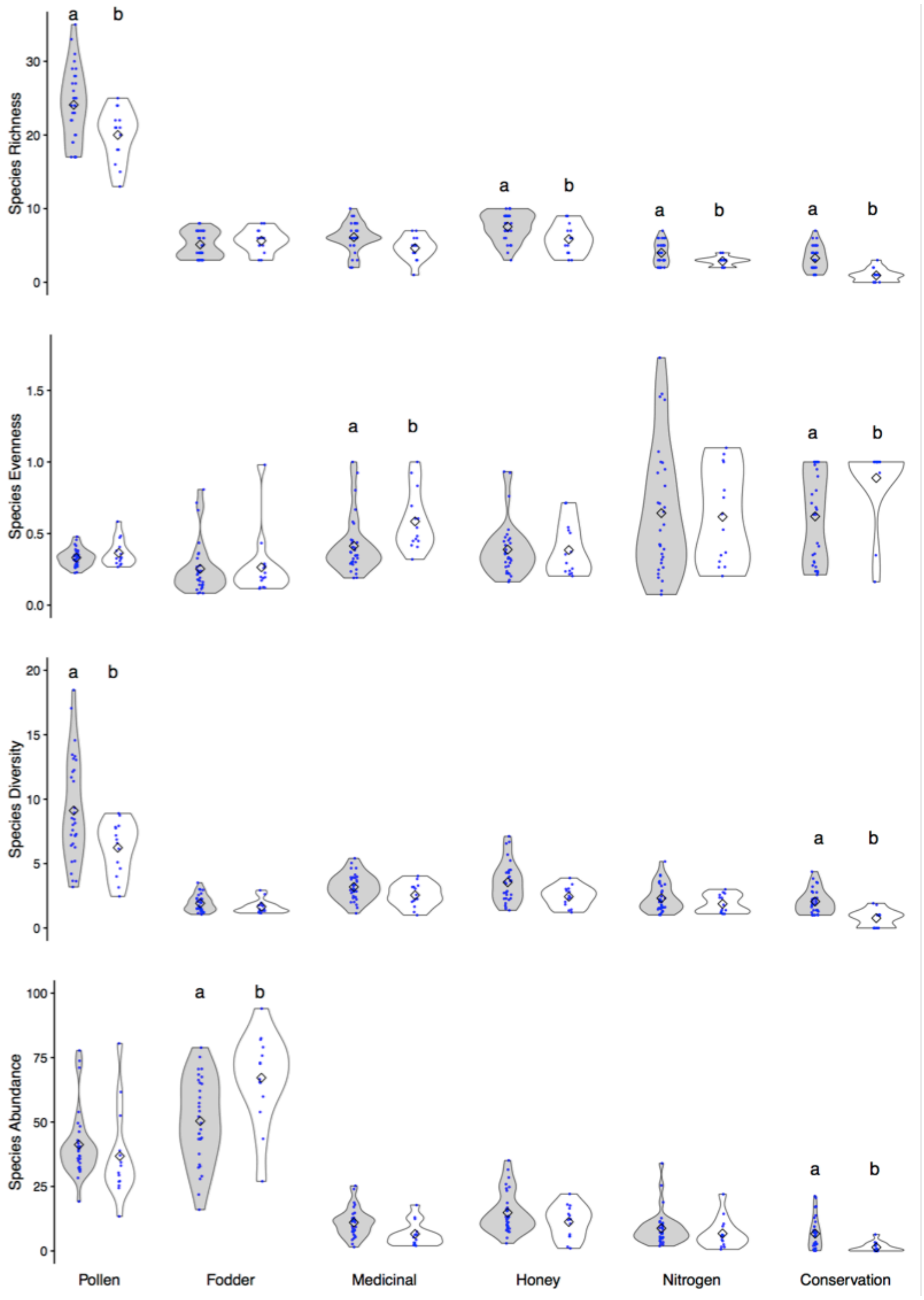


Figure 3.6: Provider group (pollen, quality fodder, medicinal and aromatic compound, honey, nitrogen fixation, and conservation concern) species richness, evenness, diversity, and abundance across the two study sites: and Kolos, higher elevation (light grey) and Somlyó, lower elevation (white) Medians are marked with diamonds inside the violin plots, and dots represent raw data. Letters *a-b* indicate significant differences between arithmetic mean values (Linear Mixed Effects models and ANOVA test details are presented in the text).

Whereas no effect of site was found on overall species evenness, species evenness within both the medicinal and aromatic compound (0.6 ± 0.2 vs. 0.4 ± 0.2) and conservation concern (0.9 ± 0.3 vs. 0.6 ± 0.3) provider groups was higher at Somlyó than Kolos sites (Site $\chi^2(1, N = 45) > 9.3$, $p < 0.01$). In agreement with the overall effect, species diversity was also significantly higher at Kolos compared to Somlyó within the pollen (9.1 ± 4.0 vs. 6.2 ± 2.1, Site $\chi^2(1, N = 45) = 6.6$, $p < 0.05$) and species of conservation concern (2.1 ± 0.9 vs. 0.8 ± 0.7, Site $\chi^2(1, N = 45) = 29.8$, $p < 0.001$) provider groups, however no effect was detected in the other provider groups.

Effect of site on relative abundance of provider groups

While no overall effect of site on total vegetation cover was found, the relative cover of the quality fodder provider group was significantly higher at the lower elevation site (Somlyó) than at the high elevation site (Kolos) (67 ± 17% vs. 50 ± 16% mean±SD, Site $\chi^2(1, N = 45) = 15.3$, $p < 0.001$) (Figures 3.4 & 3.6), whereas species of conservation concern showed the opposite pattern with greater relative cover at Kolos than at Somlyó (7 ± 6% vs. 1 ± 2%, Site $\chi^2(1, N = 45) = 15.0$, $p < 0.001$).

Discussion

Our testing of the provider group concept illustrates this is a useful approach by which to assess the impacts of management practices on the provision of ecosystem functions and services. We found that the effects of management on richness, evenness, diversity, and abundance within provider groups depend on provider group identity and the presence and direction of these effects do not always correspond with overall differences in the response metrics. Changes in diversity as a result of management affect some provider groups more than others; therefore, assessing diversity changes in response to management for the whole community may not inform on provision of particular services, functions or benefits. In addition, the metrics of species richness, evenness, diversity, and abundance responded to management differently depending on the provider group, indicating that a suite of indices should be used to assess effects of management. For services where species within a provider group are substitutable for service provision, provider group abundance may be an appropriate metric as species identity is less important, whereas for services where species are complementary (each species contributes to the provision of a different service), species richness and/or diversity may be more appropriate.

The effect of management can generally be explained by mechanisms that increase or decrease overall species richness and diversity, or favour some species over others. For example, in situations where herbivory increases light availability diversity can be maintained at local scales (Borer *et al.* 2014). Further, grazing pressure favours species with herbivore avoidance, tolerance or defence mechanisms, such as short, prostrate growth forms, high vegetative regeneration ability after defoliation, spiny leaves, and toxic chemical compounds (Grime 1979). In contrast to grazing, mowing regimes deplete nutrients from grasslands due to complete biomass removal and enables the persistence of shade-intolerant, fast growing and early season species (Tilman 1988; Zobel 1992; Tilman 1997; Grime 2001). If these ecological strategies are not evenly distributed among provider groups then we expect grazing or mowing to differentially affect benefits provided by various provider groups. General responses to both grazing and mowing were evident in our study. Several species within the quality fodder provider group avoid or recover rapidly following grazing and were, unsurprisingly, more abundant under a grazing regime (e.g. *Trifolium repens*, *Lolium perenne, Medicago lupulina*). Fast growing and early season species and species sensitive to trampling and nutrient enrichment by livestock were also well represented within the provider groups that did better under a mowing than a grazing regime (e.g. pollen providers and species of conservation concern: *Gentiana verna, Gymnadenia conopsea*).

Despite the evidence that some species did well under grazing pressure, grasslands managed as hay meadows (mowing) using traditional techniques (late mowing, no fertilization other than application of manure, no over-sowing, little mechanization) favoured more provider groups (pollen, conservation concern, quality fodder, and nitrogen fixation) than grasslands subjected to intensive grazing (which strongly favoured only the quality fodder group). This finding aligns with patterns of reduced diversity under uncontrolled, intensive grazing regimes, particularly by sheep, observed elsewhere in upland Carpathian grasslands (Baur *et al.* 2007; Başnou *et al.* 2009).

In comparison, important shifts in species composition occurred within abandoned hay meadows: the pollen provider, nitrogen fixation, and quality fodder groups were disadvantaged, in contrast to the species of conservation concern which were not disadvantaged by the practice of abandoning hay meadows. In the studied communities, abandonment of hay meadows enables the successional shift from grassland to tall forbs and then eventually to forest cover (Csergő *et al.* 2013). This successional process is expected to influence species diversity and composition due to decreasing light levels and slowed rates of mineralization and recycling, which creates conditions more favourable to competitive nutrient-conservative species (Kahmen & Poschlod 2004; Robson *et al.* 2007; Jacquemyn *et al.* 2011). The reduced cover of quality fodder species within abandoned hay meadows compared to their dominance under grazing and to some extent mowing regimes can be attributed to the sensitivity of these species to shading and displacement by tall forbs (Herben *et al.* 1994; Csergő *et al.* 2013).

Loss in species diversity within a provider group can have immediate consequences for the provision of associated ecosystem functions and services (e.g. decreased pollen production by loss of pollen provider species) and cascading impacts e.g. the loss of pollen provider species from grazed meadows may impact the retention of associated pollinator species within the landscape and disrupt species-mutualisms. However, if loss in species diversity generally is due to the loss of species that were non-contributors in the context of the desired functions or service (e.g. not a pollen provider), the provision of ecosystem function or service could remain uninterrupted. Such non-contributors could include any of the 48 species recorded in our study that did not posses any of our six attributes of interest (e.g. *Deschampsia cespitosa, Luzula campestris*, *Carex ovalis*).

Management-induced change in abundance of individual species can impact on ecosystem service provision if species are not substitutable in their ecosystem service provision even if diversity metrics are relatively insensitive to that change. Alternatively, provision of ecosystem function and services may continue if a decrease in abundance of one species removes competitive dominance and allows alternative species with the same attribute to increase in abundance, avoiding any interruption in provision of service. For example, in our system, *Festuca rubra* and *Agrostis tenuis* are more dominant in mesic, non-fertilized meadows compared to *Festuca pratensis* which may dominate in moist and more fertile meadows. It can be surmised that should conditions become wetter and more fertile, *A. tenuis* and *F. rubra* would lose dominance (and could potentially be lost from the system), whereas *F. pratensis* would become more abundant. As all these species provide quality fodder, it is conceivable that fodder provision could remain uninterrupted. Nonetheless, such seemingly innocuous loss of diversity may in fact be detrimental for functions and services not considered, or have delayed consequences if response diversity is also lost. The loss of response diversity may reduce the resilience of the system to environmental change (Laliberté *et al.* 2010).

Maintaining local populations of species of conservation concern is a key objective of conservation policies. The significant reduction of species richness, diversity, and relative abundance of species of conservation concern under a grazing regime affects the ability to meet these objectives. This can represent a conservation challenge as maintaining grasslands as pastures is a culturally vital aspect of traditional farming practices in the Carpathian Mountains. However, the diversity of species of conservation concern can conceivably be maintained across the wider landscape under a mixed use land cover that includes extensively mowed and temporarily abandoned hay meadows, even though populations are likely to decline or disappear at the local scale when subject to intensive grazing.

Abandonment-induced change in land cover can also result in the representation of provider groups not present previously. For example, carbon sequestration may be better represented by the woody species appearing in the successional process. Thus, the loss of a grassland system to an alternative system represents a shift in, but not necessarily a cessation of, services. The desirability of this shift will be determined by the local beneficiaries and the wider community (as represented by governance agencies and their policies), and certain land use practices favoured or discouraged accordingly.

The greater diversity and abundance of species of conservation concern found at the higher elevation sites is to be expected given the majority of the species in this provider group are high elevation species rarely found in lower meadows. The altered environment conditions in the lower meadows due to historic ploughing between the 1960s–1980s, more frequent mowing and contemporaneous application of manure can explain differences in spatial distribution of species that are widespread in the mountains but rare in the lower meadows (e.g. *Lathyrus transylvanicus* Fritsch. and *Anemone narcissiflora* L.). The historically low management intensity typical in the more remote areas (Kolos) may have contributed to the naturally higher levels of species diversity, as has been observed elsewhere (Gustavsson *et al.* 2007). In contrast, lower elevation meadows (Somlyó) are close to human settlements and have been subjected to a relatively more intensive management regime. Intensive mowing is known to disadvantage the development of dicotyledonous species and increase the cover of dominant grasses (Huhta & Rautio 1998; Hansson & Fogelfors 2000). This causal relationship can explain the increased abundance of quality fodder plants many of which are the dominant grasses at Somlyó (e.g. *F. rubra, A. capillaris*) and a reduced species richness of dicotyledonous species within the honey and nitrogen fixation provider groups within lower elevation hay meadows (e.g. seven *Trifolium* spp. species in Kolos, and only five in Somlyó).

All three management practices result in some degree of multifunctionality, with each of the management regimes supporting natural capital stocks that contribute to the provision of more than one ecosystem service at both sites. However, we reiterate the importance of maintaining not just within-provider group diversity but also species richness generally as insurance against uncertainties. For example, species considered as non-contributing here may in fact facilitate or complement the processes from contributing species, and their loss may have cascading impacts such as further compositional and functional shifts over time (Laliberté & Tylianakis 2012). The complexity of species interactions and uncertainties as to which species are less expendable than others provide justification for taking a precautionary approach (maintaining species diversity).

Our study shows that diversity within provider groups is influenced by both management practice and landscape heterogeneity. While we did not test for all factors that might contribute to observed differences between the two study sites, we note that confounding complexities will always accompany dynamic worked landscapes. That we did observe effects on richness, diversity, evenness, and abundance within provider groups across a heterogeneous landscape illustrates our provider group approach is a useful method for assessing the impacts of management practices. We suggest that a mixed-management regime will be required to sustain multifunctional landscapes. What constitutes a mixed-management regime at either local or landscape scales will be driven by desired outcomes as defined by stakeholders, decision makers, and existing policies and agreements (e.g. for protection of mountain hay meadows under Natura 2000, both extensive hay meadows and local pastures are key landscape management features). The biophysical response to management practice we focus on here is just one component contributing to the provision of ecosystem services. Understanding the biophysical realm of ecosystem service provision is crucial for the application of land management aimed to influence the provision of ecosystem services, but also needs to be incorporated with social concerns to more effectively manipulate both the supply of ecosystem services and the flow to beneficiaries. Structured decision making (Possingham *et al.* 2001; Gregory *et al.* 2012) can assist in integrating the biophysical and social components of ecosystem service provision and in doing so identify the best agricultural and nature conservation policies and incentivised mechanisms to implement policies in order to support delivery of required management regimes. Structured decision making involves the defining of objectives, consideration of the management options and estimated consequences of these and alternative options, and evaluating trade-offs. Where provision of specific ecosystem services are included in the objective setting, our provider group approach can assist in describing the system and understanding consequences of management actions.

Conclusions

The provider group approach is a useful and pragmatic approach for assessing the impact of management on the provision of ecosystem services. The provider group approach explicitly links attributes of natural capital stocks with the supply of required ecosystem functions and services and allows for the impacts of management practices to be assessed in this context rather than on diversity *per se*. As the provider group approach is independent of species identity, and thus generalisable, it can be applied at both local and landscape scales to inform decision making on the provision of ecosystem services. The provider group can also easily and rapidly be applied *post hoc* to any dataset that includes species diversity and abundance data, collected from any local system, under any land use or management regime with the addition of data on species contributing to different ecosystem services. Analysis can also easily be repeated over time, using alternative provider groups and/or under a different set of policy objectives. This wide application and ease of use makes the provider group approach a readily implementable way to compare the influence of different management regimes on the provision of ecosystem services.

CHAPTER FOUR:

FARMER PERSPECTIVES OF THE ON-FARM AND OFF-FARM PROS AND CONS OF PLANTED MULTIFUNCTIONAL RIPARIAN MARGINS

This chapter focuses on the critical social component of embedding ecosystem services thinking into natural resource management policy and practice. Using the reintroduction of riparian vegetation natural capital stocks into a dairy landscape as a case study, Chapter 4 further illustrates the conceptual framework presented in Chapter 2 by showing that management actions targeted at sustaining ecosystem services can also influence the benefits perceived and/or experienced by farmers as a consequence of implementing those actions. Here I also place social perspectives of planted riparian margins in the context of wider land use pressures currently challenging the attainment of multifunctional landscapes.

This study adheres to the Guidelines of the ethical review process of The University of Queensland and the National Statement on Ethical Conduct in Human Research. Approval number: 2015000319.

A version of this chapter is under review with Land Use Policy as:

Maseyk FJF, Dominati EJ, White T, Mackay AD. Farmer perspectives of the on-farm and off-farm pros and cons of planted multifunctional riparian margins.

Abstract

The planting of riparian margins is a policy option for pastoral farmers in response to land use induced environmental issues such as declining water quality, stream bank erosion, and loss of aquatic and terrestrial habitat. At its most simplistic, the planting of riparian margins can be seen as an exchange of productive land (a private cost) for the common good objectives of enhanced environmental conditions (a public benefit). We postulate that this obscures the values and ecosystem services that riparian margin plantings provide both on-farm and off-farm. We tested this theory by eliciting the views and experiences from two sets of dairy farmers from Taranaki, New Zealand: those who are or have planted riparian margins, and those who have not yet done so. The views of these farmers confirms that perceptions and experiences vary. Those farmers who have planted riparian margins identified 21 positive aspects of riparian margin plantings that included production, environmental, and social values, and 11 negative aspects of riparian margin plantings. This group of farmers considered that planting riparian margins contributed to achieving policy objectives for water quality. Additional perceived benefits identified by this group include increased biodiversity, the provision of cultural ecosystem services, immediate direct benefits to farm management and the farm system, and in some instances increased productivity on-farm. In contrast, those farmers that had fenced but not planted their riparian margins did not consider that riparian margin plantings could add further benefits to that which could be achieved by excluding stock from waterways, and associated only negative perceptions with riparian margin plantings. In particular, this group of farmers felt that policy objectives for protecting water quality could not be achieved by planting riparian margins alone. We suggest that all the costs and liabilities of implementing and maintaining margin plantings need to be considered in the broader context of the full range of benefits that can be generated from planted riparian margins. However, planting riparian margins is not a panacea for all land management issues. It is not cost neutral, and will not deliver anticipated environmental benefits in every situation. We argue that riparian margin plantings are an important ecological infrastructure investment that need to be captured within a wider policy framework, the benefits of which extend beyond the mitigation of a single negative externality generated by land use practices, such as nutrient loss, and contribute to a multifunctional landscape.

Introduction

Conversion of forested landscapes to provide for the development of agriculture has occurred throughout the world (Tanentzap *et al.* 2015). While this whole-scale transformation of landscapes has increased food production, it has come at a cost to system functions, many of which underpin the provision of other ecosystem services which food and water security and human health are also reliant upon (Gordon *et al.* 2010; Bommarco *et al.* 2013; Costanza *et al.* 2014). Spatial separation of land used for food production, from land used for other ecosystem services including biodiversity protection (i.e. land sparing (Fischer *et al.* 2008)) has reduced social-ecological flexibility of agricultural landscapes by favouring food production in most cases at the cost of all other functions (Meadows *et al.* 2008). Emphasising productivist notions of land use restricts the transition to multifunctional landscapes (Wilson 2008).

In agricultural landscapes, land management interventions aimed at improving diversity are increasingly being regulated or otherwise incentivised to mitigate the environmental impacts of agricultural practices and facilitate transitions to greater 'multifunctional agriculture' (Wilson 2009). An example of an intervention is using riparian zones to separate agricultural practice from waterways. Riparian zones (herein riparian margins) are the margin of land adjacent to waterways where direct interaction between terrestrial and aquatic ecosystems occurs. Riparian margin habitat is not found anywhere other than the riparian zone and has a disproportional influence on ecosystem function relative to the size of the catchment (Collier *et al.* 1995).

Functioning riparian margins are the source of ecological processes such as filtering the flow of nutrients and provision of organic input into aquatic food webs (Bennett *et al.* 2014). Utilising riparian margins as production land heavily compromises their ecological functionality, and removes the ability to spatially separate the detrimental impacts of land use from the receiving environment. The exclusion of livestock from riparian margins and waterways can have immediate environmental benefits (Parkyn *et al.* 2003) by protecting banks from erosion and waterways from the direct input of nutrients and bacteria. Retired, grassed riparian margins of an adequate width for local soil and slope variables also provide a buffer to the input of sediments, nutrients, pathogens, and pesticides transported by overland flow into waterways, reducing contaminant and sediment loadings in-stream (Collier *et al.* 1995). While retired single-tier grassed margins create beneficial buffers, diverse, multi-tiered riparian margin vegetation builds on and enhances the benefits provided by grassed margins increasing both riparian margin functionality and in-stream values (DairyNZ 2012). Multi-tiered riparian margins additionally buffer flood flows and reduce their effect in-stream, maintain a microclimate, increase terrestrial and in-stream habitat, structural complexity, and biodiversity, increase terrestrial carbon inputs into the aquatic system, maintain food webs, and provide shade which maintains lower summer maximum in-stream temperatures and prevents nuisance plant growth (Collier *et al.* 1995; Moller *et al.* 2008).

Management of riparian margins is considered to provide a public benefit (Cooper *et al.* 2009; Buckley *et al.* 2012) and is increasingly becoming embedded in policy and industry standards internationally, including in Europe under The European Union Nitrates and Water Framework Directives; in Ireland under the Agricultural Environmental Options Scheme; and in New Zealand under the Sustainable Dairying: Water Accord. Beyond the public benefits generated by riparian margins there is evidence to suggest planted riparian margins also provide a wide range of ecosystem services directly useful on-farm (a private benefit). The ability for incentives to effect change depends in part on the strength of the incentive farmers require to adopt a new practice (Pannell 2004). Recognising that integrating riparian margins into the farm system can self-generate incentive through the provision of private as well as public benefits is therefore critically important for developing policy or industry practice change incentives.

Programmes to reinstate lost vegetation are driving landscape transformation and manipulation of system function. We were principally interested in benefits and values that farmers perceive or experience to be associated with riparian margin plantings on their farms, and how these values are linked to farmer willingness and motivations to plant riparian margins or not. To better understand these values, we invited dairy farmers from Taranaki, New Zealand to participate in half-day workshops to explore their perspectives on the pros, cons, benefits, values, and liabilities arising from the reinstatement of woody vegetation within riparian margins. In particular we aimed to answer the following three questions:

1. What values, benefits, costs, constraints, and liabilities (pros and cons) do farmers perceive to be associated with the planting of riparian margins?
2. What do farmers see as the influence of planted riparian margins on the operation of the farm and its biological and financial performance?
3. How do identified values influence farmers' motivations for planting riparian margins and are there additional motivational factors?

Knowledge of the private-public benefits experienced by farmers can assist in refining current or developing future policy-driven land management interventions.

Methods

Riparian margin management in New Zealand

The reintroduction of vegetation (natural capital stocks) is a necessary component of replacing lost biological and structural diversity across large areas of New Zealand as historic and contemporary agricultural practices have led to substantial loss of native vegetation (Ewers *et al.* 2006; Walker *et al.* 2006; Lee *et al.* 2008; Myers *et al.* 2013). Native landscapes in lowland New Zealand have been almost completely replaced with systems dominated by exotic species introduced from the Northern Hemisphere by European settlers from the early-mid 1800s. While exotic dominated systems can deliver most functions and services necessary for food production, this shift has come at a cost to the provision of other ecosystem services. Intensification of farming practices over recent decades has accelerated the shift towards single-use landscapes where food provision is favoured over other services.

There is currently no overarching regulatory obligation or subsidised incentive scheme to compel or encourage New Zealand farmers to exclude riparian margins from the productive areas of their farm systems (Tanentzap *et al.* 2015). The statutory responsibility for controlling land use sits at the local government level administered by regional councils. Local government driven riparian margin management in New Zealand typically involves the retirement of the margin from the farm system, or 'set-back' requirements for several land use activities involving discharges into the environment such as the application to land of herbicides, pesticides, fertilisers, or effluent. Retirement of margins is typically focused on dairy systems, horticulture, and commercial forestry while set-back restrictions for discharges can also apply to other farm systems (e.g. sheep and beef). The width of a retired riparian margin varies greatly between regions and between farms and is often a farmer-negotiated distance that can be as narrow as < 1 m, and is often determined independent of the influence of adjacent slope characteristics. Under some policies or programmes, the management of riparian margins may also include planting native riparian vegetation, and it is this activity that our study focuses on. Local authorities (regional and territorial councils) also have responsibilities for the protection and maintenance of existing remnant native vegetation on-farm, including riparian margin vegetation in some cases. However, these approaches are highly variable (Maseyk & Gerbeaux 2015) and there remains no national policy to retain or increase native vegetation (Welsch *et al.* 2014).

The industry-led initiative, 'Dairying and Clean Streams Accord' (Clean Streams Accord) was signed by Fonterra (New Zealand's largest dairy cooperative), the Ministry for Agriculture and Forestry, the Ministry for the Environment, and Local Government New Zealand in 2003. The Clean Streams Accord operated at a national level to address the environmental impacts of dairy farming on waterways and included targets for stock exclusion, and effluent and nutrient management. The Clean Streams Accord was replaced by the 'Sustainable Dairying: Water Accord' (the Water Accord) in 2012. While sitting outside of legislative requirements, compliance with the Water Accord is mandatory as an industry condition of supply.

States of riparian margins

We conceptualise three typical states of riparian margins: *1. Farmed*, margins are utilised for farm productivity (e.g. cropping or grazing livestock to the waters edge); *2. Retired*, productivity is separated from the riparian zone leaving a single-tier ungrazed grass strip; and *3. Retired and vegetated*, multi-tiered riparian margin habitat including a diversity of plant forms is established and maintained. Relative functionality (environmental, productive, and social) increases from state 1 to state 3, although complete restoration of ecological riparian function is uncertain (Parkyn *et al.* 2003; Stockan *et al.* 2012), is a long-term prospect (Collier *et al.* 1995; Stockan *et al.* 2012), and is influenced by the spatial arrangement and scale of planted reaches (Parkyn *et al.* 2003).

Study site

The study was based within the 723,610 ha volcanic ring plain of Mt Taranaki in the Taranaki Region, west coast of the North Island, New Zealand. Following European settlement in the mid-1800s, the once forested landscape was rapidly and almost entirely developed into a largely homogenous pastoral landscape with small, fragmented, isolated remnants of native wetland, scrub, and forest. This transformation is similar to that experienced elsewhere in New Zealand and globally (Welsch *et al.* 2014). Native biodiversity on the Taranaki ring plain has been reduced to less than 10% of former cover and continues to decline (Lee *et al.* 2008; TRC 2008, 2014). This historic and contemporary loss of diversity from the ring plain has caused the irreversible loss of many of the native biodiversity elements and associated ecosystem services that would have been provided by a more diverse landscape. Pastoral farming operations on the ring plain are of an intensity that effectively prevents unassisted reestablishment of lost biota. Over 300 short reach (average length of 20 km), high gradient waterways radiate from Mt Taranaki flowing rapidly and steadily into the Tasman Sea. This extensive network of waterways has a total length of 7,330 km

(14,660 km of stream bank), with a total of 6,517 km (13,034 km of stream bank) on the ring plain (TRC 2011).

The Taranaki Riparian Margin Management Programme

In 1993 Taranaki Regional Council (TRC) initiated the 'Taranaki Riparian Management Programme' (Taranaki programme), a voluntary regionally-focussed riparian margin planting programme targeted at dairy systems on the ring plain, with a key objective to 'protect the water quality in Taranaki'. The Water Accord also applies to dairy farmers in the Taranaki Region, thus there are two major riparian margin programmes co-existing in Taranaki. While both are non-statutory, there are several differences in their design. Of most relevance here are: 1) the reach of the Taranaki programme is greater than the Water Accord with all streams (down to 1st order streams) and all riverine wetlands captured by the programme, while the Water Accord applies only to waterways greater than one metre in width and deeper than 30 cm and only 'regionally significant' wetlands; 2) the Taranaki programme is focused on fencing and planting while the Water Accord focuses on stock exclusion via fencing; 3) the Water Accord includes actions beyond riparian margin management (e.g. management of nutrient loss, and implementation of nutrient use efficiency) that is not part of the Taranaki programme; 4) under the Water Accord, dairy companies have committed to develop support tools (such as guidelines), while under the Taranaki programme, the regional council prepares a riparian planting plan at no cost to the farmer, facilitates supply of plants, and provides plants at wholesale costs. Critically, the Water Accord carries with it the threat of penalty as under the condition of supply agreement, dairy companies can cease to collect milk should farmers not comply with targets.

Of the total 13,034 km of stream bank on the ring plain, 11,093 km has been fenced (85%) and 2,138 km (16%) planted under the Taranaki programme. Combined with existing vegetation, the new plantings bring the total combined length of vegetated riparian margins across the ring plain to 6,874 km (53%).

Farmer workshops

Selection of participants

Our study group comprised dairy farmers farming on the Taranaki ring plain (from a total population of ~1760 dairy farmers in the Taranaki Region). Eligibility for participation was simple:

1. A dairy farmer farming on the Taranaki ring plain that had planted or was in the process of planting riparian margin vegetation (Group A), or
2. A dairy farmer farming on the Taranaki ring plain that had not planted riparian margin vegetation (Group B).

Beyond these criteria, no preference was made based on any other characteristic. The two farmer groups participated in the study via two workshops (one for each group) held in Stratford, Taranaki Region in May 2015.

A total of twenty-two farmers attended the workshops, 17 in Group A and five in Group B. One rural professional (who had previously been dairy farming) also participated with Group A (bringing Group A participants to 18).

Workshop design

We were interested in farmers' perspectives on the environmental, social, and production values provided by riparian margin management and their motivations for planting riparian margins. Workshop activities were designed to answer the research questions.

Facilitated discussions followed a mixed-method approach including semi-structured break-out group and whole-group discussions and feedback, and structured voting methodologies (open and blind) to elicit responses from participants. The definition of riparian margins was presented during the introduction of both workshops to establish consistent context amongst participants and between workshops. Discussions were prefaced with a brief presentation and parameterisation of the topic for discussion and their duration time-restricted (between ten minutes and half an hour), but not obstructed otherwise. Before breaking into discussion groups, participants were asked to reflect on the topics raised in the previous discussion. In this way, each subsequent discussion advanced the prior and allowed for further detail to emerge.

Workshop with Group A

Following an initial discussion and feedback session participants were each allocated three votes which they used to indicate three aspects of riparian margin plantings from the list generated during the group discussions that they felt best captured what was most relevant to them. Voting was open and conducted as a group exercise. This exercise produced the 'top ten' responses that were of most relevance to the group as a whole. Following the workshops, the responses were evaluated to identify thematic similarities and retrospectively grouped into categories (Table 4.1). Each comment was assigned to one of nine categories. Negative comments were taken to represent 'cons' and positive comments to represent 'pros' within each category.

Table 4.1: The nine categories summarising farmer perceived pros and cons of planted riparian margins.

Category		Themes captured
A. Environmental		
1	Environmental	Environmental responsibility; environmental enhancement
B. Social		
2	Social responsibility	Personal values; relationships; perceptions of the non-farming community
3	Equity	Distribution of costs and benefits between farmers and non-farming community; differing requirements between farms
4	Community-scale benefits	Benefits to local economy; benefits to local community
5	Aesthetics	Visual amenity
C. Financial		
6	Financial	Farm productivity; direct on-farm costs and savings
D. On-farm management		
7	Practical considerations	Practicalities of establishing and maintaining riparian margin plantings; interaction with wider farm management
8	Welfare	Safety of personal; safety and welfare of stock
E. Taranaki programme specific		
9	Programme design	Programme requirements; priorities; programme implementation; supporting policies; messaging

To further gauge variance in opinion between participants, the language of the top ten responses from the first discussion (pros and cons of riparian margins) was refined to remove potential ambiguity and transposed into statements (the focal statements) (Table 4.2). Participants were then asked to indicate their level of agreement with each of the ten focal statements using a five-point fixed Likert scale (5, Strongly agree, 4, Agree, 3, Neutral, 2, Disagree, 1, Strongly disagree). This exercise was conducted 'blind', using interactive Turning Technologies software (TurningPoint version 5.3.1) and hand-held voting clickers (Turning Technologies, ResponseCard RF LCD). This method allowed the votes to be confidential, addressing any potential peer pressure and maintaining independence from the group dynamic.

Table 4.2: The ten focal statements describing aspects of planted riparian margins. The statements were transposed from the top ten aspects of riparian margin plantings of most relevance to Group A (voted by Group A from a group-generated list of 32 pros and cons). The category (Table 4.1) that each statement falls within is shown in the fourth column.

Rank	Theme	Statement	Category
1	Biodiversity	Riparian margin plantings increase biodiversity value	1
2	Stock management	Riparian margin plantings assist in stock management (avoided losses)	6
3	Pasture management	Riparian fencing helps with pasture management	7
4	Shade & shelter	Riparian plantings provide multiple functions (shade and shelter for stock)	8
5	Water quality	Riparian plantings have benefits for improving water quality	1
6	Council-farmer relationship	The riparian planting programme has fostered good relationships between farmers and regional council	2
7	Costs	Different farms are subjected to different costs with riparian plantings	3
8	Farm management	Riparian plantings make you think about management in a broader way	7
9	Weeds & pests	There are ongoing weed and pest maintenance costs from riparian plantings	7
10	Public perceptions	Riparian plantings improve public perceptions of dairy farming	2

Finally, participants were presented with a questionnaire investigating why farmers had planted riparian margins and how they felt about them. The questionnaire proposed 26 fixed statements of which the respondent could tick as many as they agreed with. Respondents were also given the option for a ‘don’t know’ response or to provide their own statement(s). Sixteen participants completed the questionnaire. Responses to the questionnaire were summarised with descriptive statistics.

Workshop with Group B

The Group B workshop followed the same format as that for Group A. Following their own discussion, response feedback, and preference voting on the pros and cons of planting riparian margins, Group B were presented with the focal statements generated during the workshop with Group A. Group B participants were then asked to indicate their level of agreement with each of the ten focal statements using the same five-point fixed Likert scale (Strongly agree, Agree, Neutral, Disagree, Strongly disagree) and blind polling methodology. Comparison between the two groups of the level of agreement with the ten focal statements was conducted using Pearson’s t-tests conducted using R Studio version 0.98.1091 (R Core Team 2014).

Online surveys

To gauge how representative the responses generated by both workshop groups was of the wider Taranaki ring plain dairy farming community, the views of the TRC Land Management Officers

(LMOs) were elicited via an anonymous online survey. The LMOs are directly involved on a day-to-day basis with implementing the Taranaki programme and are regularly engaged in discussions with farmers regards the merits or otherwise of riparian margin plantings. The LMOs were asked to indicate how frequently (All the time, Frequently, Infrequently, Never) they heard each of the ten focal statements derived from responses generated by Group A and all of the responses generated by the Group B (n = 15). Respondents were also given the option for a ‘don’t know’ response or to provide their own comment(s). The online survey was emailed to seven potential participants (four current LMOs and three who were recently but are no longer working as a LMO with the TRC). Four responses were received. While the sample size of the LMO survey is small, each LMO interacts with a large number of farmers on a regular basis. Responses to the online survey were summarised with descriptive statistics and evaluated alongside the outcomes of the farmer workshops.

Results

There was a notable difference in how each group perceived riparian margins. Group A always assumed plantings to be present when they conceptualised riparian margins, while Group B explicitly differentiated between fenced, single-tier grass strip riparian margins and planted or multi-tier riparian margins and considered the difference to be critical in their assessment of the potential for riparian margins to generate benefits.

Group A identified 32 pros and cons of planted riparian vegetation while Group B identified 15. Group A’s list covered a broader range of aspects that could be aggregated into nine categories (Table 4.1), while Group B’s list only populated four of the same categories. Group A’s list predominantly identified positive aspects (pros, 65%) while Group B’s list contained only negative aspects of riparian margin plantings (Figure 4.1). The notable disparity between the two groups reflects the general consensus of Group A that riparian margin plantings provide benefits beyond just the protection of water quality values (“*Riparian plantings make you think about management in a broader way”*) while Group B struggled to identify benefits additional to those achieved by excluding livestock from waterways, openly questioning the ability of plantings to protect water quality (“*Goals are unrealistic and unattainable*”).

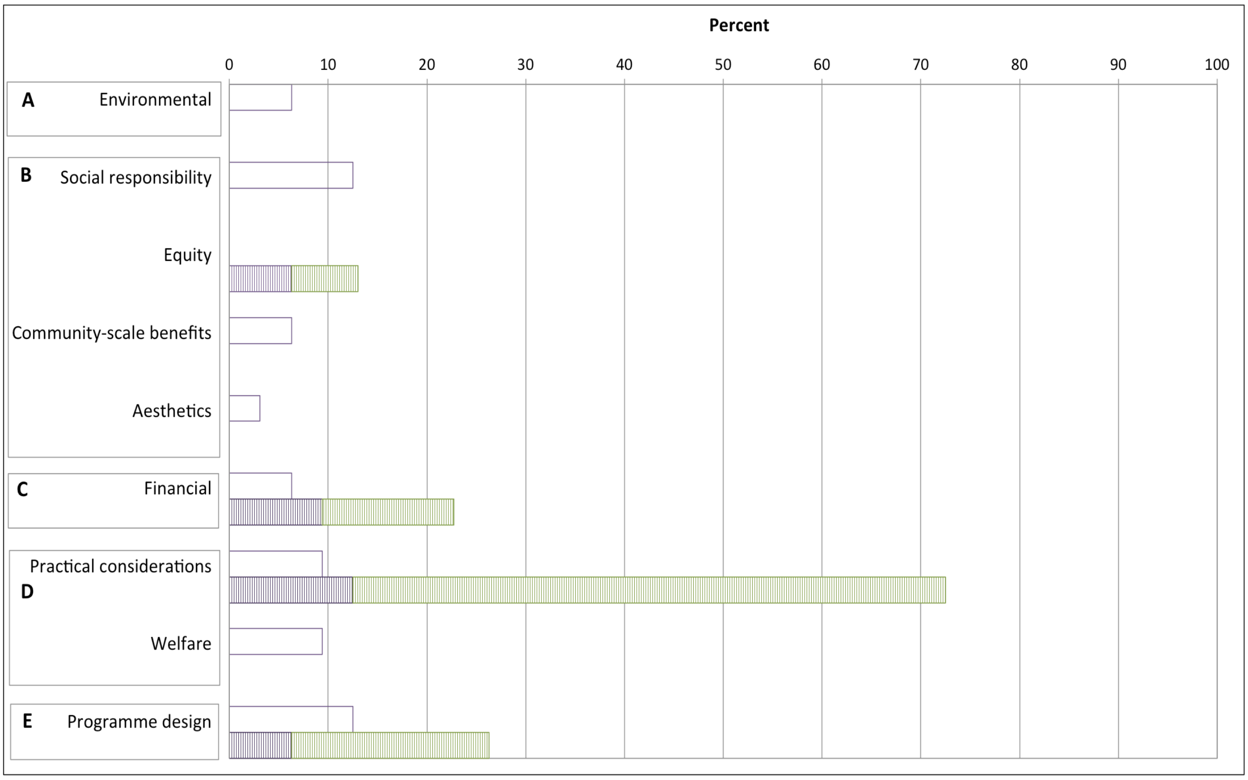


Figure 4.1: Aspects of planted riparian margins by category showing the percentage of pros (upper hollow bars) and cons (lower striped bars) generated by farmers. Categories are grouped by type: A = Environmental; B = Social; C = Financial; D = On-farm management; E = Taranaki programme specific. Group A (Planters) identified a total of 21 positive aspects (hollow purple bars); and 11 negative aspects (purple vertical striped bars); Group B (Non-planters) identified zero positive aspects and 15 negative aspects (green vertical striped bars).

The ten focal statements derived from responses generated by Group A are presented in Table 4.2. The farmers in Group A (n = 17) took part in the Likert voting on the focal statements, returning a total of 168 (of the 170 potential) responses (only 16 participants responded to two of the statements). Group B returned a total of 50 responses (all five participants responded to each of the ten statements).

Group A participants showed a tendency to Strongly agree (71, 42%) or Agree (77, 46%) with the group generated statements although some individuals were Neutral (18, 10%). Disagreement with the statements by Group A participants was minor with only two (1%) Strongly disagree responses. Group B participants' responses were more evenly spread across the levels of agreement (Strongly agree, 11 (22%); Agree, 8 (16%); Disagree, 9 (18%); Strongly disagree 9 (18%)) with the highest proportion of responses falling into the Neutral category (13, 26%). However, a significant difference in level of agreement between the two groups was detected for five of the ten focal statements (Figure 4.2).

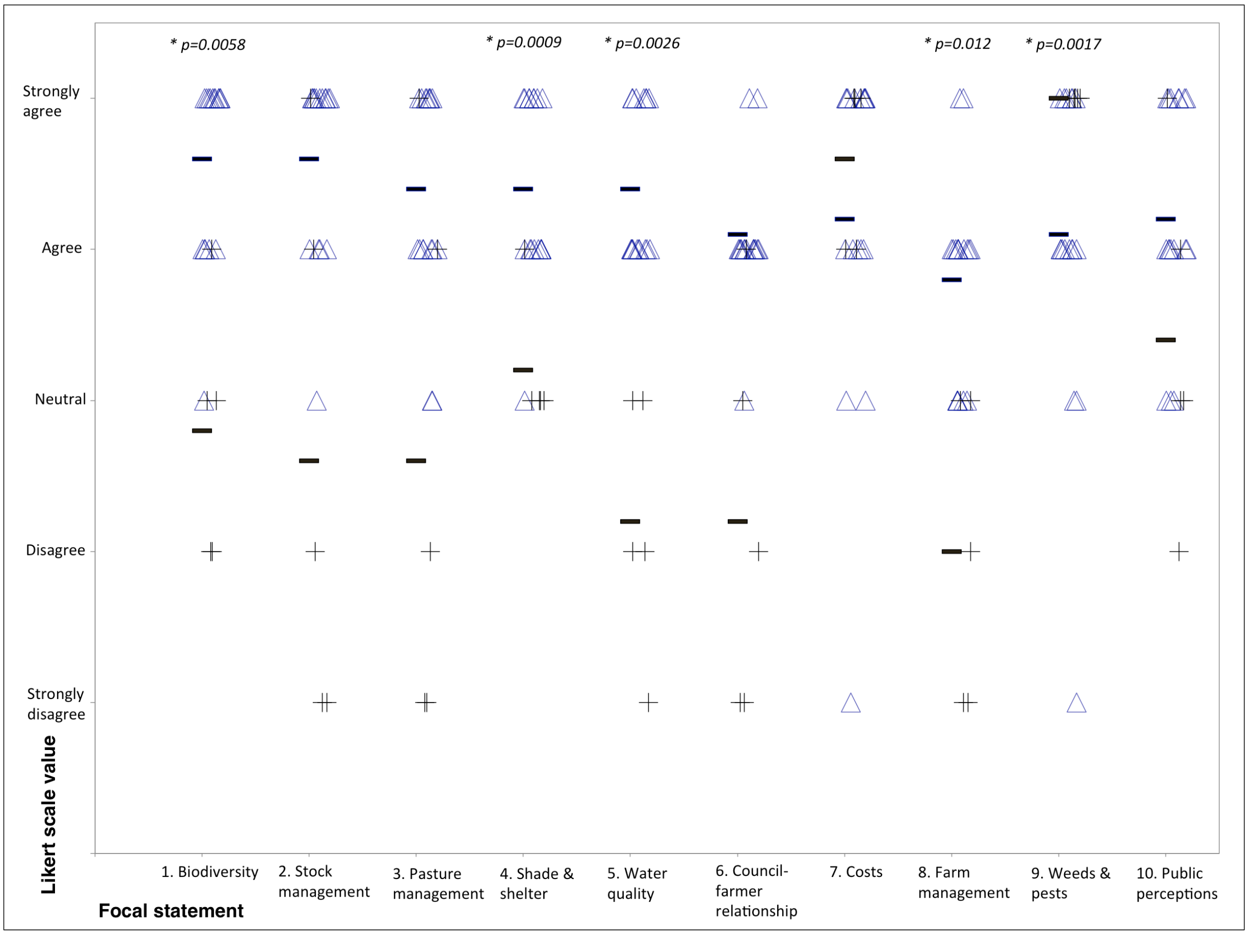


Figure 4.2: Farmer agreement with the ten focal statements generated by Group A describing aspects of riparian margin plantings. Level of agreement was measured using a five-option Likrt scale (y-axis). Blue triangles denote Group A responses, mean values indicated by blue horizontal bars. Grey crosses denote Group B responses, mean values indicated by grey horizontal bars. A significant difference in level of agreement (95% confidence level) between the two groups was detected for five statements (1, 4, 5, 8, and 9) as indicated by an asterisk (*).

Farmer perceived pros and cons of planted riparian margins (Q1)

The range of pros and cons identified by Group A include environmental, social, and production values (Figure 4.3). Productivist benefits included gains in the management of livestock (including animal welfare), pasture growth, water quality and supply, and reduced labour costs. Long-term management of plantings was identified as a liability, and loss of production land and increased pest and weed control were identified as some of the associated costs. Participants connected riparian margins with environment benefits such as improved water quality (reduced nutrification and reduced sedimentation) and ecological values such as increased terrestrial and aquatic habitat. Social values were also identified as flowing from planted riparian margins including improving the farm appearance, the ability to attract better staff, and increased property values. Several of the pros and cons identified also arise with fenced-only grass-strip riparian margins (Figure 4.3).

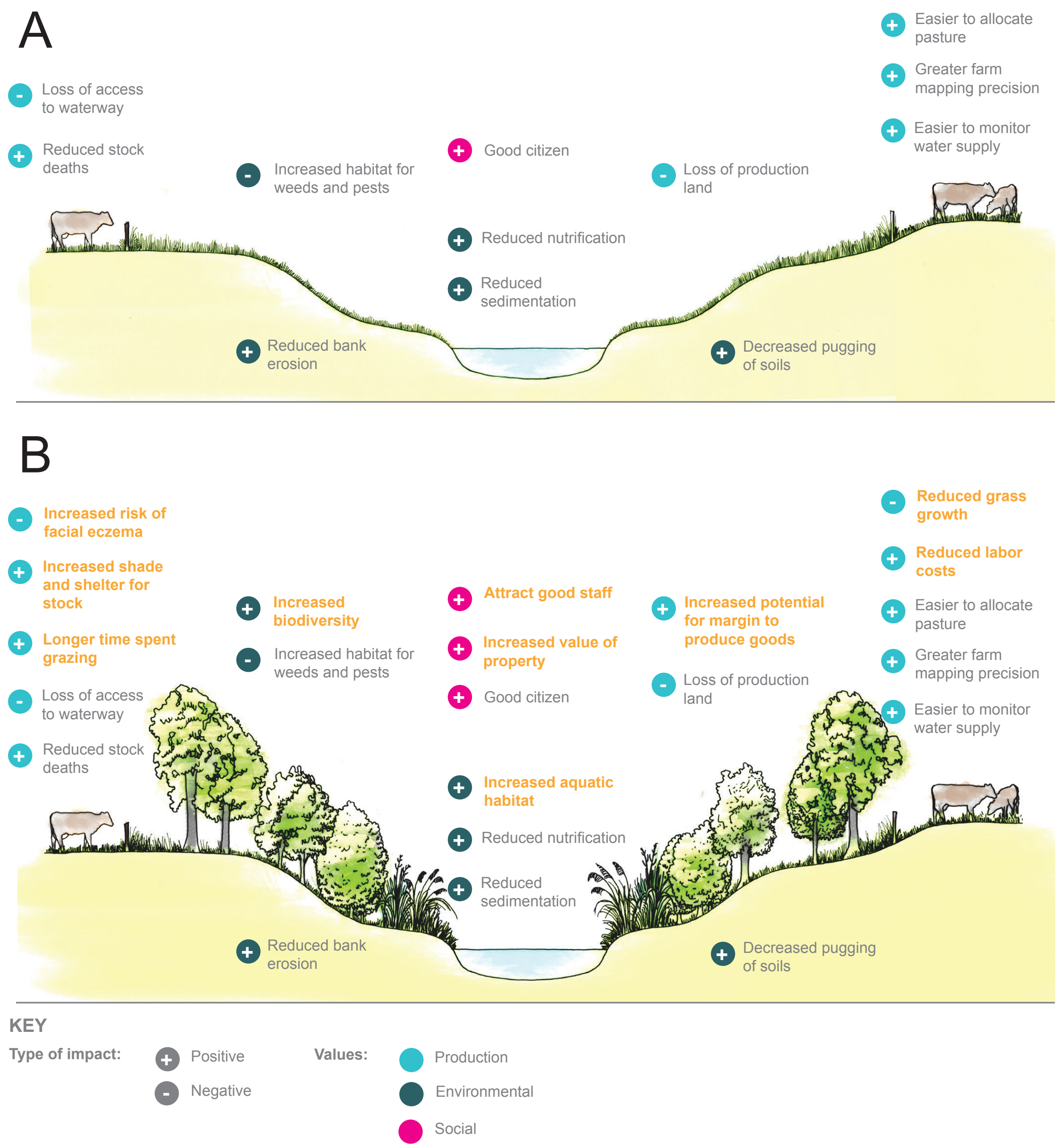


Figure 4.3: Schematic of pros and cons of retired single-tier grass strip riparian margins (A) and retired multi-tier planted riparian margins (B) as identified by dairy farmers farming the Taranaki ring plain (n = 23). *Stock exclusion from waterways* avoided livestock deaths of 2% of the herd/year, saving $2,000/cow; $50/calf and 2 hr labour/day. Fencing costs $5/m ($1–$15/m depending on fence type and topology, authors figures). *Reduction of production land* can range between 5–15% and leads to the reduction of stock numbers. The loss of grass due to retirement of riparian margins may require the use of supplementary feed. *Greater farm mapping precision* allows for better allocation of feed and greater paddock selection for rotational grazing. Better utilisation of grass can deliver the same or better profit and if grazing riparian margins. *Better management water supply* avoids stock illness and death due to liver fluke ($2,000/cow). *Planting of riparian margins* costs $5/m (or ranges from $10–$30/m for 5 m wide multi-tiered planting, authors figures) and requires ongoing maintenance. *Labour costs* are reduced by not having to care for sick animals or dispose of dead animals (2 hr/ day). Permanent fencing removes the need to erect temporary fences around drains and waterways during winter (1–2 hr/day). *Value of property* is increased as a fully fenced and planted farm is more attractive to buyers as they save on fencing and planting costs. Planted riparian margins have the potential to produce goods (e.g. firewood, stock fodder, crop trees) but are required to be greater than 20 m wide to gain carbon credits. Individual attributes of riparian margins can contribute values across the spectrum, for example, a well managed farm attracts better staff which is shown here as a social value, but also ultimately contributes to great productivity of the farm (a production value). All figures and trade-offs supplied by participant farmers from Group A or supplemented by the authors as indicated. Dollars are given in New Zealand dollars (NZ$100 = USD$75 on 14 May 2015).

The enhancement of biodiversity and environmental values was perceived as an important benefit of planting riparian margins for Group A participants (focal statement 1: *"Riparian margin plantings increase biodiversity values"* and focal statement 5: *"Riparian plantings have benefits for improving water quality"*). Individually, 94% of the 17 Group A participants who responded indicated they either Strongly agreed (65%) or Agreed (29%) with statement 1 (one participant was neutral), and all participants Strongly agreed (41%) or Agreed (59%) with statement 5. Group B's lower level of agreement with the statement *"Riparian margin plantings increase biodiversity values"* is not unanimous but is significantly different (p = 0.0058) to Group A's high level of agreement with the statement. Notably, none of the matters raised by Group B when considering the pros and cons of riparian margin plantings covered biodiversity values. The participants of Group B were, however, more concerned about water quality and felt strongly that while excluding stock from waterways had merit, planting riparian margins did not contribute further to meeting water quality objectives. There was also a sentiment expressed by Group B that Taranaki's water quality was of a high standard and therefore riparian planting programmes were not only ineffective, particularly in comparison to other practices (*"efficiency of margins vs. tiles and drains"*; *"targeting of point sources"*), but also unnecessary (*"…retain water quality that is already well above the world standard"*).

Group A identified improved relationships with the council as a positive outcome of planting riparian margins under the Taranaki programme specifically (focal statement 6: "*The riparian planting programme has fostered good relationships between farmers and regional councils*"). Twelve percent of Group A participants Strongly agreed with this statement, while 81% Agreed, and 6% were neutral, although the group was explicit this was only in the context of TRC and not other regional councils in New Zealand. Group B did not share the same enthusiasm for improved relationships between farmers and TRC, with 40% Strongly disagreeing with this statement, 20% Disagreeing, 20% neutral, and 20% Agreeing (Figure 4.2). Group A also acknowledged the ongoing one-on-one engagement with council staff, advice, and assistance as a benefit of, and a motivation for, engaging in the Taranaki programme to implement riparian plantings.

Both groups felt the costs of riparian margin management fell unfairly on them ("*life-stylers have no responsibility"*), while benefits accrued to the wider community (*"benefit to the wider community at cost of farmer with margins"*; "*cost to famers vs. community benefits*") creating perceived equity issues.

Farmer perceived influence of planted riparian margins on the operation and performance of the farm (Q2)

Group A participants identified a wide range of pros and cons of the influence of riparian margin plantings on farm productivity and management of the farm system (Figure 4.3). While acknowledging that there were cons (and costs) involved with riparian margin planting and long-term management of riparian margins, the Group A participants indicated that these were countered by the benefits that riparian margin plantings contributed to the farm system. Two things were notable in the discussion. First, that many negative aspects could be overcome by best practice implementation and management and were not a consequence of riparian margin plantings *per se* (e.g. facial eczema management or plants causing electric fences to earth). Second, while not all perceived benefits of riparian margin plantings directly influenced productivity, they ultimately did. For example, any labour savings gained by not needing to erect temporary fencing in winter or rescue stock from waterways *"can be used to think about feed"* and the ability to attract better quality staff and the improved condition of the herd due to the provision of shelter results in *"more milk in the vat"*.

The loss of effective productive land was identified as a direct and immediate trade-off of excluding production from riparian margins. However, the majority of Group A stated that the ability to improve pasture management and the condition of the herd compensated for this loss and profits were the same or better. Group B also identified the loss of production land as a cost of riparian margin management placing it second in importance (after water quality goals of riparian margin planting programmes being unrealistic and unattainable). However, Group B didn't identify any benefits to farm productivity that riparian margin plantings could bring to their farm systems (Figure 4.1), thus for these farmers, the loss of production land was an unmitigated cost rather than a trade-off between values.

In regards practical considerations (Figure 4.1), both groups recognised the negative association between riparian margin plantings and increased pest and weed issues on-farm (*"maintenance of weeds and pests, costs, impacts on development"*; *"building a home for pests we are trying to get rid of"*). However, Group A did acknowledge that the extent of weed issues is related to the type and age of the riparian margin plantings. Responses from the LMO survey confirmed that weed and pest management in relation to riparian margin plantings is a pervasive issue for farmers with riparian margin plantings and a perceived issue for farmers without plantings. The surveyed LMOs reported hearing comments to this effect: Frequently (50%), All the time (38%), or Infrequently

(12%), although it was noted that the tolerance for weeds in riparian margin plantings varies between farmers.

Both groups also identified damage to fences, the blockage of drains and culverts, and damage to infrastructure by plants washed downstream in flood events as negative aspects of riparian margin plantings. The surveyed LMOs reported that the damage caused by washed out plants was raised: All the time (25%), Frequently (25%), or Infrequently (50%). However, the LMOs did note that the likelihood of this being an issue was dependent on where in the catchment the plantings were located, and how established the plantings were, with newer plantings being more susceptible to being washed out. The damage caused to fences by riparian margin plantings was raised: Frequently (50%), Infrequently (25%), or Never (25%) with comments of this nature typically referring to the shorting of electric fences. Group B also reflected that impeded drainage by plants blocking drains resulted in further loss of productivity as the land becomes saturated. Riparian margin plantings not only caused blockages but prevented access to remove plant material and sediment from waterways (*"no access to waterway for maintenance"*).

While Group B considered labour costs would increase as a result of these practical problems caused by riparian margin plantings, Group A, while recognising these factors as negative issues, were more likely to acknowledge they could be overcome by improved implementation of riparian margin management (e.g. allowing enough space between fence lines and stream banks to allow plants to grow and planting more high-flow tolerant plants at the waters-edge).

Group A had given thought to the integration of the riparian margins into their farming system (Figure 4.3) and saw only small outstanding cost implications. Ten of the 16 (63%) Group A participants who completed the questionnaire agreed with the following statement *"I think that the on-farm benefits from riparian margins are greater than the cost to maintain them"* (questionnaire results). In contrast, planting riparian margins was seen by Group B not just as an unnecessary step that came at a cost (*"loss of productive land"; "labour costs"*), but an intervention that did not add to the structure or value of the farm.

Farmer motivations for planting riparian margins (Q3)

In responding to the questionnaire, Group A farmers indicated environmental stewardship responsibilities and perceived on-farm benefits as key motivators for planting riparian margins. All Group A participants agreed with the statement *"I planted riparian margins because I believe I* *have a responsibility for environmental protection and enhancement"* and 75% agreed with the statement *"I planted riparian margins because I wanted to improve water quality for future generations"* (questionnaire results). Ten of the 16 (63%) Group A participants agreed with the following statements *"I planted riparian margins because I was confident that they would improve the productive performance of my farm"* (questionnaire results).

Group A farmers were also motivated by the non-farming communities perceptions of dairy farming and recognised that riparian margin plantings, as a highly visible feature, provided tangible evidence of farmers being pro-active thus improving the image of dairy farmers ("*riparian margin plantings improve public perceptions of dairy farming*"). Feedback from LMO survey confirmed this motivation, stating farmers will often prioritise plantings visible from the road in order to be "*seen to be doing something*" (LMO survey).

In regards the Taranaki programme specifically, Group A farmers indicated their preference to participate in the voluntary Taranaki programme over the Water Accord with its associated threat of compliance ("*double message from regional council and Fonterra —incentives carrot vs. sticks*"). The group also expressed their perception that voluntary participation now will avoid being subject to regulation in the future ("*participation in programme keeps regulation away*").

Discussion

The experience of our participant farmers was that retired grass-strip margins provide a range of private and public benefits. The group of farmers who had planted riparian margins perceived the range of benefits flowing from riparian margins to increase due to the addition of riparian vegetation. These benefits fell across the environmental, ecological, social, and production realms. This was in contrast to the group of farmers who had not planted riparian margins, who perceived that retired grass strip riparian margins were adequate to provide the water quality benefits they and the authorities were interested in. Both groups of participant farmers perceived the pros and cons of riparian margin management across a spectrum of scales (paddock, farm, whole catchment) and beneficiaries (self, neighbours, non-farming community). Our results add production and social values to existing environmental and ecological values that have been associated with riparian margin plantings.

Benefits and values farmers perceive to be associated with planted riparian margins

Several of the benefits for productivity and managing the farm system of planted riparian margins as identified by our participant farmers can also be delivered by fenced only riparian margins. Fencing creates infrastructure that allows for improved farm design and feed allocation that improves farm performance. For example, improved farm mapping of infrastructure assists with the allocation and utilisation of forage, enables greater precision around inputs and management, and prevents injury or death of both livestock and farm staff.

Fenced riparian margins are effective in removing animals from waterways, reducing stream bank erosion, and filtering pollutants (phosphorus and pathogens) transported in overland flow to waterways (Collier *et al.* 1995; Parkyn *et al.* 2003). In situations where grassed riparian margins are limited in their capacity to filter nutrients (e.g. such as where the phosphorus retention and buffering capacity of soils is limited (Aye *et al.* 2006), riparian margin plantings can enhance the functionality of the riparian margin through the uptake of plant available phosphorus. However, the inherent capacity of natural capital stocks to filter and retain nutrients is finite and riparian margins are less effective in reducing the losses of nutrients lost by leaching through the soil profile (e.g. nitrogen in porous soils) than in overland flow processes (Muscutt *et al.* 1993; Parkyn 2004; Buckley *et al.* 2012). When combinations of soils, stocking rates, hydrological flows, and farm performance lend themselves to greater nutrient loss, planted riparian margins are less likely to be successful without further management intervention (Quinn *et al.* 2009; Howard-Williams *et al.* 2010; Stockan *et al.* 2012). Therefore, in areas that experience nutrient (particularly nitrate-nitrogen) issues, riparian margin planting programmes should be part of a bigger initiative focused on sustainability and multifunctionality at the catchment scale based on incremental and transitional nutrient management programmes.

The addition of multi-tier planting to retired riparian margins combines natural capital stocks (riparian vegetation) with built capital which, despite the limitations outlined above, were perceived to deliver further benefits for the farm system. These benefits include shade and shelter for livestock and potential reductions in evapotranspiration of pastoral species. Further, the combination from different natural capital stocks within riparian margins influences margin utility for both farm performance and environmental enhancement. However, to achieve benefits as far-reaching as possible riparian margins need to be considered as fully integrated components of the farm system and not as exclusive strips on the farm-edge. Group A participants appear to have made this transition, based on the wider range of impacts and benefits they recognise to be provided by their riparian plantings.

The Taranaki programme explicitly links riparian margin plantings to improved water quality. This suggested causal influence has resonated with our study group, who either believed riparian margin plantings contributed to improved water quality (Group A) or questioned this relationship (Group B). However, it is difficult to untangle the benefits of livestock exclusion from waterways and the stabilisation of stream banks from the capacity of riparian plantings to intercept nutrients and other drivers of change impacting on water quality in the Taranaki Region. For example, currently half of the dairy farms in Taranaki legally discharge pond-treated dairy effluent directly into waterways, although there has been a gradual shift to land-based dairy effluent treatment systems in recent times. Eliminating discharge of treated dairy effluent to water is expected to reduce nitrogen loss by an estimated 20% (TRC 2015b). Concurrently, improvements to municipal wastewater treatment systems have driven measurable improvements for the catchments to which they discharge. The strong conviction of Group A farmers that there is a causal relationship between planted riparian margins and improved water quality may be attributable to messaging regards the benefits of riparian margin plantings, that they conceptualise plantings to occur hand-in-hand with fencing and are thus conflating impacts of stock exclusion from waterways with plantings, or other factors such as pre-existing world-views which we did not test for. We also did not investigate which parameters of water quality (e.g. water clarity, nutrient concentrations, water temperature etc.) farmers perceived riparian margin plantings to enhance.

Both sets of farmers also identified costs and liabilities associated with planting riparian margins. Many of these (e.g. pest management and maintenance of fences) are relevant at the farm scale and can be overcome by improved implementation practices and, as riparian margin plantings become more integrated into the farm system, associated costs are likely to be seen more as a component of the wider farm operation and less of an additional cost. The participant farmers also identified the loss of planted vegetation and infrastructure (culverts and fences) during flood events at the farm scale and consequential downstream damage at the landscape scale as considerable liabilities of fencing and planting riparian margins. These losses are an inevitable outcome of major flood events that would require substantial reinvestment to replace. However, the risk of loss due to severe events is also true for other infrastructure and stock on-farm but this doesn’t prevent such assets from being routinely established or replaced when lost. A greater understanding of the full range of benefits that can be generated from riparian margin plantings could provide justification for the investment required to maintain riparian margins as an integral part of the farm system long-term.

It is evident from our study that planting riparian margins generates a number of perceived benefits across a range of values. However, great care needs to be taken in extrapolating perceived benefits to actual benefits where these lie beyond what margins realistically can deliver. Importantly, re-establishing riparian margins is not a panacea for all the environmental challenges confronting agricultural landscapes. This was a critical point for some of the participant farmers who recognised that the objective of protecting water quality was unachievable by planting riparian margins alone (Group B). These farmers felt connecting water quality objectives with planting riparian margins was 'misguided', and this perception obscured recognition of all other potential values and benefits in planting riparian margins and prevented these farmers from implementing plantings within their riparian margins. Lessons can be learnt for engaging farmers in riparian management programmes elsewhere in the country whereby wider uptake may be achieved through both broadening the objectives for riparian margin planting programmes to more adequately reflect the potential for benefits beyond improved water quality, and recognising the inability of planted riparian margins on their own to fully address the national water quality challenge.

Farmer identified motivations for planting riparian margins

The recent National Policy Statement for Freshwater Management (Ministry for the Environment 2014) has given greater urgency to nutrient management. As a consequence, greater regulatory intervention nationally for nutrient management in some form is on the horizon. Although Taranaki does not experience nutrient loss issues of the same magnitude as elsewhere in New Zealand due to the high, frequent rainfall and short, fast stream flows, combined with good soils and a long history of dairy farming with relatively low stocking rates (TRC, 2015), participant farmers possess a well-tuned awareness of national water quality issues. Despite TRC operating a non-regulated approach to nitrogen management, participant farmers expressed anticipation of increased regulation on-farm to combat declining water quality in agricultural catchments. Taranaki Regional Council were proactive in creating the Taranaki Programme, being the first council in New Zealand to implement a riparian margin planting programme and pre-dating industry-driven programmes to exclude stock from waterways. Under the Taranaki Programme, a large number of riparian plans were developed well before water quality became especially topical in the public conversation nationally. However, in more recent times, the 'threat' of future regulatory action may have increased the appeal of the Taranaki programme. In partaking in the voluntary Taranaki programme, farmers are anticipating impending nationally-led regulation and pre-empting future obligations.

A further benefit of planting riparian margins expressed by Group A participants was improving non-farmer perceptions of dairy farming. Farmers are increasingly conscious of the wider community's perception of them as individuals and of their industry as a whole which has attracted the unenviable label of 'dirty dairying'. The public's perception of farming and reduced tolerance for land use induced environmental degradation is likely to be providing a concurrent motivation for farmers that haven't already done so to voluntarily plant riparian margins on their farms. Fenced and planted riparian margins are highly visible features in the landscape and thus send a tangible message to the non-farming community that good land use practices are being implemented.

The continued trends of native biodiversity decline (Ministry for the Environment & Statistics New Zealand 2015), suggest that enhancing biodiversity values is not perceived to be beneficial enough on-farm to be a sufficient motivator for farmers to engage in riparian margin planting, or perceived to be a beneficial outcome of doing so. Contradicting this assumption, Group A perceived a causal relationship between riparian margin plantings and increased biodiversity values (in particular bees, birds, and frogs). However, the definition of 'biodiversity' (and therefore the associated values) referred to by our study group is not restricted to native biodiversity or that required to achieve conservation objectives. The planted riparian margins are not providing complete ecological equivalents for lost native habitat, and where they differ in species assemblage, are unlikely to be in the future. Thus, while Group A farmers perceive biodiversity benefits to flow from planted riparian margins these benefits are not analogous with biodiversity conservation benefits. Rather, a wider view of biodiversity is reflected relating to, for example, greater structural diversity, benefit of biodiversity for production values, and amenity values associated with diversity in the landscape. This illustrates an encouraging step towards integrating riparian margins into the farm system in its entirety, although conservation challenges are likely to persist.

Both groups of participant farmers expressed a preference for a non-regulatory approach to riparian margin management. In the absence of compulsion, the voluntary motivation to undertake specific actions requires not just an attitude change but also a behaviour change (Rhodes *et al.* 2002). When left entirely to voluntary mechanisms this shift in behaviour (adoption of action) is based on subjective rather than objective decision making and can be very slow (Pannell *et al.* 2006). This can be problematic in situations where riparian margin management is being used in response to urgent resource issues.

A switch in focus from riparian margins sitting outside the farm system to riparian margins being integrated into the farm system would likely expedite their adoption by demonstrably providing a suite of services and benefits. If retired and planted riparian margins can be shown to have a relative advantage over fenced-only or farmed riparian margins the practice will likely become more economically and socially appealing. Land management practices that have a relative advantage over alternative actions are more likely to be adopted (Pannell *et al.* 2006). Further, effective voluntary schemes require effort, commitment and clear expectations from both the implementing agency and land owner. Ultimately, voluntary schemes should be time-bound and replaced with regulation to capture the minority of land owners who choose not to partake voluntarily. When these aspects are in place, uptake is likely to be more rapid and implemented by more landowners. The Taranaki Programme illustrates this well, where sustained support and encouragement from TRC over the past 20 years has resulted in large-scale uptake of the programme (c.99.5% of dairy farmers on the ring plain now have a riparian planting plan in place, TRC *pers com*), while those farmers who have not yet planted their riparian margins will in the future be obliged to do so under proposed rules in the Regional Freshwater Plan (RFWP).

Many studies have recognised the value of riparian margin management for ecological or environmental benefits (e.g. Wilcock *et al.* 2009; McCracken *et al.* 2012; Bennett *et al.* 2014). Other studies take a more productivist view and recognise the values to the farm system that riparian margin habitat can provide, such as provision of habitat for pollinators and fauna beneficial for pest control or potential benefits of native vegetation for animal nutrition (Wratten *et al.* 2012; Hahner *et al.* 2014; Cole *et al.* 2015). Our study brings productivist, ecological, environmental, and social values together, providing a broader foundation of information that is useful for refining future policy. These findings have particular relevance for other regions who have yet to implement riparian margin management programmes. Further quantification and qualification of the raft of values provided on and off-farm by planted riparian margins is required to identify and incentivise riparian margin management that best supports multifunctional farm systems.

Conclusions

Our findings show that farmers with planted margins perceive the introduction of vegetation natural capital stocks into riparian margins to provide many benefits and have started to recognise and value the environmental, production, and social functions of riparian margins in an integrated way. Strengthening multifunctional agriculture is not only positive for the farm system but can resonate with the non-farming community who see this as a preferable model for farming (Wilson 2008).

The reinstatement of native vegetation within riparian margins in highly modified landscapes like the Taranaki ring plain, creates novel ecosystems, the establishment of which can generate social, environmental, biodiversity, and functional benefits. The management of riparian margins is not a panacea for all land management issues and the practice does necessitate trade-offs. We suggest that multi-tiered riparian margins can become an integral part of the farm system and can contribute to multifunctional landscapes. However, the planting of riparian margins needs to sit within a more comprehensive policy framework providing incremental mitigation options if a wider range of negative externalities generated by land use practices are to be reduced.

CHAPTER FIVE:

A DISAGGREGATED BIODIVERSITY OFFSET ACCOUNTING MODEL TO IMPROVE ESTIMATION OF ECOLOGICAL EQUIVALENCY AND NO NET LOSS

The loss of biodiversity results in a loss of ecosystem services critical for human survival and wellbeing. Thus, biodiversity offsetting represents a critical application of ecosystem services thinking to conservation decision making. Offsetting also represents extreme manipulation of natural capital stocks in that biodiversity elements are lost from one place and time in exchange for their replacement elsewhere (in both time and space). Currencies are used to describe what is exchanged in a biodiversity offsets trade and estimate the size of offset required and thus have a substantial influence on the outcomes of biodiversity and associated ecosystem services. Biodiversity offsetting will always result in the permanent loss of some element of biodiversity and ecosystem function. What is critical is that currencies explicitly account for those biodiversity elements that are important — for delivering conservation outcomes or sustaining a wider range of ecosystem services. In this chapter I describe a disaggregated accounting model I developed to improve the estimation of ecological equivalency in biodiversity offset exchanges that allows for just this.

The model described in this chapter was developed under contract with the New Zealand Department of Conservation (DOC). Intellectual Property Rights (IP) of the model and user manual remain in the ownership of DOC. The candidate retains IP rights to content within this thesis. The physical model template comprises a series of Excel spreadsheets (designed and developed in Microsoft ® Excel ® for Mac 2011; Version 14.5.7), and can be (along with the user manual) freely accessed from http://www.doc.govt.nz/about-us/our-policies-and-plans/guidance-on-biodiversity-offsetting/biodiversity-offsets-accounting-system/.

A version of this chapter is under review with Biological Conservation as:

Maseyk FJF, Barea LP, Stephens RTT, Possingham HP, Dutson G, Maron M A disaggregated biodiversity offset accounting model to improve estimation of ecological equivalency and no net loss.

Abstract

Biodiversity offsetting is a mechanism aimed at achieving biodiversity gains to compensate for the residual impacts of development activities on biodiversity. Estimating the ecological equivalence of biodiversity lost to development with that gained by the offset requires a currency that captures the biota of interest and an accounting model to evaluate the exchange. Ecologically robust, and user-friendly decision support tools improve the transparency of biodiversity offsetting and assist in the decision making process. Here we describe a tool developed for the New Zealand Department of Conservation that offers a mechanism to transparently design and evaluate biodiversity offsets intended to deliver no net loss. It is a relatively disaggregated accounting model that balances like-for-like biodiversity trades using a suite of area by condition currencies to calculate net present biodiversity value (NPBV) to account individually for each measured biodiversity element of interest. The NPBV is used to evaluate whether a no net loss exchange is likely for each biodiversity attribute. More disaggregated currencies have an advantage over aggregated currencies (which use composite metrics) in that they account for each itemised biodiversity element of concern. The disaggregated model we present can be used to account for a variety of biodiversity types in an offset exchange, and for different scales and complexities of development and impacts within both statutory and voluntary frameworks.

Introduction

Biodiversity is in decline globally (Butchart *et al.* 2010) and will remain under pressure as the world population and demand for resources increase (Brown 2012). Continued biodiversity losses due to development provide wealth for some while eroding the wellbeing of others (Kumar 2010). Biodiversity offsetting is an evolving mechanism that attempts to mitigate losses and manage associated risks (BBOP (Business and Biodiversity Offsets Programme) 2013). The approach requires development-induced losses in one place and time (the impact site) to be addressed by delivering biodiversity gains at another place and time (the offset site) with the goal of achieving no net loss. The practice of biodiversity offsetting is becoming increasingly popular as a way to compensate for development impacts (Calvet *et al.* 2015; Gonçalves *et al.* 2015; Ives & Bekessy 2015; Rainey *et al.* 2015; Maron *et al.* 2016a).

Biodiversity offsetting is controversial because it has yet to establish a compelling track record of achievement of either implicit or explicit goals (Harper & Quigley 2005; Matthews & Endress 2008; Walker *et al.* 2009; Maron *et al.* 2012; Brown *et al.* 2014). The concept is often used by development advocates to promise 'win-win' outcomes, a claim which attracts scepticism and controversy (Gordon *et al.* 2015). Biodiversity offsetting relies on using techniques with uncertain outcomes (e.g. 'restoration' Hobbs *et al.* 2011) to generate future gains in biodiversity values, assumes there is sound scheme design, and that regulators and developers will honour offsetting agreements on behalf of the public who would bear the costs of any net biodiversity loss. Consequently offsetting is a polarising concept criticised for the risks to biodiversity (e.g. Walker *et al.* 2009; Maron *et al.* 2010; Walker 2010; Spash 2015) but supported for its potential to enhance biodiversity outcomes (e.g. Norton 2007; Norton & Warburton 2014; Holmes *et al.* 2016).

There is general agreement within this wider debate that sound offsetting requires as prerequisites: i) strict adherence to the mitigation hierarchy, whereby an offset arrangement is only applied to residual impacts after all other impacts on biodiversity have been avoided, minimised, and rehabilitated/restored on site and ii) a recognition that some elements of biodiversity are irreplaceable or vulnerable, limiting what can be offset. Key conditions that should also be met include: a) the technical feasibility and success of proposed restoration/management actions have been demonstrated, or uncertainty in the chance of success has been accounted for; b) anticipated gains are demonstrably adequate to compensate for the losses; c) time lags between losses and gains occurring are adequately addressed; d) all additional aspects of uncertainty beyond success of offset action are accounted for, and e) currencies used to describe and account for the biodiversity being traded are transparent and rely on defensibly measurable units (McKenney & Kiesecker 2010; BBOP 2013; Gardner *et al.* 2013; Maron *et al.* 2016a).

We note that these conditions are aspirational because acceptable thresholds of compliance are poorly defined (e.g. what is 'adequate' avoidance?). How to determine that compliance has been achieved, who makes this decision, and who bears the cost of noncompliance remain contentious. Despite this, there remains scope for improving biodiversity offsetting by developing tools and processes that address each of the problematic conditions. Here we present a decision support tool in the form of a disaggregated accounting model (herein the Disaggregated Model) for estimating ecological equivalency, which we suggest improves on more aggregated metrics by explicitly describing and measuring biodiversity elements of interest and thereby providing a more robust and transparent estimation of ecological equivalency demonstrated by offset proposals (condition *e* above). Our Disaggregated Model incorporates aspects of all the key conditions listed above, but its principal advantage is its use of disaggregated currencies. To fully appreciate this advantage, we first turn our attention to the importance of currencies in trading biodiversity and why (dis)aggregation matters.

Central to the concept of biodiversity offsetting is the requirement first to measure, quantify, and express as currencies the biodiversity lost to development and gained via the offset, and second, to balance this exchange to establish whether or not no net loss has been demonstrated. Currencies describe how much of what is exchanged in a biodiversity offset trade and have a substantial influence on the outcomes for biodiversity (Strange *et al.* 2002; Bull *et al.* 2014; Gonçalves *et al.* 2015). Therefore, a currency needs to capture what is important, both ecologically and to society, and should minimise exchanges of biodiversity elements not explicitly accounted for (Salzman & Ruhl 2000).

Currencies can either aggregate measures of biodiversity into a composite unit or individually account for each measured biodiversity element of interest (i.e. more disaggregated currencies). However, it is misleading to perceive a strict dichotomy of aggregated or disaggregated currencies, and the concept is better expressed as a continuum along which specific characteristics are expressed to a greater or lesser degree (Table 5.1). For example, hollows in trees could be counted, or they could be described more finely before being measured. Reviews comparing offset policies and currencies across various jurisdictions which further illustrate the continuum have been well summarised elsewhere (e.g. McKenney & Kiesecker 2010; Bull *et al.* 2014).

Table 5.1: Key characteristics of currencies used to evaluate biodiversity offset proposals related to the degree of aggregation within the currency.

Characteristic	**More aggregated** ⟷	**disaggregated**
Measure of biodiversity elements of concern	Composite or surrogate measure to describe many elements	Many and/or direct measures of all biodiversity elements of interest
Risk of concealed trades	Higher	Lower (occurs only below level of disaggregation)
Ability to substitute biodiversity elements	Higher	Lower (occurs only below level of disaggregation)
Transparency of what is being traded (ability to evaluate offset proposal, and to track performance of offset action)	Less transparent	More transparent
Opportunity for offset market	Wider (easier to find a match of a composite measure of biodiversity)	Narrower (more difficult to find a match across multiple elements of biodiversity, may require multiple offset sites)
Examples	Habitat hectares (Parkes *et al.* 2003); Quality hectares (Temple *et al.* 2012); UK pilot metric (2012)	Disaggregated Model (this paper); Units of Global Distribution (Temple *et al.* 2012); Loss-gain calculator (Gibbons et al. 2015)

All currencies variously aggregate elements of biodiversity and so will result in some level of concealed trade. Concealed trades are exchanges of biodiversity elements that are not explicitly accounted for and which are either offset implicitly or lost in the exchange (e.g. different canopy tree species within the same vegetation type, or genes within species). Therefore, what is critical in designing a currency for biodiversity offsetting is that the elements of biodiversity for which no net loss is the desired outcome are not aggregated in such a way that unintended substitution can occur. The target biota for which no net loss is a specific goal are likely to be determined by a range of factors such as those required to meet conservation objectives, or provide required ecosystem services. For example, if maintaining critical components of a forest habitat is the goal and canopy cover is one of those components, it may be acceptable to aggregate canopy cover of functionally-similar species within a measure to represent canopy cover. This level of aggregation would not be appropriate if the level of interest was individual tree species that contribute to canopy cover, or if species of concern have a strong preference for particular tree species. Likewise, canopy and understory measures should not be aggregated into a single measure if both these things are individually of interest.

More aggregated currencies tend to be favoured in offset scheme designs because they reduce complexity (by virtue of having fewer measures to find an adequate offset for), minimise transaction costs, and support market flexibility by allowing substitution of one element for another within the currency. Thus matches do not need to be precise. If the aggregated currency allows substitution among elements of biodiversity that are individually valuable, then in effect, they allow out-of-kind trades to occur (Gibbons & Lindenmayer 2007; Walker *et al.* 2009; van Teeffelen *et al.* 2014).

The limitations of using more aggregated currencies to evaluate offset exchanges share conceptual similarities to the risks of aggregation within other market based instruments such as Payment for Ecosystem Services (PES) schemes. PES schemes are intended to incentivise sustainable management of biodiversity and ecosystem service provision by trading payments (credits) for observable proxies for ecosystem services (such as actions or outcomes, Jack *et al.* 2008). Aggregation occurs within these markets via 'stacking' (separate payments for multiple ecosystem services derived from the same spatial unit) or 'bundling' (single payment for management action that influences provision of multiple ecosystem services). While bundling has the potential to achieve a wider range of benefits (Wendland *et al.* 2010; e.g. Deal *et al.* 2012) and stacking ecosystem services provides the potential to focus on multiple functions and services, this aggregation can also create accounting challenges and lead to concealed or unequal trades

(Robertson *et al.* 2014). There is a crucial difference between PES schemes and biodiversity offsetting in that PES schemes are designed to encourage and reward positive outcomes (gain, or maintenance of biodiversity and ecosystem services) whereas biodiversity offsetting is designed to balance negative outcomes (loss). Therefore, aggregation within PES schemes carries with it less risk than does using more aggregated currencies to evaluate biodiversity offsetting proposals, except when PES schemes are operating as compensatory mechanisms to address negative impacts (e.g. Robertson *et al.* 2014).

More disaggregated currencies may be more costly and complicated to use than more aggregated currencies as they replace single composite measures with many separate measures, each of which must be separately accounted for. However, many aggregated currencies require measurement of the biodiversity elements that are ultimately bundled into a single metric (e.g. the 'habitat hectares' approach (Parkes *et al.* 2003) requires in-field measurement of ten habitat elements). Requiring a greater total number of biodiversity elements to be independently measured and accounted for may well increase cost and complexity of using more disaggregated currencies. However, in many situations these disadvantages are likely to be outweighed by the reduced risk of unaccounted-for biodiversity loss, improved transparency as to what is included in the trade, and enhanced ability for evaluation of the offset proposal by stakeholders leading to improved social equity. This is particularly true for nations where biodiversity is reasonably well known such as New Zealand, or where there is a willingness to undertake thorough assessments in order to add to biodiversity knowledge. Elsewhere, such as areas where information on biodiversity is poor, the additional time and cost associated with using highly disaggregated currencies may prove prohibitive.

The Disaggregated Model we describe here provides an improved means to evaluate ecological equivalency by individually describing all biodiversity elements of interest and, for each, balancing losses at an impact site with anticipated gains at an offset site. Thus 'what we care about' is explicitly identified and matched with what is measured and individually evaluated. Exactly what constitutes 'what we care about' (the target biota to be offset) is a crucial component of an offset design but can be challenging to define (Bull *et al.* 2016; Maron *et al.* 2016a). The Disaggregated Model does not dictate the target biota to be offset and this would ideally be defined by clearly stated policy or conservation objective, or, in the absence of such direction, determined by stakeholders. Although designed to estimate biodiversity offset requirements within the largely voluntary New Zealand context, it is equally applicable for making transparent the set of assumptions behind any offset calculation. While the targets of biota to be exchanged would differ, the logic of calculating losses and gains in an offset exchange are universal, thus the model's non-prescriptive structure allows its application across diverse developments and planning frameworks.

The concept of offsetting has also been used to counteract losses of ecosystem services, most notably through compensatory mitigation of wetlands (a habitat trading programme) in the United States (Palmer & Filoso 2009). However, although there are regulatory provisions within this programme that are well suited to incorporating ecosystem services, this inclusion is only implicit (Ruhl & Gregg 2001) and the success of wetland restoration in providing ecosystem services has not yet been proven (Palmer & Filoso 2009). Further, it remains that ecosystem services are not in themselves being traded (Womble & Doyle 2012) but rather the underlying biota is what is subject to exchange. Consequently, the challenges faced when trading ecosystem services share similarities with biodiversity offsetting. For example, the lack of common metrics to describe and measure ecosystem function (Ruhl & Gregg 2001) and incomplete understanding of restoration outcomes (Palmer & Filoso 2009). In addition, consideration of landscape context and location of the natural capital stocks necessary for many ecosystem processes that give rise to ecosystem services to occur is crucial when trading ecosystem services. Social values of equity, proximity, and access across space and time (who wins and who loses) are also critical considerations in locating sites designed to deliver ecosystem services (Corbera *et al.* 2007; Womble & Doyle 2012). The non-prescriptive structure of the Disaggregated Model does however provide some potential for use in assessing the biophysical component of the provision of ecosystem services at an impact site and offset site. In these situations the target biota to be offset would be identified through a ecosystem services lens rather than a purely conservation lens. Thus, the biodiversity elements entered into the model would be those stocks of natural capital that are known to contribute to the provision of required ecosystem services and that are also responsive to offset actions (Maseyk *et al.* 2016).

Planning framework

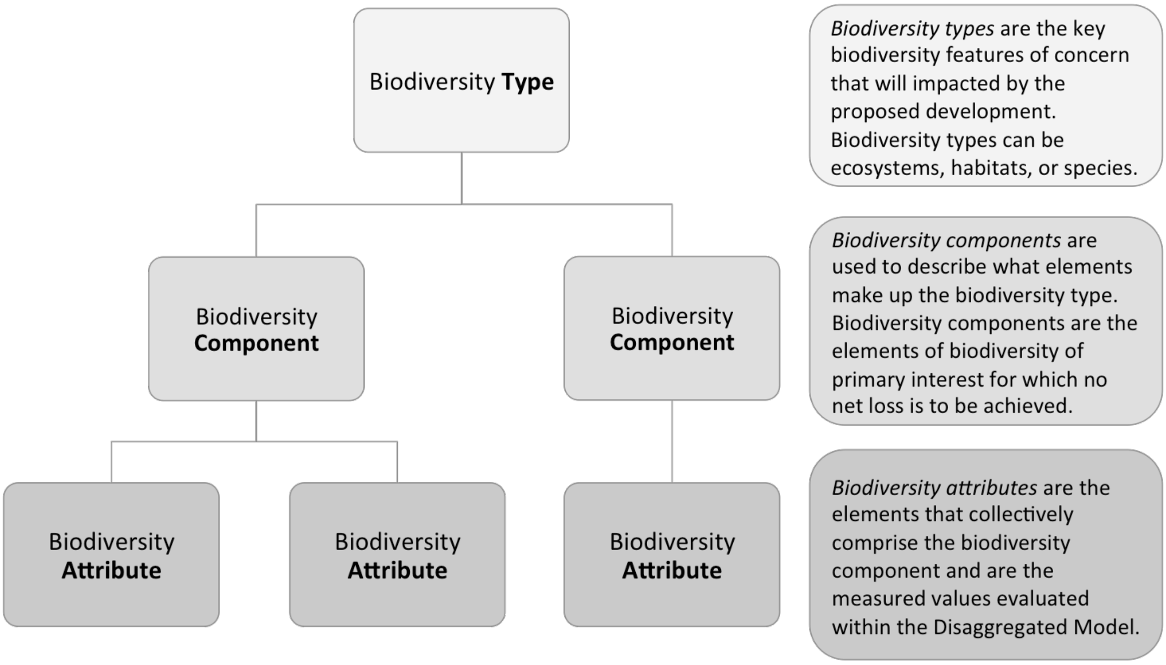
New Zealand has been identified as a global biodiversity 'hotspot' largely due to its high rates of endemism (e.g. 65% of vertebrate species and 51% of plant species) (e.g. managable natural capital stocks, Myers *et al.* 2000) and diversity of ecosystems. Despite the recognised conservation importance of New Zealand's indigenous biodiversity, the country is facing a biodiversity crisis, with a higher proportion of its native species listed by the IUCN as threatened or at risk of extinction than any other country (Bradshaw *et al.* 2010). Since human occupation *ca.* 1280 A.D (Wilmshurst *et al.* 2008), the devastation of New Zealand's indigenous biota has been extensive and rapid. New Zealand's landscapes have also been heavily transformed since human occupation with the loss of approximately 75% of the indigenous forest cover (Ewers *et al.* 2006) and 90% of wetland habitat (Myers *et al.* 2013). Notwithstanding the high proportion of New Zealand's land mass protected as public conservation land (32% (8.5 million ha)) administered by the Department of Conservation (Department of Conservation 2014a), much unprotected biodiversity remains on privately owned land where indigenous vegetation cover and biodiversity continues to decline (Craig *et al.* 2000; Walker *et al.* 2006; Myers *et al.* 2013).

New Zealand currently lacks overarching national policy for biodiversity conservation and associated direction for biodiversity offsetting. Consequently, while statutory obligations for offsetting in local jurisdictions are increasing, there is neither a coherent statutory requirement to offset biodiversity losses, or a voluntary policy framework that recommends requirements for an offset proposal. Despite the lack of clear and consistent policy, voluntary biodiversity offsetting is being proposed to address high profile development proposals in New Zealand.

In 2014 the New Zealand Government released non-statutory guidance for the development and assessment of biodiversity offsets aimed at demonstrating no net loss (Department of Conservation 2014b). The development of the Disaggregated Model described here was instigated by the Department of Conservation as a publically-available tool in support of this guidance.

The mechanics of the Disaggregated Model

The assessment of ecological equivalence requires that the same metric is used to measure and describe both losses and gains (Quétier & Lavorel 2011). To provide a measure of equity in biodiversity exchange, the Disaggregated Model uses net present biodiversity value (NPBV) which combines concepts from systematic conservation planning (e.g. biodiversity value functions) and finance (e.g. net present value and time discounting) (Overton *et al.* 2013). The structural foundation of the Disaggregated Model is a hierarchical framework developed within New Zealand's good practice guidance (Department of Conservation 2014b) that categorises biodiversity elements into three levels: type; component; and attribute. The biodiversity type is key biodiversity feature of concern and can be an ecosystem, a habitat, or a species (e.g. Podocarp/tawa forest); biodiversity components are used to describe the biodiversity type and represent the biota of primary interest for which no net loss is to be achieved (e.g. emergent trees within the area of Podocarp/tawa forest); biodiversity attributes are the elements which comprise the biodiversity components and are measured and balanced within the Disaggregated Model (e.g. number of individuals or emergent trees). The three levels collectively describe the biodiversity at both the impact and offset sites (Figure 5.1).



Biodiversity Type	Biodiversity Component	Biodiversity Attribute
Podocarp/tawa forest	Emergent trees	Number of individuals
		Crown cover
	Canopy	Native cover
North Island kaka	Breeding population size	Number of breeding pairs

Figure 5.1: Conceptual diagram of the hierarchal levels used to categorise biodiversity in the design of an offset (as follows the Guidance on Good Practice Biodiversity Offsetting in New Zealand (Department of Conservation 2014b) illustrated with hypothetical examples. Collectively, the levels describe 'what we care about' in the context of a biodiversity offset proposal. In this example from New Zealand, a proposed development will impact on two 'types' of biodiversity: podocarp/tawa forest and the North Island kaka.

The Disaggregated Model individually evaluates each biodiversity attribute (the measure with the highest resolution within the hierarchy) using NPBV to demonstrate no net loss at the attribute level, and aggregates attributes to evaluate NPBV and demonstrate no net loss at the biodiversity component level. When aggregating to the component level, individual attribute level NPBV are retained as outputs, thus explicitly identifying any tradeoffs among attributes. Aggregation can only occur *within* biodiversity components and not at the two higher levels (i.e. aggregation cannot occur across biodiversity components or biodiversity types). As such, biodiversity components are the elements of biodiversity of interest, and of which no net loss is to be achieved. Data inputs required for the Disaggregated Model are listed in Table 5.2 and a worked example is provided in Table 5.3.

Table 5.2: Data requirements for the Disaggregated Model. These data are entered for *each* biodiversity attribute of interest. As the Disaggregated Model is non-prescriptive, these values are not fixed within the model and all are required to be input by the user.

Data requirements	**Explanation**
Impact Model	
Measurement unit	The unit (e.g. count, percentage, frequency etc.) used to measure the biodiversity attribute.
Area of impact	The area (recorded in hectares) that supports the biodiversity type and over which the biodiversity attribute will be impacted by the proposal.
Benchmark	The benchmark value is specific to each biodiversity attribute. Measurements of ecological condition or quality require reference to a benchmark state that reflects a 'natural', 'pristine' or other desirable condition. The benchmark provides an objective framework and a common reference point for the evaluation of biodiversity losses and gains at impact and offset sites.
Measure prior to impact	The measured value of the biodiversity attribute at the impact site prior to the proposed impact occurring. This measure is adjusted against the benchmark value within the Disaggregated Model calculations.
Measure after impact	The estimated value of the biodiversity attribute at the impact site assuming the proposed impact has occurred. This measure is adjusted against the benchmark value within the Disaggregated Model calculations.
Offset Model	
Discount rate	A discrete discount rate to address equity over time as chosen and justified by the user.
Proposed offset actions	Brief detail of the proposed offset actions (management intervention) is entered into the Disaggregated Model. This information is not used in the calculations, but provides useful context and justification for the chosen confidence level.
Offset area	The area (recorded in hectares) over which the offset actions related to the biodiversity attribute will be implemented.
Confidence in offset actions	This is an estimation of the likelihood that the proposed offset actions will be successful within the specified time horizon. This estimation reflects that even with proven management techniques some uncertainty regarding outcomes is always present. There are three fixed levels of confidence within the Disaggregated Model from which the user choses one for each offset action. The confidence levels are defined as: *Low confidence* The proposed offset action uses methods that have either been successfully implemented in the situation and context relevant to the offset site but infrequently, or the outcomes of the proposed offset action are not well proven or documented, or success rates elsewhere have been shown to be variable. Likelihood of success is >50% but <75%. *Confident* The proposed offset action uses well known and often implemented methods which have been proven to succeed greater than 75% of the time although enough complicating factors and/or expert opinion exists to not have greater confidence in this offset action. Likelihood of success is greater than 75% but less than 90%. *Very confident* The proposed offset action uses methods that are well tested and repeatedly proven to be very reliable for the situation and context relevant to the offset site; evidence-based expert opinion is that success is very likely. Likelihood of success is >90%. The Disaggregated Model prevents offset actions with a confidence of <50% to be proposed within an offset design. The user-selected confidence level dictates the multiplier applied to the condition measure at the offset site to account for uncertainty in the offset action being successful.

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Data requirements	Explanation
	The multiplier is set as the mid-point of the range of likelihood of success within each confidence level (low confidence midpoint = 62%, multiplier = 0.62; Confident midpoint = 82.5%, multiplier = 0.825; Very confident midpoint = 95.5%, multiplier = 0.955).
Benchmark	The benchmark value entered for each biodiversity attribute in the Input Model is transposed into the Offset Model.
Measure prior to offset	The measured value of the biodiversity attribute at the offset site prior to the proposed offset occurring. This measure is adjusted against the benchmark value within the Disaggregated Model calculations.
Measure after offset	The estimated value of the biodiversity attribute at the offset site assuming that the proposed offset has occurred. When using the five-yearly time interval option of the Offset Model, this is initially the estimated value of the biodiversity attribute at the offset site prior to the proposed offset occurring, and thereafter the estimated value at each time interval (Year 1, Year 5 etc.) This measure is adjusted against the benchmark value within the Disaggregated Model calculations.
Time until endpoint	The anticipated number of years from the time of implementing the offset actions until the offset actions are expected to achieve the offset goal. This is a static time-horizon (e.g. 25 years) when using the finite time horizons option of the Offset Model, and set at Year 1, 5, 10…35 when using the five-yearly time interval option of the Offset Model.

Table 5.3: A hypothetical example using the Disaggregated Model. Panel A shows how biodiversity elements to which the model calculations will relate are described and measured; benchmark measure for each attribute; and proposed offset action. Panel B shows measures and biodiversity value for each attribute at both the impact and offset sites. Panel C shows the net present biodiversity values and whether a no net loss exchange between the impact and offset site at both biodiversity attribute and biodiversity component has been demonstrated, using a discount rate of 3%. This example illustrates how the Disaggregated Model aggregates within a biodiversity component and thus allows for substitution between biodiversity attributes but transparently identifies tradeoffs between attributes. Users and decision makers would decide whether a loss of a given attribute is acceptable or not. A simplified example is provided here, for more detailed scenarios and worked examples see the Disaggregated Model User Guide (Maseyk *et al.* 2014).

A

Biodiversity Type	**Biodiversity Component**	**Biodiversity Attribute**	**Measurement unit**	**Benchmark**	**Proposed offset action**	**Area of offset (ha)**
1 Podocarp/tawa forest	**1.1** Emergent trees	**1.1a** Number of emergent individuals	Count (#)	25	Restoration of habitat condition	600
		1.1b Crown cover	Percentage cover (%)	90		
		1.1c Basal area	Basal area (m^2/ha)	6		
	1.2 Canopy	**1.2a** Native representation	Percent (%)	100		
		1.2b Cover of native vascular plant species in the canopy	Percentage cover (%)	95		
		1.2c Height of canopy	Height (m)	20		
2 North Island kaka	**2.1** Breeding population size	**2.1a** Number of breeding pairs	Count (#)	4	Restoration of habitat condition and intensive predator control around nest sites	600

B

		Impact site				*Offset site*				
Biodiversity attribute		**Area of impact (ha)**	**Measure prior to impact**	**Measure after impact**	**Biodiversity value at impact site**	**Confidence in offset action**	**Measure prior to offset**	**Measure after offset**	**Time until end point (years)**	**Biodiversity value at offset site**
1.1a	Number of emergent individuals	150	13	0	-78	Very confident (> 90%)	13	13	10	0
1.1b	Crown cover		50	0	-83		65	85		95
1.1c	Basal area		1	0	-32		1	2		12
1.2a	Native representation		45	0	-68		40	95	25	151
1.2b	Cover of native vascular plant species in the canopy		50	0	-79		50	90	25	115
1.2c	Height of canopy		20	0	-150		15	20	30	59
2.1a	Number of breeding pairs	150	1	0	-38	Confident (75–90%)	1	2	35	44

C

Attribute level Net Present Biodiversity Value calculations				*Component level Net Present Biodiversity Value calculations*			
Biodiversity attribute		**Attribute Net Present Biodiversity Value**	**No Net Loss at attribute level**	**Biodiversity component**		**Component Net Present Biodiversity Value**	**No Net Loss at component level**
1.1a	Number of emergent individuals	-78	No				
1.1b	Crown cover	11	Yes	**1.1**	Emergent trees	-29	No
1.1c	Basal area	-19	No				
1.2a	Native representation	83	Yes				
1.2b	Cover of native vascular plant species in canopy	36	Yes	**1.2**	Canopy	9	Yes
1.2c	Height of canopy	-91	No				
2.1a	Number of breeding pairs	6	Yes	**2.1**	Breeding population size	6	Yes

No net loss is demonstrated for each attribute when NPBV equals or exceeds zero. This core output of the Disaggregated Model clearly identifies ‘winners and losers’ within an exchange when no net loss is demonstrated for some attributes but not for others (Table 5.3). In the example provided in Table 5.3, a no net loss of exchange is only demonstrated for four of the seven biodiversity attributes modelled (and two of three biodiversity components). At this point of proposal evaluation, offset actions can be adjusted (e.g. applied to a larger area, or additional actions modelled that may have a greater success of increasing biodiversity gain at the offset site). For some biodiversity attributes it can be very difficult to demonstrate a no net loss exchange either because these attributes do not respond directly to management actions, or respond over very long time-periods beyond those acceptable for an offset (e.g. attributes associated with emergent trees). Such biodiversity attributes represent limits to what can be offset and would be permanently lost or otherwise socially inequitable if a proposal was to proceed.

The model outputs can also be assessed at the component level by aggregating with equal weighting (averaging) attribute NPBV scores to produce a NPBV at the biodiversity component level — the level of primary interest. The model makes no presumptions as to whether some attributes are more or less ‘important’ than others, and weights all attributes equally by calculating the arithmetic mean of the attribute NPBV. The arithmetic mean is appropriate here as the average is calculated across the common metric of the NPBV score and not across the attribute condition measures. This allows a degree of transparent substitution at the attribute level where a gain in one attribute can compensate for a loss in another (Table 5.3). However, what is captured by this aggregation is explicit, and the structure of the Disaggregated Model prevents substitution at the component and type levels of the biodiversity hierarchy. Thus, both aggregation and the arbitrary weighting of attributes can only occur below the level of ‘what we care about’ (as captured by biodiversity components) and not beyond. Whether this aggregation is acceptable in any given offset situation becomes part of the decision making process.

The Disaggregated Model requires data inputs in the form of both measured and estimated values (Table 5.2) that typically would require ecological expertise to generate (e.g. suitably qualified ecologists or conservation scientists). Measurement, estimation, or prediction of data for input into the Disaggregated Model requires effort commensurate with the degree and complexity of impact on biodiversity, availability of existing information, and level of uncertainty in outcomes of proposed offset actions. This will often involve a pragmatic compromise between the cost of obtaining data and stakeholder expectations. Stakeholders expect greater precision and certainty for larger impacts on highly-valued biodiversity and transparent documentation of how these data were derived. As the methods used to generate the data required to account for losses and gains in an offset proposal (e.g. predictive models, analysis of field data etc.) are not incorporated into the Disaggregated Model, we recommend that the user compiles exhaustive and transparent supporting documentation.

The Disaggregated Model does not account for spatial location of the offset, except implicitly, insofar as inputs must account for all factors influencing biodiversity, including spatial context. Thus spatial considerations of an offset design can be addressed outside the model, potentially using spatial planning processes, or restraining the spatial scale within which an offset is valid. Beyond space (where), time (when), and type (what), there are many other types of flexibility in biodiversity offsets including the why, who, and how (Bull *et al.* 2015). These and other types of flexibility identified and described by Bull *et al.* (2015) are not addressed within the Disaggregated Model, with the exception, in part, of 'how'. We treat different means of achieving the same gains in biodiversity values as interchangeable (Bull *et al.* 2015).

Mathematical approach

The Disaggregated Model uses currencies in the form of explicit field measurements or quality scores for each biodiversity attribute relative to a benchmark for that attribute (Overton *et al.* 2013). The benchmark provides a reference condition measure specific to each biodiversity attribute which reflects a 'natural' condition and therefore the desirable state. Measures of attribute condition at both the impact site and offset site are capped at 100% of the benchmark value. In this simplified approach an exceedance of the benchmark value is considered neutral. However, this unintentionally allows for any negative outcome of exceeding the benchmark value to be regarded as a gain, highlighting the importance of maintaining ecological expertise and oversight in the application of the model. The model also assumes a positive linear relationship between a measured quantity of a biodiversity attribute and the value of that attribute to the point where it plateaus as the benchmark value is reached (or exceeded). Thus the model uses 'proportion of the benchmark' as a proxy for relative quality when measuring biodiversity attributes pre and post impact and offset. In theory, many of the implied linearity in our approach could be made non-linear, but to do so would be both difficult due to the current limited understanding of such relationships, and impractical in terms of maintaining simplicity of the model.

In essence, the model compares the Biodiversity Value (BV) at the impact site (a negative value) to that at the offset site (a positive value). The BV at the impact site is the difference in the condition of a biodiversity attribute before and after impact, divided by the benchmark value and multiplied by the area of impact. The BV at the offset site is the difference in the condition of a biodiversity attribute post and prior to offset, divided by the benchmark value, adjusted to account for uncertainty, discounted for time delay between the impact and the offset benefit occurring, and multiplied by the area of the offset. The final calculation returns the NPBV for each biodiversity attribute and component (Figure 5.2, Table 5.3).

The Disaggregated Model calculations estimate ecological equivalency against a static baseline of 'before the impact/offset'. Thus, the model does not permit averted loss credits (e.g. from protecting habitat that would otherwise be under threat) and deliberations regarding any offset benefits from averted loss remain a component of the wider decision making process external to the model. It also does not allow for scenarios in which biodiversity values would improve without intervention.

Accounting for uncertainty

The Disaggregated Model is limited in its treatment of uncertainty, incorporating only an estimate of the likelihood of success of an offset action. This uncertainty is expressed as the level of confidence that the proposed offset action will result in the predicted gain in biodiversity value ('low confidence', >50% <75% chance of success; 'confident', >75% <90%; 'very confident', >90% as defined in Table 5.2). The level of confidence is determined by a user with appropriate ecological expertise. Evidence or justification for this choice should be provided in accompanying documentation. A multiplier, the rate of which is set at the mid-point of the percentage range associated with each confidence level (e.g. selecting the low confidence level returns a multiplier of 0.62), is applied to the post-offset condition value for each biodiversity attribute. This multiplier differs from others (Moilanen *et al.* 2009; e.g. Laitila *et al.* 2014) in that a value < 1 is applied to the expected gain at the offset site to discount estimated benefit, compared with a value (typically > 1) applied to the loss at the impact site to estimate the gain required. The mathematical principle of the two approaches are the same with both adjusting the amount of offset gain required to achieve no net loss.

Accounting for equity across time

The Disaggregated Model uses discounting, incorporated into the calculation of NPBV (Figure 5.2), to address the time-lag between future biodiversity gains (at the offset site) being realised and biodiversity losses occurring (at the impact site). Using a discount rate allows future values to be expressed in equivalent terms to present values, as long as a feasible time preference can be estimated (Dunford *et al.* 2004). Structurally embedding the application of a discount rate into the model ensures that accounting for equity across time is directly integrated into calculations of ecological equivalency. However, the Disaggregated Model does not prescribe the rate of discount requiring the user to determine this value.

Input (user defined)		Impact Model	Offset Model
Attribute benchmark	B_i	Yes	Yes
Area of impact / offset	a	Yes	Yes
Attribute measure pre-impact/offset	$M_{pre}A_i$	Yes	Yes
Attribute measure post-impact/offset	$M_{post}A_i$	Yes	Yes
Rate of discount	d		Yes
Uncertainty multiplier (derived from level of confidence in success of offset action)	x_i		Yes
Time at which offset is expected to demonstrate no net loss	t		Yes

A. Impact Model

1. *Condition of attribute pre and post impact*
The post and prior to impact measures of a biodiversity attribute are divided by the benchmark value for that attribute
$C A_i$
$\frac{M_{post}A_i}{B_i}$ $\frac{M_{pre}A_i}{B_i}$

2. *Difference in condition pre and post impact*
The difference in the condition of a biodiversity attribute post and prior to impact is calculated
ΔA_i
$C_{post}A_i - C_{prior}A_i$

3. *Change in condition multiplied by area calculation*
The condition of a biodiversity attribute is multiplied by the area of the impact
$\Delta A_i \times a$

4. *Biodiversity value (BV) at the Impact Site*
Impact BVA$_i$

B. Offset Model

5. *Condition of attribute pre and post offset*
The post and prior to offset measures of a biodiversity attribute are scored against the benchmark value for that attribute
$C A_i$
$\frac{M_{post}A_i}{B_i}$ $\frac{M_{pre}A_i}{B_i}$

6. *Difference in condition pre and post impact*
The difference in the condition of a biodiversity attribute post and prior to offset is calculated
ΔA_i
$C_{post}A_i - C_{prior}A_i$

7. *Change in condition adjusted for confidence in offset action*
A user chosen multiplier is applied to the attribute condition value to account for uncertainty of the offset action being successful
$\Delta A_{i\ adjusted}$
$\Delta A_i \times x_i$

8. *Discount for time lag*
The adjusted condition value is further adjusted to account for the time-lag between the impact occurring and the offset being achieved
$\Delta A_{i\ discounted}$
$\frac{\Delta A_{i\ adjusted}}{(1 + d)^t}$

9. *Discounted condition multiplied by area calculation*
The discounted condition of a biodiversity attribute is multiplied by the area of the offset
$\Delta A_{i\ adjusted} \times a$

10. *Biodiversity value (BV) at the Offset Site*
Offset BVA$_i$

11. *Net Present Biodiversity Present Value (NPBV) at the attribute level* The difference in the Offset BVA and the Impact BVA is calculated to derive the NPBV for that attribute
NPBVA$_i$
Offset BVA$_i$ – Impact BVA$_i$

12. *Net Present Biodiversity Present Value (NPBV) at the component level* The mean of attribute NPBV is calculated for each component.
NPBVC$_i$
$\frac{\sum (NPBVA_i)}{\text{Total number attributes}}$

Figure 5.2: Process diagram of the Disaggregated Model showing inputs and step-wise calculations for both impact and offset models.

Determining an appropriate rate and form of discounting is no easy task given the uncertainties inherent in trading biodiversity and difficulties in truly compensating for temporal losses (Bekessy *et al.* 2010; Walker 2010; Gardner *et al.* 2013), but it is an important consideration as calculations are sensitive to the rate used (Moilanen *et al.* 2009; Denne & Bond-Smith 2011; Gibbons *et al.* 2015). Users of the Disaggregated Model may choose, for example, to refer to New Zealand's good practice guidance document (Department of Conservation 2014b) and supporting documentation (Denne & Bond-Smith 2011) for guidance on choosing an appropriate discount rate. Evans *et al.* (2013) provide an overview of use of discount rates and introduce an approach to account for 'ecological time preference' over time and space when calculating offset requirements for threatened species. Dunford *et al.* (2004), Laitila *et al.* (2014), and Cole (2011) also provide valuable discussion on discounting for equity across time. We recommend that discount rates used within the Disaggregated Model are described and justified within supporting documentation.

Calculating net present biodiversity value across time-steps

The Disaggregated Model calculates the total biodiversity gains at a stated end-point of an offset proposal. However, biodiversity gains often accrue gradually. In some situations it may be of use to decision makers to understand when gains in biodiversity attributes are expected or when no net loss is demonstrated. The Disaggregated Model allows for this as an option via a sub-model that calculates cumulative NPBV at five-yearly time intervals to a maximum of 35 years (the maximum life of a resource consent (permit for activities controlled by a resource management plan) in New Zealand).

Discussion

Our Disaggregated Model is an advance on models that use simple, aggregated currencies because it reduces concealed trades. It does this by explicitly calculating NPBV for each biodiversity attribute, identifying any tradeoffs between attributes when aggregating to the component level, and preventing aggregation beyond the component level. Other offset accounting systems also use net present value to evaluate offset proposals such as the Australian Federal Government's Environmental Protection and Biodiversity Conservation Act (EPBC) Environmental Offsets Policy assessment guide (Miller *et al.* 2015), and a recently developed, simple loss-gain calculator (Gibbons *et al.* 2015). Both these calculators can accommodate disaggregated currencies, but do so less explicitly than the Disaggregated Model. The Disaggregated Model allows for flexibility in a number of factors including: type of offset action; discount rate; the anticipated number of years from the time of implementing the offset actions until the offset actions are expected to achieve the offset goal; calculation of NPBV at a stated end-point, or cumulatively over five-yearly intervals, and can relatively easily be run multiple times for different offset scenarios and for different target biota. This flexibility shares similarities with other approaches for evaluating equivalency such as the Habitat and Resource Equivalency Analysis (HEA and REA) applied in natural resource damage assessment (Quétier & Lavorel 2011). Although the HEA/REA can be applied in a disaggregated way, this is not always the case and the use of proxies or aggregation of several metrics can occur.

Limitations of the Disaggregated Model

Use of an area by condition currency

Single area by condition currencies rest upon two fundamental assumptions: 1) *substitution*, that both area of habitat and condition of habitat, and different elements of condition, are equitable and exchangeable; and 2) *surrogacy*, that area and condition adequately capture all other aspects of biodiversity representation by proxy, although there is no theoretical basis that validates these assumptions. Therefore, such currencies are unlikely to adequately account for the full range of biodiversity at an impact site (Overton & Stephens 2015). The Disaggregated Model avoids the surrogacy issue by applying the area multiplier separately to the condition measure (biodiversity value) of each biodiversity attribute entered into the model. However, it is still subject to potential substitution between area and condition within each attribute.

Unrealistic counterfactual scenarios

Counterfactual scenarios describe what is expected to happen in the absence of the offset and thus provide a baseline trajectory against which offset gains can be estimated (Maron *et al.* 2013; Maron *et al.* 2015). Unrealistic counterfactual scenarios are an issue in biodiversity offsetting practice generally, which the Disaggregated Model does not resolve. The Disaggregated Model assumes a counterfactual scenario based on a static baseline (no expected background loss or gain). In making this assumption the Disaggregated Model disregards gains made by averting loss (e.g. through protection and maintenance of condition which otherwise would decline) and accepts gains that would have occurred anyway in the absence of the offset (for example, if the condition of a site is improving without intervention).

While a static baseline is an unlikely counterfactual scenario for many elements of biodiversity, it was adopted due to the high uncertainty in empirically describing counterfactuals and setting accurate and defensible baselines. Allowing the use of declining baselines carries risks: if the rate of decline is overestimated, this steep declining biodiversity trajectory becomes entrenched as the net outcome from the impact and the offset need only match the baseline trajectory, i.e. a smaller offset is required (Gordon *et al.* 2015; Maron *et al.* 2015). Conversely, an overly optimistic trajectory of improvement would require greater offset gains to be generated than required to demonstrate no net loss. Both errors incur costs: in the latter scenario these costs fall to the developer while in the former scenario they fall to the wider community and are paid for with further biodiversity loss. In developing the Disaggregated Model, the consequences of incorrectly describing the baseline trajectory were considered to have greater likelihood of jeopardising 'no net loss' goals than did assuming a static baseline, and thus accounting for dynamic baselines was not built into the model. However, where dynamic baselines specific to each biodiversity attribute can be reliably calculated (outside of the model), this background rate of change can be incorporated into the estimates of attribute condition post offset that are entered into the model. Any alternative baselines incorporated into estimated measures of biodiversity attributes should be well described and justified in supporting documentation to maintain transparency.

Dealing with uncertainty

The Disaggregated Model relies on a simplistic multiplier to account for the uncertainty of offset actions being successful but does not account for any other forms (e.g. theoretical, technical, operational, institutional, and financial) of uncertainty which should be addressed outside of the model during offset design. Other disciplines (e.g. engineering) have developed sophisticated approaches to dealing with uncertainty and risk that incorporate assessment of information adequacy at the outset, sequential decision making, iterative project management, and contingency planning (Pich *et al.* 2002). However, adequate multiplier rates will remain elusive until enough is known about trajectories of restoration and habitat recovery.

The incomplete manner in which the Disaggregated Model addresses uncertainty is not uncommon in other offset design support tools due to the current lack of methods to account for uncertainty. Uncertainty is a largely undeveloped aspect of biodiversity offsetting accounting that requires more research and development. However, biodiversity offsetting is a high-stakes endeavour, especially so if applied to threatened species and habitats. Despite the limitations, uncertainties, and ethical dilemmas (e.g. Walker 2010; Maron *et al.* 2012; Devictor 2015; Ives & Bekessy 2015; Moreno-

Mateos *et al.* 2015) associated with biodiversity offsetting, it has become increasingly popular as a policy approach and conservation tool and the practice is unlikely to cease. Thus, increasing robustness and transparency of offset proposals and maintaining a strict adherence to the mitigation hierarchy, limitations of offsetability, and other generally accepted principles remains imperative. Addressing uncertainty should be a critical component of the wider decision making process. This will require input from suitably qualified experts for each area of uncertainty (Miller & Lessard 2001). The current lack of ability to account adequately for uncertainty within biodiversity offsetting highlights the importance of monitoring and adaptive management.

Uncertainty can also be introduced into the model via the estimation or prediction of biodiversity attribute measures where direct measurement can not occur (e.g. the prediction of change in biodiversity attribute due to offset action). When expert judgement is relied on to generate data, inherent bias and uncertainty can be reduced using structured elicitation methods for arriving at estimations such the Delphi Method which has been used in other ecological models (e.g. MacMillan & Marshall 2006; Kuhnert *et al.* 2010).

Key features of the Disaggregated Model structure

Simplicity

The Disaggregated Model is driven by simple calculations and multipliers to account for time-lags and uncertainty. While not without limitations, these are transparent and manipulable, making the mechanics and outputs of the model easy to comprehend and avoiding mystifying 'black box' computations. To generate calculations, the Disaggregated Model requires only the kind of data that developers can reasonably be expected to assemble, such as reproducible field measurements of biodiversity attributes obtained using standard methods. Estimates of future measures (e.g. post impact and post offset) may require a further level of complexity (e.g. predictive population modelling or stand dynamics modelling), but could also potentially be informed by expert judgement when sufficient knowledge exists to inform it. Such estimates reflect a formalisation of assumptions that are always made (although often implicitly) in evaluating offsets (Maron *et al.* 2013). It is expected that the level of complexity of inputs would be commensurate with the complexity and conservation value of the impacted biodiversity. In any case, the relatively simple structure of the Disaggregated Model means input measures can easily be reviewed by stakeholders, facilitating objective assessment. Further, this relatively user-friendly model structure and interface allows for less arduous calculation of alternative offset proposals and easy comparison of model outputs.

Use of benchmark values

Benchmarks are used to describe a reference reflecting a natural or otherwise desirable ecological condition against which current and estimated future values at both the impact and offset site can be compared. Locating reference sites in a condition state suitable to define a benchmark can be challenging, especially in landscapes where biodiversity has become much reduced or modified. The Disaggregated Model requires benchmarks for each attribute. When defensible reference sites are lacking, modelling or expert opinion will be needed to replace field measurements. Although feasible, this can be more costly and potentially less transparent and robust. However, the use of structured expert elicitation methods can help reduce the later two issues.

Despite challenges in defining defensible values, the use of benchmarks allows for a fairer approach for both sides of the trade. Referencing sites to a benchmark reduces the risk of false claims of poor condition at either an impact or offset site, which could lead to under-estimation of loss and over-estimation of gain. The use of benchmarks also caps predicted gains from offsets by restricting values to no more than 100% of their potential (benchmark) value. The limitations and complexity associated with benchmarks adds complexity to the input requirements of the Disaggregated Model, but maintains ecological robustness and in doing this so explicitly improves transparency.

The potential for application of the Disaggregated Model to estimate out-of-kind exchanges of biodiversity

Strict equivalency requirements have been questioned on the grounds that they can increase costs as finding a suitable offset (action or location) can be difficult (Habib *et al.* 2013), creating fewer options (Gibbons *et al.* 2015). We suggest that restricting divergence between that lost and that gained is appropriate for biodiversity of conservation concern, with restrictions tightening as level of conservation concern increases (Pilgrim *et al.* 2013).

The adaptability of the Disaggregated Model to a simple approach for restricted out-of-kind (but ecologically similar) compensatory exchanges has recently been demonstrated (Overton & Stephens 2015). While we agree that there is scope for out-of-kind exchanges to achieve strategic conservation outcomes, we caution against the application of the Disaggregated Model for this purpose in the absence of well-defined restrictions on trade options and socially acceptable exchange rates among dissimilar attributes. Such a framework should at the very least ensure that the exchange always results in 'trading up' (greater gains for lesser losses) (BBOP 2012; IUCN 2014).

The potential for application of the Disaggregated Model for balancing losses and gains of ecosystem services

As the loss of biodiversity results in changes to or loss of ecosystem function and ecosystem services (MEA 2005; Hector & Bagchi 2007; Cardinale *et al.* 2012), the application of the Disaggregated Model as described here can be used to indirectly evaluate loss and gain of ecosystem services. The Disaggregated Model also has potential to more explicitly account for ecosystem services when trading biodiversity. By shifting the identification of target biota to offset from a conservation perspective to an ecosystem services perspective, biophysical elements linked to the supply of ecosystem services can be identified and entered into the model using the same hierarchical structure and thus maintaining a disaggregated approach. For example, instead of 'biodiversity type', ecosystem services type (e.g. water quality) can be described; instead of 'biodiversity component', natural capital stocks that contribute to the provision of water quality would be individually listed (e.g. soil and vegetation natural capital stocks); and instead of 'biodiversity attribute', attributes of natural capital stocks that are both involved in the supply of the identified ecosystem service and able to be managed (via an offset action) identified (e.g. top soil strength and root tensile strength). Benchmark values for natural capital stock attributes could be set at critical thresholds required for the supply of the ecosystem service in question, but this is likely to be challenging as our understanding of the relationship between quantity and quality of natural capital stocks and the provision of ecosystem services is incomplete. Our lack of understanding of this relationship warrants a precautionary approach to setting benchmarks for natural capital stock attributes.

Further limitations present themselves when contemplating the use of the Disaggregated Model in its current form to account for loss and gain when trading ecosystem services. The assumed linear relationship between quantity of a biodiversity attribute and value of that attribute within the Disaggregated Model would carry over when using the Disaggregated Model to estimate equivalency in exchanges of ecosystem services. That is, an assumed linear relationship between quantity of natural capital stock and supply of ecosystem service to the point it plateaus as the benchmark value for that attribute is reached. This represents a simplistic and likely unrealistic relationship between natural capital stocks and the provision of ecosystem services for many services (particularly regulating services) and fails to take into account critical interactions between different natural capital stocks.

The Disaggregated Model can only provide guidance on the biophysical realm of the provision of ecosystem services and this in itself does not measure ecosystem services. Therefore, the model outputs would need to be incorporated with evaluations of the social realm of the ecosystem services concept. In particular, aspects of demand, delivery flows to beneficiaries, and social equity across space and time are important considerations that need to be incorporated into the decision making process alongside the biophysical considerations. Without this full evaluation of the losses and gains in delivery of ecosystem services, and who wins and who loses, can not be meaningfully understood.

Despite these limitations, the Disaggregated Model theoretically would retain its key advantage of a highly disaggregated approach leading to more explicit estimation of equivalency and increased transparency of model outputs when applied using an ecosystem services frame, although this has yet to be tested.

Conclusions

As it is impossible to fully account for biodiversity loss across type, space, and time, offsetting is an imperfect and controversial response ameliorating the impact of development (Maron *et al.* 2016a). Despite this, and as development pressures on biodiversity intensify, it is probable that decision makers will continue to use offsetting to manage impacts. Thus, the need to improve how offset requirements are calculated remains an urgent challenge. The pervasive issues associated with biodiversity offsetting (across offset design, accounting, governance, and compliance) may prove intractable. In the meantime, advancing support tools such as the Disaggregated Model provides a useful contribution to offset design and assessment globally. The use of disaggregated currencies to describe biodiversity losses and gains in an offset exchange improves on aggregated metrics by allowing for unequivocal description and measurement of the biodiversity elements of interest, more explicit estimation of whether an offset proposal can achieve no net loss, reduced risk of concealed trades, and greater transparency in decision making.

CHAPTER SIX:

CONCLUSIONS

This final chapter reviews the results of the thesis, and identifies key learnings and limitations. I conclude the thesis by discussing potential directions for future research and provide closing comments.

Research overview and thesis contributions

In recent decades there has been an increase in ecosystem services research (Fisher *et al.* 2009; Dick *et al.* 2011; Martinez-Harms *et al.* 2015) and, even more recently, a growing recognition of the importance of natural capital stocks for the provision of ecosystem services (Bristow *et al.* 2010; Dasgupta 2010; Dominati *et al.* 2010; Bateman *et al.* 2011; Daily *et al.* 2011; Mace *et al.* 2011; Robinson *et al.* 2013; Barbier 2014). Framing the ecosystem services concept to recognise the critical role of natural capital has clarified how ecosystem services are derived and has been a crucial step in shifting ecosystem services thinking from the conceptual to the operational. In addition, much effort has been invested in developing practical linkages tools and resources for practitioners and decision makers to guide on-the-ground implementation of an ecosystem services approach (e.g. by The Natural Capital Project, The Nature Conservancy, and see Olander *et al.* 2015). Despite this considerable progress, there remain gaps in our understanding of how to make better decisions for the purpose of influencing ecosystem function and the provision of ecosystem services. This is a critical gap to fill as we face a global biodiversity crisis (Cardinale *et al.* 2012) and as the inherent capacity of ecosystems to provide for human wellbeing and survival continues to deteriorate globally (MEA 2005; Rockström *et al.* 2009; Steffen *et al.* 2015). Sustaining ecosystem function and the full range of ecosystem services (provisioning, regulating, and cultural) at the landscape scale requires the integration of multiple land uses and management actions which usually operate at the paddock or property scale. Thus, it is imperative that our ability to operationalise the ecosystem services concept at the property scale is improved. The work in this thesis is aimed at reducing the gaps between science, policy, and practice so as to increase the relevance and applicability of ecosystem services thinking to natural resource management and conservation decision making.

In this thesis I integrate data with ecological and social concepts to conclude that:

a) Formalising the relationship between natural capital stocks and the associated provision of ecosystem services enables more effective management of the provision of ecosystem functions and services.

b) Understanding how the benefits that arise from ecosystem functions and services and the costs in providing them are perceived and/or experienced by landowners can influence land use practices at the local scale as well as broad-scale decision making targeted at providing for multifunctional landscapes.

c) Ecologically robust accounting systems that avoid the use of surrogates and instead individually evaluate the consequences of actions for target biota improve the transparency of biodiversity offsetting proposals and assist in the decision making process.

In Chapter 2, *Managing natural capital stocks for the provision of ecosystem services*, I developed the conceptual framework that underpins the central principle of this thesis. The foundational framework encapsulates i) the relationship between natural capital stocks and the provision of ecosystem services, and ii) the relationship between management interventions and condition of natural capital stocks, thus showing how actions targeting natural capital stocks influence the provision of ecosystem services and associated benefits and values. This chapter advances the argument that natural capital stocks have attributes that are both manageable and less manageable, a notion first introduced into ecosystem services thinking by Dominati *et al.* (2010). Here I extend this concept to vegetation natural capital stocks and emphasise the importance of interactions between natural capital stocks. The differentiation between manageable and less manageable attributes is critical as it identifies management actions that will have the greatest effectiveness in influencing ecosystem service provision. I illustrated how a structured decision making process can be used to implement the framework using an example of comparing management scenarios for the conservation of soil natural capital. This chapter advances current ecosystem services research by providing both the conceptual framework and practical approach needed to operationalise ecosystem services thinking.

In Chapter 3, *Effect of management on natural capital stocks underlying ecosystem services provision: a 'provider group' approach*, I use a 'provider group' approach to empirically test the conceptual framework presented in Chapter 2 demonstrating how land management practices directly impact on the condition of vegetation natural capital stocks using a case study of a traditional farmed grassland system in the Southeastern Carpathians, Romania. The provider group approach categorises vegetation natural capital stocks into sets of species based on species attributes that directly or indirectly contribute to the provision of ecosystem services. For example, species that posses palatability and nutritional value for livestock are grouped into the 'quality fodder' group, and indirectly provide provisioning ecosystem services in the form of food and raw materials; whereas species producing pollen are grouped in the 'pollen' group both directly (via medicinal resources) and indirectly (via eventual food production) also provide provisioning ecosystem services. The use of this provider group approach provides a pragmatic and readily implementable methodology for assessing the effects of management practices on the provision of ecosystem services and improving understanding of local systems.

The benefits that flow from ecosystem service provision are an important component of the ecosystem services concept, as reflected in the conceptual framework presented in Chapter 2. However, human communities, and the individuals within them, do not universally share the same value sets, and can experience different benefits from the same ecosystem services (Turner & Daily 2008; Dick *et al.* 2011). Understanding what benefits and values (and thus 'relative advantage') may be perceived and/or experienced by particular groups of people (e.g. resource users or land owners) can increase understanding of their likely responses to management interventions (Pannell *et al.* 2006). In Chapter 4, *Farmer perspectives of the on-farm and off-farm pros and cons of planted multifunctional riparian margins,* I show that views of the pros and cons of planting riparian margins varies between farmers, with those that have planted riparian margins perceiving and/or experiencing benefits beyond the target objective of improving water quality and also acknowledging a variety of costs and liabilities. Farmers that had not planted riparian margins had different views of the potential benefits to be gained from planting. This group of farmers were sceptical about the casual relationship between the actions (planting riparian margins) and the stated objectives for doing so (water quality), preventing them from undertaking planting. The chapter progresses understanding of the socio-political drivers influencing farmer uptake of environmental interventions. This knowledge is a necessary contribution to closing the gaps between science, policy, and practice currently associated with the ecosystem services concept.

Biodiversity offsetting is a high-stakes endeavour that risks condoning and entrenching biodiversity losses (Walker *et al.* 2009; Maron *et al.* 2010; Walker 2010; Spash 2015) and consequently compromising flows of ecosystem services. This thesis' central theme — that natural capital stocks can be manipulated to influence the provision of ecosystem services — is central to the biodiversity offsetting narrative. Biodiversity offsetting has become increasingly popular as a policy approach and a conservation tool (Calvet *et al.* 2015; Gonçalves *et al.* 2015; Ives & Bekessy 2015; Rainey *et al.* 2015; Maron *et al.* 2016a), and despite the risks and limitations, some of which are highly entrenched and possibly intractable (Maron *et al.* 2016a), the practice is unlikely to cease. Thus, increasing the robustness and transparency of offset proposals is an urgent requirement. In Chapter 5, *A disaggregated biodiversity offset accounting model to improve estimation of* *ecological equivalency and no net loss*, I developed a disaggregated accounting model that explicitly and individually accounts for target biota for which no net loss is a specific goal, avoiding the use of surrogates or proxy measures. The model represents a crucial improvement on current accounting systems as greater disaggregation within offset accounting systems allows for more explicit estimation of whether an offset proposal can achieve no net loss for the target biota and thus provide greater transparency in decision making.

In sum, this thesis advances the application of ecosystem services thinking by identifying and illustrating that targeting manageable attributes of natural capital stocks provides a practical and readily applied pathway by which to target property scale management actions for the purposes of influencing the provision of ecosystem services. This thesis provides a collection of tools and methods to implement the framework and support decision making.

Caveats

While this thesis provides new knowledge to achieve change on-the-ground, it narrows, but does not close, the gap between science, policy, and practice. The outstanding limitations of the thesis are threefold:

1. *Quantification*. How much marginal change in ecosystem function or service is provided by specific management actions, and how this varies across space and time are important questions not answered in this thesis.
2. *Valuation*. The ability to assign value to ecosystem services and associated benefits allows us to compare the consequences of management actions and further increases the ability to internalise these considerations into decision making. This is a critical component of ecosystem services thinking that this thesis does not address.
3. *Uncertainty and risk*. There are a myriad of uncertainties and risks inherent in natural resource management and conservation decision making. The resolution of uncertainties which influence decision making and the improved understanding of the risks associated with actions (or non-actions) is critical for achieving better outcomes. This thesis does not advance methodologies for assessing risk and deals with uncertainty inherent in biodiversity offsetting in a limited and simplistic manner.

However, Chapter 3 and Chapter 4 do take preliminary steps in the quantification and/or valuation of ecosystem services and benefits arising from specific management action scenarios. In Chapter 3 the condition of natural capital stocks contributing to ecosystem service provision are described using species diversity and abundance measures. These measures are used as surrogate indicators of provision of specific ecosystem services, but are analysed without quantifying the flow of ecosystem services. In Chapter 4 participant farmers provided some limited quantitative estimations to describe and assist in valuing the services and benefits they perceived and/or experienced to flow from the planting of riparian margins. Accounting for uncertainty is one of the seven steps that form a structured decision making process (Gregory *et al.* 2012), and while I present structured decision making in Chapter 2 as a useful mechanism by which to operationalise ecosystem services thinking the subsequent chapters did not illustrate this process further. Chapters 3 and 4 focus on testing the central tenet of thesis — that attributes of natural capital stocks can be manipulated to influence the provision of ecosystem services — and make no attempts to define uncertainty.

Further limitations of this thesis are manifest in Chapter 5. While the biodiversity accounting offset model developed as part of this thesis represents an advance on existing accounting approaches, it is not without limitation as discussed in detail in Chapter 5. The model deals with uncertainty in a very incomplete and simplistic manner. Uncertainty is a largely undeveloped component of biodiversity offsetting and the limitation of the model to adequately deal with uncertainty is reflective of the current lack of methods to do so. The model also doesn't account for spatial inequities when trading biodiversity or the interactions between biodiversity components that may be required to sustain at landscape scales long-term. The model also uses a discount rate to account for equity through time but does not resolve philosophical and ethical ambiguities around rates of time preference appropriate for biodiversity values. While these issues are current limitations to biodiversity offsetting generally and not to the model specifically, this thesis has not made any headway in resolving them.

Addressing the shortcomings outlined above is required to more effectively apply ecosystem services thinking to decision making, and provide a useful starting point for future research.

Future research

Quantification and valuation

Mixed land uses will involve a compromise for all resource uses and beneficiaries of ecosystem service flows, particularly where systems are operating at the edge of their limits. Thus, the relative costs and benefits of particular land use and land management actions need to be clearly articulated in order to fully account for consequences and meaningfully incorporate this knowledge into decision making processes. The conceptual framework presented in Chapter 1 could be extended to incorporate costs, benefits, and efficiencies of various management actions. In support of this conceptual evolution, the development of methodologies to describe and measure services and benefits using a common (thus comparable) metric is a must. A required first step is the quantification of change in condition of natural capital stocks in response to management actions. This would increase understanding of critical thresholds of natural capital stock condition required to sustain functioning ecosystems and enable appropriate targets for maintaining natural capital stocks to be set (such as for example, a risk register as described in Mace *et al.* 2015). Concepts and advancements in environmental accounting, and particularly attempts to develop a common currency for measuring change in environmental assets (Wentworth Group of Concerned Scientists 2008; McDonald 2014), could be borrowed and adapted to quantify the impact of management actions on natural capital stocks.

Building on this idea, common currencies are also required to quantify and evaluate the full range of service provision (provisioning, regulatory, and cultural) flowing from the natural capital stocks subject to specific management actions. In Chapter 2 I presented an example of choosing between actions to address loss of soil natural capital stocks due to accelerated hill country erosion, damage on-farm due to erosion, and downstream due to flooding to illustrate how structured decision making can be used to integrate ecosystem services thinking into natural resource management. The study (Dominati *et al.* 2014) used in this example quantified the flow of a range of provisioning and regulating services in response to planting spaced trees in hill country and applied economic valuation to these ecosystem service measures not presented in this thesis. Dominati *et al.* (2014) provide a comprehensive study of the relationship between management actions at the property scale and ecosystem service provision. This type of assessment could be further built on to compare scenarios that would also allow for comparative evaluation of the contribution of different types of natural capital stocks. For example, is the flow of services greater from exotic or native natural capital stocks? How does this change when the full range (e.g. including cultural services) of services is included in the analysis? These questions are of particular relevance in landscapes with depleted native biodiversity, or where decision makers wish to align natural resource management and conservation policies. Finding a common currency is challenging when comparing between ecosystem services which more readily lend themselves to monetary valuation (e.g. provisioning services) and those that don't (e.g. cultural services). In the absence of valuation, quantification (particularly of change) becomes even more critical.

Cultural and social values contribute to whether policy driven management actions are adopted or not (Pannell *et al.* 2006) and value sets are not universal — individuals and groups of people perceive and value benefits differently from each other and through time (Turner & Daily 2008; Dick *et al.* 2011). Thus, there is a need to better understand the perceived and/or experienced benefits and associated values attributable to management actions at the property scale *alongside* the quantification of the full range of service flow resulting from these actions. Chapter 4 of this thesis provides some foundational work that would benefit from further exploration. A key next step is to further build on farmers' views and feelings by eliciting how the pros and cons of undertaking specific management actions on their properties translates into numbers. For example, farmer observations of the causal connections between planted riparian margins, provision of shade and shelter for livestock, increased grazing time, and "*more milk in the vat*" can be expressed numerically — *how much* more shade, shelter, grazing and milk? This is particularly important for quantifying provisioning and cultural services and this knowledge could then sit alongside ecological quantification of regulatory ecosystem services (e.g. water quality) to provide a deeper understanding of the full range of services and benefits that result from particular management actions. Choice modelling could be used to explore farmers preferences for certain benefits associated with planted riparian margins and willingness to invest in riparian margin management in order to gain particular benefits and how this varies between benefits, or sets of benefits. Such quantification of the cultural and social benefits and associated values is useful to evaluate current policy, guide current and future policy implementation practices, and inform the development of new policy.

Implementing ecosystem services thinking at either local or regional scales will require a variety of tools to increase knowledge and to support decisions. In situations where a system is not fully understood, or resources are not available for comprehensive data-collection, predictive modelling can be useful to generate a more quantitative description of ecosystem service flow. For example, modelling the predicted change in flow of a range of ecosystem services in response to various management scenarios would be a useful next step to build on the work presented in this thesis. Bayesian Belief Networks (BBN) have been proven to be a useful tool for exploring natural resource management questions (e.g. McCann *et al.* 2006; Quinn *et al.* 2013) and could be used to test the relationship between management scenario and change in flow of ecosystem services. A BBN can incorporate different types of data and levels of prior knowledge and thus, would allow the combination of social data collected in Chapter 4 with biophysical information. The simplicity and transparency of BBNs also make them a useful communication tool to describe issues and explore cause and effect. Model outputs would enable comparative analysis of marginal change in response to specific management actions and identify trade-offs and synergies between ecosystem services. Advancing such knowledge is useful to inform policy aimed at establishing and maintaining multifunctional landscapes and to direct priority actions at the property scale. While use of a BBN has potential to advance quantification of *change* in the provision of modelled ecosystem services under different management scenarios, they are limited by their inability to incorporate feedback loops or allow for two-way flow of causality. More sophisticated descriptions of causality, and absolute quantification of service provision will likely require more complex system models that explicitly integrate processes and interactions. However, describing marginal changes at local scales is important in terms of exploring trade-offs and synergies in the provision of ecosystem services (Haines-Young *et al.* 2012) and is a useful step towards improved quantification. Utility functions have been used to reflect different value sets and understand consequences of trade-offs between these (Grechi *et al.* 2014) and could be coupled with a BBN to improve valuation of marginal changes in ecosystem service provision under different management scenarios. The social data collected in Chapter 4 could provide a starting point to inform utility functions.

The conceptual framework presented in this thesis could be expanded to include quantification and valuation. Doing so would not only increase transparency in decision making by better accounting for trade-offs, but would also provide a robust foundation for determining whether current policies and practices are sustaining or depleting the natural capacity of ecosystems.

Better biodiversity offsets

The scope for improving biodiversity offsets is both wide and deep, with contentious issues across ethical, social, technical and governance realms (Maron *et al.* 2016a). The highly disaggregated model developed in Chapter 5 has increased the robustness of biodiversity accounting systems, but it could be further improved. While the model has progressed dealing with uncertainty associated with accounting for equity across biodiversity type, further theoretical and methodological developments are required to account for equity across space and time. This is a technical priority for biodiversity offsetting globally.

The use of a highly disaggregated currency within the model makes it more ecologically robust than more aggregated accounting systems. However, the question of *which* biodiversity elements to disaggregate and which are appropriate to aggregate (e.g. different canopy tree species within the same vegetation type; genes within species) remains. This decision ultimately determines which elements of biodiversity a like-for-like exchange is sought and is thus central to the debate — no net loss of what and compared to what? These are ethical and social issues (Maron *et al.* 2016a) with diverse views ranging from complete like-for-like exchanges favoured by environmentalists to the desire for the simplification of biodiversity into a single currency favoured by economists. Both these positions are impossible to deliver and it remains a matter of judgement as to whether, and where, the 'middle ground' can be found. Biodiversity offsetting policies can provide guidance by including explicit statements identifying target biota and frame of reference, but agreement on levels of what to include in currencies remains a philosophical issues that will be harder to resolve (Maron *et al.* 2016a).

Further improvements in biodiversity accounting systems are needed to better evaluate losses and gains across the full range of ecosystem services associated with trading biodiversity. The model developed as part of this thesis is organized around a hierarchical three-tiered structure that describes target biodiversity elements. This concept could be further developed to describe natural capital, natural capital stocks, and natural capital stock attributes that a) contribute to the provision of a specific ecosystem service, and b) are known to be amenable to management actions (i.e. proposed offset action). Restrictions on allowable substitution would be required to ensure all stocks contributing to service provision are accounted for (e.g. soil condition is not substituted for vegetation condition), and stock attributes are not substituted (e.g. tensile root strength for canopy interception of precipitation) where it would compromise service provision to do so. This, and other key principles of biodiversity offsetting, such as recognising limits to what can be offset (e.g. irreplaceable natural capital), should be built on to develop better methodologies and accounting systems that explicitly account for the full range of ecosystem services (i.e. beyond cultural services). In particular, spatial configuration of natural capital stocks is more critical for provision of some ecosystem processes and services (e.g. water quality) than others (e.g. carbon sequestration), and will need to be recognised as non-offsetable in certain contexts.

Concluding comments

The challenge to better understand how we can manage landscapes for the sustained provision of all the ecosystem services and associated benefits required for life is a universal one with global relevance. The ecosystem services concept promises new hope for multifunctional landscapes and greater equity in the distribution of benefits — promises that are largely yet to materialise. That this new thinking (or new packaging of older good ideas) has taken some time to become embedded into natural resource management and conservation decision making is not to suggest a lack of relevance. Rather, fully adopting an ecosystem services approach requires a considerable shift in mindset by individuals, organisations, and governments — a process that is never rapid. However, globally we are at a point where it is not a case of preventing change but of adapting to it. We urgently need to take action to halt further biodiversity declines and to more sustainably and equitably manage our finite natural resources. Ecosystem services thinking should not be seen as a threat to current attempts to do this, but as a complementarity pathway that broadens the scope for sustainable natural resource management and conservation objectives. There currently remains a gap between theory and practice that compromises the implementation of ecosystem services thinking. This thesis makes an important contribution to bridging that gap.

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