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Accounting for the links between social and ecological systems for effective nature conservation

Angela María Guerrero González

(BEng, MEnvMan)

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Abstract

Environmental problems and their solutions are determined by the complex ways in which social and ecological systems interact. Yet knowledge on these interactions and the challenges they pose to biodiversity conservation is scarce. Analytical and planning approaches to help understand and account for these challenges are needed. This is particularly important in the context of conservation initiatives that take a collaborative multi-stakeholder approach to decision-making. Such initiatives face numerous challenges, including the *problem of fit* – when governance systems do not fit or match the characteristics of the biophysical system. In this thesis novel methodological approaches and existing social-ecological system frameworks are employed to investigate the problem of fit in the context of biodiversity conservation, and to demonstrate how the interactions between social and ecological systems can be accounted for in conservation planning. This thesis represents an important contribution to social-ecological research.

I examine the relevance of the problem of fit to biodiversity conservation through a review of the challenges for the science of conservation planning related to a particular type of 'fit challenge': *scale mismatch*. A scale mismatch occurs when the scales for planning and implementing conservation actions do not match the scale of biophysical processes (Chapter 2, published in Conservation Biology). I identify how scale mismatches arise in association with the different stages of conservation planning (e.g. problem assessment, formulation of actions). I find that this type of fit challenge can result in partial solutions, ineffective actions, or actions that are not implemented. This review highlights the need for researchers and practitioners to understand and account for fit challenges in conservation research and management, and suggestions are provided on how this might be achieved.

I investigate the problem of fit through an assessment of the capacity for collaborative conservation initiatives to address three key challenges associated to social-ecological fit: *spatial scale mismatch (chapters 3 and 4)*, the *common management of areas (chapter 4)*, and the *management of interconnected ecological units* (chapter 4). In these empirical studies I analyse data collected through semi-structured interviews and a survey of stakeholders involved in 'Gondwana Link', - a large-scale conservation initiative that aims to restore ecological connectivity in the south west of Australia.

I characterise the interactions between stakeholders in this initiative as a conservation social network. I assess if the structure of the interactions enable the coordination of plans and actions across scales of planning and management (Chapter 3, published in

Conservation Letters). I apply a novel network theoretical approach to statistically analyse the different forms of stakeholder interactions, including cross-scale collaboration. I find that the structure of stakeholder interactions predisposed cross-scale collaboration for invasive animal control, an action where coordination of activities is necessary. For revegetation activities I find little evidence of collaboration across scales. This result suggests that addressing spatial scale mismatch could improve effectiveness of revegetation efforts. To achieve this, the conservation initiative should provide support to those stakeholders acting in a 'scale-bridging' role.

I extend the social network conceptualisation developed in Chapter 3 to incorporate interactions between elements of the ecological system (i.e. the level of connectivity between vegetation patches) and the conservation social network (Chapter 4, submitted to Global Environmental Change). I characterise the ecological interactions, the interactions between stakeholders, and the ways stakeholders are linked to different parts of these ecosystems (i.e. management in one or more locations) as a social-ecological network. Employing new theory and methodological approaches I identify different social-ecological network configurations that capture the hypothesised ways in which collaborative approaches could address specific social-ecological fit challenges, *including the common management of areas*, *spatial scale mismatch* and the *management of interconnected ecological units*. I apply new statistical models of multi-level networks to test the relative importance of possible social-ecological configurations in the observed social-ecological network. I find that co-management occurs when stakeholders manage the same spatially defined ecological resource, but not when they manage different yet interconnected ecological resources. This implies that Gondwana Link's governance structure lacks capacity to detect the effects of management actions that could affect outcomes beyond the ecological unit to which the management action is applied. This study provides empirical support for how collaborative approaches to governance can address the problem of fit, but also reveals that collaborative approaches do not necessarily solve all challenges associated with social-ecological fit. Through this approach I highlight that integrating social with ecological information can lead to more accurate assessments of the problem of fit compared to an approach that considers the social and ecological system in isolation.

In Chapter 5 I further explore the value of integrating social with ecological information when making and implementing conservation decisions (Chapter 5, submitted to Biological Conservation). I explore how a social-ecological system framework can be utilised to facilitate the systematic consideration and integration of relevant ecological and social data to determine areas of conservation opportunity. I show how this approach can be used to identify priority areas that require different implementation strategies, from areas that are suitable for immediate engagement to areas requiring implementation over the longer term in order to increase on-the-ground capacity and identify mechanisms to incentivise implementation. This study highlights the value of applying a social-ecological framework to conservation planning, helping translate priorities for action into implementation strategies that account for the social-ecological complexity of conservation problems.

This thesis addresses one of the greatest challenges faced by conservation researchers and practitioners: understanding and accounting for the social-ecological complexity that characterises most global environmental problems. This thesis makes theoretical and empirical contributions to research on the *problem of fit* that extend beyond the conservation planning field. It provides empirical support for how collaboration approaches to governance can enable the coordination of actions across different management scales, and demonstrates how interactions between the social and ecological systems can be accounted for in conservation planning decisions, and in assessments of the effectiveness of environmental governance arrangements.

Declaration by author

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

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

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Publications during Candidature

Guerrero, A. M., R. R. J. McAllister & K.A Wilson (2015). "Achieving cross-scale collaboration for large scale conservation initiatives" *Conservation Lette*rs 8(2): 107-117

McAllister, R. R. J., C.J. Robinson, K. MacLean, **A.M. Guerrero**; K. Collins, B.M Taylor, P. De Barro. (2015). "From local to central: a network analysis of who manages plant pest and disease outbreaks across scales." Ecology and Society 20, http://www.ecologyandsociety.org/vol20/iss1/art67/.

Mills, M., J. G. Álvarez-Romero, K. Vance-Borland, P. Cohen, R.L Pressey, **A.M. Guerrero**, H. Ernstson (2014). "Linking regional planning and local action: Towards using social network analysis in systematic conservation planning." *Biological Conservation* 169(0): 6-13.

Guerrero, A. M., R. R. J. McAllister, J. Corcoran & K.A. Wilson (2013) "Scale Mismatches, Conservation Planning, and the Value of Social-Network Analyses" *Conservation Biology* 27(1): 35-44.

Papers in review

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Publications included in this thesis

Four published or submitted papers are reproduced in entirety as chapters forming part of this thesis; my contributions to them are as follows:

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Contributor	*Statement of contribution*
A. M. Guerrero	Original idea Literature review (100%) Written work (70%)
R. R. J. McAllister	Discussion and development of idea Wrote and edited paper (15%)
J. Corcoran	Discussion and development of idea Commented on earlier drafts of the written material
K.A. Wilson	Discussion and development of idea Wrote and edited paper (15%)

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A. M. Guerrero	Original idea Research design (100%) Data collection (100%) Data analysis (70%) Wrote the paper (80%)
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K.A. Wilson	Discussion and development of idea Wrote and edited paper (30%)

Contributions by others to the thesis

Associate Professor Kerrie Wilson (UQ) and Dr Ryan McAllister (CSIRO) have made significant contributions to the conception and design of the project. Blythe Spraggins has undertaken data collection work in the quantitative stage of the research study. Hawthorne Beyer (UQ) provided technical support for the ecological connectivity analysis (Chapters 4 and 5). Dr Ryan McAllister made substantial input into the analysis and interpretation of the research data, and Dr Kerrie Wilson has critically revised the thesis.

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

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Key words

collaborative governance, conservation planning, conservation implementation, conservation opportunity, environmental governance, exponential random graph modelling, scale mismatch, social-ecological fit , social-ecological networks, social-ecological system

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List of Abbreviations used in this Thesis

ERGMs	Exponential Random Graph Models
GS	Governance System
MLERGMs	Multilevel Exponential Random Graph Models
MSE	Management Strategy Evaluation Framework
PPD	Press-Pulse Dynamics Framework
RS	Resource System
RU	Resource Services and Units
SES	Social-Ecological Systems framework
SNA	Social Network Analysis

Chapter 1. Introduction

The conservation crisis is well known. The needs for nature conservation far outstrip current efforts while underlying pressures on the environment, including demographic growth, human consumption and climate change, continue to intensify (Millennium Ecosystem Assessment, Butchart et al., 2010; Rands et al., 2010). Only 15% of all threatened birds, amphibians and mammals, and 36% of ecoregions are adequately represented in protected areas (CBD, 2011; Venter et al., 2014; Watson et al., 2014). In addition, 40% of all protected areas have been found to have major management deficiencies (Leverington et al., 2010). Moreover social and economic pursuits (e.g. housing development, natural resource extraction) are leading to the downgrading, downsizing and degazettement of protected areas (Mascia et al., 2014). Poor governance is considered a key aspect influencing the effectiveness of protected areas to contribute to nature conservation goals (Borrini-Feyerabend et al., 2012).

The challenges associated with achieving good environmental governance are not limited to protected areas. Approaches to nature conservation have evolved from the formulation of single solutions focusing on protected areas and involving localised actions, to solutions that focus on addressing multiple threats and involving diverse management actions in both protected areas and human-dominated landscapes (e.g. Fitzsimons et al., 2013b; Pressey et al., 2007; Wilson et al., 2010; Wilson et al., 2007). This has led to the involvement of multiple stakeholders ranging from local resource users to global actors, each with diverse knowledge systems, objectives, and views. This diversity of stakeholders is both supported and constrained by governance arrangements and decision-making processes for which social considerations are as important as ecological ones (Armitage et al., 2012; Sabatier, 2005; Smith et al., 2009).

Despite these challenges, support for multi-stakeholder conservation initiatives continues to increase and successes are being reported (Fitzsimons et al., 2013b). There are however also examples of how the governance challenges associated with managing complex social-ecological systems have led to disappointing results (e.g. Wyborn, 2014). These governance challenges are often the result of complex interactions between social and ecological systems.

It is increasingly recognised that conservation efforts grounded on knowledge of both social and ecological systems, and the way they interact, are more likely to be effective

(Cowling and Wilhelm-Rechmann, 2007; Rapport et al., 1998; Robinson, 2006; Sutherland et al., 2009). For instance, by incorporating socioeconomic considerations into analyses of conservation problems, and by accounting for how economic, cultural and institutional factors affect how people engage with the natural world, the solutions posed can be more accessible and relevant to society. This has been shown to lead to greater adoption of management recommendations and lead to policy that is more likely to be supported by the public (Robinson, 2006). An example is provided by the fishing industry where rather than focusing on restricting harvest to achieve conservation goals (e.g. based on ecological equilibrium models), an understanding of *how* people harvest resources can lead to practical recommendations on ways to regulate harvesting, and in this way make harvesting limits achievable (e.g. Berkes et al., 2006). Despite recognition of the importance of the social sciences to conservation, from over 25,000 manuscripts on the general topic of biodiversity conservation, only around 2000 of these studies make explicit reference to social dimensions, and 500 to institutional dimensions (Web of Science, accessed 12th Sep 2014).

This thesis focusses on the complex linkages between social and ecological systems. The overarching research question this thesis addresses is *how can the linkages between social and ecological systems, and the challenges these linkages pose, be accounted for in conservation planning?* In this chapter I provide the background motivation for this body of research. I introduce the social-ecological context of conservation planning problems. I introduce social and ecological factors that can help understand the complex ways in which social and ecological systems interact, and discuss how they have been considered in conservation studies. I then explore the challenges the social-ecological context poses to conservation efforts, and the approaches proposed for addressing these challenges. I focus on challenges related to the *problem of fit* - when governance systems do not fit or match the characteristics of the biophysical system. I find that 'fit challenges' and their consequences are poorly understood in the context of biodiversity conservation and methods for their analysis are underdeveloped.

In this thesis I address this gap by researching the *problem of fit* in the context of biodiversity conservation, and demonstrating how novel methodological approaches and analytical frameworks adopted from the natural resource and social sciences fields can be used for improving the way the social-ecological context is taken into account when planning and implementing conservation actions. This thesis adopts an interdisciplinary perspective, integrating concepts and methods from diverse fields in the natural and social sciences, including ecology, conservation planning, natural resource management, policy studies, and social network research. I conclude this chapter with an overview of the aims and structure of the thesis.

1.1 Conservation planning and its social-ecological context

Biodiversity conservation is concerned with the long term viability of species, ecosystems and evolutionary processes (Soulé, 1985), and increasingly being related to the improvement of human well-being (Kareiva and Marvier, 2012). Conservation planning is an important mechanism used to make decisions about how best to respond to threats affecting biodiversity decline. Systematic conservation planning is a framework used to identify and design priority actions (e.g. species, areas) in time and space, to maximise the outcomes achieved with limited financial resources (Margules and Pressey, 2000; Pressey and Bottrill, 2009).

Biodiversity conservation efforts occur as part of complex social-ecological systems. A social-ecological system is a complex and perpetually dynamic system defined by several spatial, temporal, and organisational scales (Redman et al., 2004). In these systems humans and ecosystems interact creating dynamic feedback loops in which humans both influence and are influenced by ecosystem processes (Berkes and Folke, 1998; Gunderson and Holling, 2002; Liu et al., 2007). It is widely accepted that social and ecological factors need to be integrated in conservation planning assessments (Cowling and Wilhelm-Rechmann, 2007; Knight et al., 2006b). This is reflected in an increasing number of conservation planning studies that incorporate social factors, although the majority of studies focus on social data related to threats or costs (e.g. Naidoo et al., 2006; Wilson et al., 2007; Table 1.1). It is attention to factors such as values and the behaviours of individuals, and governance aspects that will help understand the social-ecological complexity and help translate priority actions into conservation outcomes (Cowling and Wilhelm-Rechmann, 2007). However, exploration of these aspects to date has been limited. Governance aspects such as the level of political stability, regulatory provisions, and decision-making processes can be crucial for the effective implementation of conservation plans (e.g. Cowling and Wilhelm-Rechmann, 2007; McCreless et al., 2013; Sutherland et al., 2009; Wilson et al., 2011). The integration of social and ecological factors in conservation planning analyses is the key focus of this thesis. Particular attention is given to understanding key governance challenges from a social-ecological perspective.

The natural resource management and social sciences offer rich literature and tools that can aid conservation science researchers and practitioners understand the social-ecological context and how it can affect conservation outcomes (e.g. Aswani and Aswani, 2010; Ban et al., 2013; Berkes, 2007; Bodin and Crona, 2009; Cowling and Wilhelm-Rechmann, 2007; Prell et al., 2009; Redman et al., 2004). In the paragraphs that follow I briefly discuss some of the social and ecological factors that can help understand the social-ecological context of biodiversity conservation efforts.

Table 1.1. Social factors considered in systematic conservation planning

Social factor	***Example***	***References***
Attitudinal and behavioural factors	Willingness to participate/to sell land	Adams et al., 2014; Guerrero et al., 2010; Knight et al., 2011b
	Farmer uptake of conservation schemes	Dutton et al., 2008
	Landholder attitudes towards revegetation activities and remnant vegetation	Jellinek et al., 2014; Raymond and Brown 2011
	Public behaviour affecting conservation outcomes	Ng et al., 2014
Social capital	Social connectivity measures	Mills et al., 2014
Collaboration	Multiple stakeholders/ objectives	Bode et al., 2010; Bryan et al., 2010a
	Economic trade agreements	Levin et al., 2013
Social values	The social value assigned to ecosystems	Bryan et al., 2011; Bryan et al., 2010b; Whitehead et al., 2014
	Multiple stakeholder interests	Klein et al., 2008
	Community support	Game et al., 2010
	Social equity	Halpern et al., 2013

Social factor	*Example*	*References*
Financial	Cost of land	Ando et al., 1998; Polasky et al., 2001
	Management costs	Balmford et al., 2000; Moore et al., 2004
	Multiple conservation costs	Naidoo et al., 2006
	Return on Investment	Murdoch et al., 2007; Tear et al., 2014; Wilson et al., 2007
	Uncertainty of land availability / property costs /	Carwardine et al., 2010; McBride et al., 2007; McDonald-Madden et al., 2008
	Investment uncertainty	Carwardine et al., 2008; Faith et al., 1996; Stewart and
	Opportunity costs (socio-economic interests)	Possingham, 2005; Venter et al., 2013; Williams et al., 2003
Anthropogenic threats	Population & agriculture pressure	Ceballos et al., 2005 Wilson et al., 2005 Wilson et al., 2006
	Land conversion	Wilson et al., 2010
	Land use / degradation	Wilson et al., 2007
	Climate change	Faleiro et al., 2013
	Risk of habitat loss	Wilson et al., 2011
Governance factors	Corruption	Eklund et al., 2011
	Legislative effectiveness	Wilson et al., 2011
	Implementation feasibility	Sewall et al., 2011

1.1.1 Social factors influencing conservation outcomes beyond costs and threats

There are several social factors that influence the success of conservation efforts that previously have not been captured in conservation planning assessments. Factors related to *human behaviour* influence the decisions individuals make and therefore the impact they have on conservation outcomes. While some factors related to human behaviour have been captured in conservation planning assessments (Table 1.1) there are several that remain unexplored. These include cognitive, psychological and institutional factors, such as values and attitudes towards conservation, past behaviour, the effects of social influence, institutional and project design, and conflicting value systems due to a diversity of preferences and perceptions (Abrahamse and Steg, 2013; Barr and Gilg, 2007; Beratan, 2007; Brooks et al., 2012; Cinner et al., 2014; Dovers, 2001; Fischer et al., 2012; Ives and Kendal, 2014; Milner-Gulland, 2012; Ostrom, 1990; Pollnac et al., 2010; Rustagi et al., 2010; Schirmer et al., 2012; St John et al., 2010).

Economic and political factors also influence the success of conservation efforts. For example, scarce funds for conservation might be available because of competing societal priorities, decisions made at different jurisdictional scales might be in conflict (Cash et al., 2006), conservation outcomes might be strongly influenced by market access (Cinner et al., 2012), or implementation might be affected by poor governance (Smith et al., 2003).

In addition, diverse *social processes* can be critical for the formulation of effective responses and their implementation. For example, learning processes can facilitate adaptation of responses when the effects of conservation and management actions are uncertain (Berkes et al., 2003; Davidson-Hunt, 2006; Folke et al., 2005; Gunderson and Holling, 2002), and collaboration processes that can lead to the coordination of actions across jurisdictional or governance boundaries (Aswani et al., 2013; Carlsson and Berkes, 2005; Olsson et al., 2007; Sabatier, 2005; Wondolleck, 2000; Wyborn and Bixler, 2013).

Knowledge on how this diversity of social factors influences the success of conservation efforts is scarce. While some of these factors are captured in this thesis (i.e. collaboration processes, governance aspects), the focus is on how these factors can be integrated with ecological factors in conservation planning analyses.

1.1.2 Ecological factors needing attention in conservation and management studies

Devising effective responses is also compounded by the complexity and dynamics of the ecological system being managed. Most conservation planning assessments involve representation of species diversity patterns, but relatively few consider ecological processes or dynamic threats to biological diversity (Pressey and Bottrill, 2009; Pressey et al., 2007). For example, a key aspect gaining increased attention in natural resource management studies is the importance of accounting for the *interconnected nature of ecological systems* (Bodin and Tengo, 2012; Cumming et al., 2010; Galaz et al., 2008; Janssen et al., 2006). In ecological systems elements are connected to one another through diverse interactions such as predation, pollination or nutrient cycles. The survival of a species might depend on a critical level of connectivity between ecological units (e.g. habitat patches). In addition the dispersal ability of disease and invasive species is influenced by the connectivity between different land parcels or vegetation patches. Making decisions about how best to manage ecological systems therefore requires an understanding of how elements of the ecological systems are connected to one another (e.g. Chades et al., 2011). Management and conservation efforts that do not account for these connections unlikely to be able to respond to spreading ecological changes (e.g. dispersal of an invasive species) and cascading effects of actions (e.g. the effects of farming on marine ecosystems through nutrient loading; Crowder et al., 2006).

Another key consideration is *scale*. Ecological processes operate at diverse scales, so the processes observed depend on the spatial and temporal scale of observation (Levin, 1992; Wiens, 1989). This introduces uncertainty and makes it difficult to respond to changes and formulate management actions at appropriate scales that will be effective through time (Levin, 1998). Management at inappropriate scales can lead to increased vulnerability, adverse effects and inefficiencies (e.g. Wilson, 2006).

1.1.3 Social and ecological systems are coupled and interdependent

The interdependencies that exist between social and ecological systems can also affect the effectiveness of conservation efforts. Humans interact with elements of the ecological system in a continual interchange of inputs and outputs (Berkes et al., 2003; Gunderson and Holling, 2002; Redman et al., 2004). Inputs can relate to land and resource use and conservation and management actions, while outputs can include harvest, cultural, biodiversity outcomes and those related to ecosystem services. The interactions between social and ecological systems can result in positive or negative feedback loops (Cinner 2011). For example, appropriate governance systems can facilitate conservation and management actions (e.g. habitat restoration or invasive species control) that effectively respond to threatening processes and changes in the ecological system. On the other hand, the common-property nature of many natural resources can drive resource exploitation in the absence of appropriate and functioning governance systems (Berkes and Folke, 1998; Hardin, 1968; Lee, 1993; Ludwig et al., 1993; Ostrom, 1990; Ostrom et al., 1999). When social and ecological systems are strongly interdependent reciprocal interactions can push social-ecological systems towards increased vulnerability and a loss of resilience (Anderies et al., 2006; Galaz et al., 2008; Liu et al., 2007; Rockstrom et al., 2009). Conservation decisions that do not account for social-ecological interdependencies might fail to detect important feedback loops and thus fail to adapt and respond to changes in the ecological system.

There are diverse frameworks that have been developed in the natural resource management field aimed at capturing the connections between social and ecological systems. These include the Social-ecological Systems framework (SES), the Press–Pulse Dynamics framework (PPD), and management strategy evaluation framework (MSE) (Bunnefeld et al., 2011; Collins et al., 2010; Ostrom, 2007). These frameworks differ in terms of their purpose and their conceptualisations of the social and ecological systems and their dynamics (Binder et al., 2013). The SES framework was developed as a diagnostic approach designed to aid the identification of common drivers of sustainability outcomes in natural resource systems (McGinnis and Ostrom, 2014; Ostrom, 2009); the PPD framework was developed to guide integrated social and ecological research (Collins et al., 2010); and the MSE framework is a simulation approach used in fisheries for testing management options under a range of uncertainties (André and Greg, 2007). The extension of some of these frameworks to conservation is currently being investigated (Ban et al., 2013; Bunnefeld et al., 2011; Milner-Gulland et al., 2010; Nuno et al., 2014). The application of a social-ecological framework could see conservation planning evolve in an important direction, aiding understanding of key social-ecological interdependencies that can affect conservation efforts, and informing the development of implementation strategies that can account for them.

1.1.4 Planning for implementation

Implementation strategy development was recently recognised as a step in the conservation planning framework (Pressey and Bottrill, 2009), but it is an area requiring greater attention (Driver et al., 2003; Knight et al., 2008; Pressey and Bottrill, 2008). While general operational models for implementing conservation action have been proposed (e.g. Knight et al., 2006a), little attention has been given to how social data can be utilised to inform implementation strategies. Attempts have focused on identifying where the values of the community align (or otherwise) with scientifically defined ecological values (e.g. Bryan et al., 2011; Whitehead et al., 2014). Application of a social-ecological framework could guide and improve the integration of social with ecological data in conservation planning analyses to analyse and understand how social-ecological connections affect conservation efforts. This knowledge could then be use to inform the development of an implementation strategy that accounts for the complexity of the social-ecological system.

1.2 The role of governance in conservation

Central to responding to the challenges presented by the complex character of social-ecological systems are governance systems. Science can inform what to protect, where to act and what actions to implement (Costello and Polasky, 2004; Wilson et al., 2006), but without effective governance systems that respond to the characteristics of ecological and social systems, and their interactions, the effectiveness of conservation and management efforts will be reduced (Crowder et al., 2006; Ostrom, 2010b).

Different types of governance shape and influence conservation and management responses and play a key role in translating them to conservation outcomes. Governance can be defined as ‘the process of guiding societies towards outcomes that are socially beneficial and away from outcomes that are harmful’ (Young et al., 2008). An important component of governance systems are institutions – the norms, rules, rights and decision making processes designed to drive change, and influence the way groups and individuals interact with, and mediate the effects they have on the natural system (Berkes and Folke, 1998; Ostrom, 1990; Young, 2002). This include state-based regimes such as those created for the conservation of biodiversity, climate change regimes, and legally binding agreements negotiated between governments (Newell et al., 2012).

Other types of governance include multi-actor approaches involving a diversity of non-state actors who produce, enact and implement environmental governance (Newell et al., 2012), including public, and private actors acting individually and as collectives through diverse governance arrangements (e.g. partnerships, networks, customary management institutions; Carlsson and Sandstrom, 2008; Cinner et al., 2011; Lauber et al., 2011). In multi-actor approaches to governance, public authority and responsibility is shared (e.g. public-private partnerships) or delegated (e.g. market mechanisms; Green, 2008). Multi-stakeholder approaches to governance constitute attempts to build and improve upon the limits of central government responses to environmental threats, and can develop informally around specific issues to form more or less defined coalitions of common interests and beliefs (Lubell et al., 2002; Newell et al., 2012; Schneider et al., 2003; Young et al., 2008). Examples of multi-stakeholder governance approaches in conservation settings include large-scale conservation initiatives such as Yellowstone to Yukon (Y2Y) in North America, and The Great Eastern Ranges in Australia. Such initiatives are often characterised by multiple land tenures and jurisdictions, heterogeneous land uses and land covers, and numerous stakeholders involved in diverse activities that span multiple ecological and management scales (Fitzsimons et al., 2013b; Worboys et al., 2010).

1.3 Governance challenges: The problem of fit

Devising effective governance approaches to conservation and environmental management problems has been an important area of focus of interdisciplinary research in the last few decades (Ostrom, 2010a; Rands et al., 2010). A key challenge associated with collaborative governance is the degree to which governance systems ‘fit’ or match the characteristics of the biophysical system (Armitage et al., 2012; Brown, 2003; Folke et al., 2007; Galaz et al., 2008; Ostrom, 2010b; Young, 2002; Young et al., 2008). The *problem of fit* is increasingly being explored in the natural resource management and ecosystem-based management literature, but interpretations of the concept of ‘fit’ vary and its definition remains unclear (Vatn and Vedeld, 2012). In this thesis I use ‘problem of fit’ as an overarching concept of the challenges governance systems face in relation to characteristics of the biophysical system being managed. One aspect is the spatial scale at which ecological processes occur. A ‘fit challenge’ is faced when for example the management of fisheries is too broad to recognise ecosystem processes operating at finer scales, or is limited to jurisdictional boundaries (Crowder et al., 2006; Holling and Meffe, 1996; Wilson, 2006). Another fit challenge arises when governance responses to conservation problems ignore how the dynamics or spatial organisation of ecological units (e.g. areas of vegetation or key habitat, or dispersal of species) facilitates key ecological processes (Berkes, 2006; Coutts et al., 2013; Esler et al., 2010). Such fit challenges are common and difficult to respond to, can lead to ineffective management, and affect the capacity of governance responses to deal with unforeseen changes (Cumming et al., 2006; Ekstrom and Young, 2009; Galaz et al., 2008; Young, 2003; Young et al., 2008).

Biodiversity conservation problems are prone to diverse fit challenges, but these are poorly understood and have not been explicitly analysed in the conservation planning literature (Mills et al., 2010; Pelosi et al., 2010; Rouget, 2003; Sarkar et al., 2006). Fit challenges are common in the conservation of migratory species (e.g. Berkes, 2006), wildlife connectivity conservation (e.g. Wyborn, 2014), and in the management of invasive species characterised by uncertainty and rapid cumulative changes (e.g. Coutts et al., 2013). An improved understanding of fit challenges and how they influence the effectiveness of conservation efforts would benefit conservation professionals, by aiding their timely recognition when planning for conservation actions and the development of strategies to address them.

1.4 Collaborative governance: a solution to the problem of fit?

An increased attention to fit challenges in the conservation and management of ecological systems has resulted in a shift from state-centric approaches to collaborative governance approaches that can enable adaptive management across multiple scales (Armitage et al., 2009; Brondizio et al., 2009; Carlsson and Berkes, 2005; Folke et al., 2005; Lemos and Agrawal, 2006; Olsson et al., 2004; Ostrom, 1998, 2010b). Collaborative governance approaches have been proposed to promote learning and flexibility to enable actors to respond to change in the face of complexity and uncertainty (Armitage, 2007; Gunderson and Light, 2006). Collaborative governance can facilitate the coordination of distinct sources of governance acting at different levels (local, regional, supra-regional). Collaboration can bring together actors with diverse interests, knowledge, and resources, and the commitment needed to understand the conservation problem, and support information sharing, learning and the implementation of actions, across different ecological and management scales (Armitage et al., 2007; Armitage et al., 2009; Fazey et al., 2007; Folke et al., 2005; Huntjens et al., 2012; Imperial, 1999; Olsson et al., 2007; Ostrom, 1961; Young et al., 2008). Collaborative approaches that enable cross-scale linkages between stakeholders are thought to increase the capacity of governance systems to deal with fit challenges, thereby enabling the learning and adaptation needed to deal with the complexity that characterise social-ecological systems (Gunderson and Holling, 2002; Holling, 1978; Walters and Holling, 1990).

Collaborative governance approaches may have the potential to address fit and mismatch challenges. But, they come at a cost, and their benefits are not easily realised. Initiating, and ensuring the longevity and sustainability of collaborative relationships can be a lengthy process that can be impacted by budget limitations and funding cycles (Fitzsimons et al., 2013b; Wyborn, 2014). It can also be negatively impacted by the dominance of individuals or organisations, obstructing disagreements between partners (Young, 2006), and actors becoming distracted by complex collaborative arrangements with numerous other actors (Lazer and Friedman 2007). Also, engaging in certain types of collaborative interactions might not lead to sustained collaboration (e.g. exchange of technical information; Berardo, 2010). In addition, collaborative approaches can become trapped in complex governance arrangements, which can undermine their capacity to connect actors across scales of decision making and thus fail to support the necessary functions of coordination across multiple-scales (e.g. Wyborn, 2014). Simply prescribing formal collaborative governance arrangements will not necessarily overcome these barriers and translate into more effective collaboration that enables governance across scales (Carr, 2013; Lubell, 2004). Strategic approaches to the formation and support of effective collaborative governance arrangements are needed.

1.5 A network perspective

A focus on the ways actors interact in a conservation setting can provide insights on the effectiveness of different collaborative governance arrangements. The interactions between actors within a governance arrangement can be characterised as a social network and analysed to assess the potential for formulating and implementing conservation actions at the required scales. Social network theory is used to analyse the behaviour of individuals, groups, and organisations on the basis of its structure (i.e. patterns of relations; Emirbayer and Goodwin, 1994). The application of network theory to the study of collaborative processes and structures has been applied in diverse disciplines, including in the social sciences (Borgatti et al., 2009), natural resource management and conservation (Bodin and Crona, 2009; Bodin and Prell, 2011), sustainability science (Henry and Vollan, 2014) and policy studies (e.g. Berardo and Scholz, 2010; Sandstrom and Carlsson, 2008). Typically, applications of social network theory in conservation link the structural characteristics of the whole of network to theory about the social processes that underpin effective conservation governance (e.g. learning and innovation). Such characteristics can be described by network centrality, cohesion and density metrics (e.g. Cohen et al., 2012; Isaac et al., 2007). Fewer studies have analysed whole networks by examining minimal network structures, or 'building blocks', exhibited by stakeholders who interact within the network. Through this approach, specific patterns of social and social-ecological interdependencies, characterised by particular building blocks, can be theoretically linked to specific governance challenges (Bodin and Tengo, 2012). The 'building block' approach combined with new statistical models for multi-level networks (Wang et al., 2013), can be used to empirically analyse social and ecological data to assess diverse governance challenges related to the management of social-ecological systems, including fit challenges (e.g. Bodin et al., 2014; Bodin et al., in review).

1.6 Aim, scope and outline of the thesis

In summary, while the importance of the social-ecological context in which conservation actions are implemented is increasingly recognised in the conservation planning literature, better integration of social data is still needed. Furthermore, little attention has been given to utilising knowledge of the social-ecological system to inform approaches and strategies that can lead to effective implementation of actions. In a social-ecological system multiple objectives compete, solutions that cross jurisdictional boundaries are required, and social and ecological factors interact at multiple scales. This complexity makes it difficult to devise effective responses to reverse the rate of biodiversity loss and environmental degradation. Fit challenges can negatively affect conservation initiatives, but have not been explicitly addressed, and are rarely considered by conservation professionals (Mills et al., 2010). The theory and concepts behind how collaborative approaches to governance have the potential to deal with fit challenges has been well developed but empirical explorations of this potential are rare (Galaz et al., 2008; Wyborn, 2014).

In this doctoral thesis I attempt to address this gap. I demonstrate how the social-ecological system can be considered to increase the effectiveness of collaborative multi-stakeholder approaches to conservation, and to inform conservation planning decisions.

The overall research question and aims of this thesis, and the publications resulting from each chapter, are summarised in Table 1.2.

In Chapter 2 I explore the fit challenges faced by the science of conservation planning. I focus on challenges related to scale mismatch, explaining how they manifest and are associated with the different stages of conservation planning. I explore how social network approaches could be used to analyse collaborative structural patterns in conservation initiatives, in order to understand the degree to which plans and activities can be coordinated across scales. This chapter reveals the need for conservation planners and practitioners to account for fit challenges by encouraging cross-scale collaboration. This can lead to planning decisions that improve the fit between management actions and conservation problems.

In chapter 3 I explore how stakeholders interact within large-scale collaborative arrangements. Characterising the interactions between stakeholders of a large-scale conservation initiative as a social network, I statistically explore the different forms of stakeholder interaction for different types of activities, to determine the propensity of the network to facilitate collaboration across scales. This chapter reveals options for the conservation initiative to improve cross-scale collaboration, and offers an approach for assessing the capacity of collaborative governance arrangements to address fit challenges related to scale.

Building on the previous chapter, in Chapter 4 I apply a social-ecological network perspective to analysing the capacity of governance arrangements to deal with different types of fit challenges. Ecological data of the same large-scale conservation initiative is combined with the social network data to characterise social-ecological interactions as a multi-level social-ecological network. The connectivity of different vegetation patches is characterised as an ecological network. The interactions between the ecological network and the social network characterise the focus that stakeholders have on different vegetation patches. Recent theoretical and methodological developments that permit the empirical analysis of social and ecological data are applied to determine the propensity for the structure of stakeholder interactions to address fit challenges related to scale, to the management of common areas, and to the management of interconnected ecological units. Focus is placed on specific network configurations, or ‘building blocks’, that can be associated to these types of fit challenges. These building blocks are statistically tested to evaluate how well they are represented in the multi-level social-ecological network data. This study reveals the type of fit challenges that the conservation initiative has the capacity to address and demonstrates a research approach to assessing the problem of fit. This study provides support to the proposition that collaborative approaches can increase the capacity of governance systems to address the problem of fit, but also reveals that collaborative approaches may not necessarily solve all challenges associated with social-ecological fit.

The conceptualisation of social-ecological systems as networks of interactions, together with the application of statistical network methodologies is an innovative approach in the conservation and natural resource management fields. To my knowledge this is the first empirical study demonstrating the benefits of social-ecological approach to analysing governance challenges related to the conservation and management of natural resources.

Chapter 5 shows how a social-ecological system framework can be utilised to guide the integration of social and ecological factors into analyses to help identify conservation priorities and strategies for their implementation. The science of conservation planning lacks explicit methods to analyse social-ecological systems for informing conservation planning and management decisions (Knight et al., 2011a; Pierce et al., 2005). Empirical social-ecological analyses are rare in the conservation planning literature (but see Mills et al., 2013). Utilising this thesis' case study data, this chapter demonstrates how the systematic consideration and integration of ecological and social data can help translate priorities for action into implementation strategies that respond to the social-ecological complexities surrounding conservation problems.

Table 1.2. Thesis summary and publications arising from each chapter

Overarching research question: *How can linkages between social and ecological systems, and the challenges they pose, be accounted for in conservation planning?*		
Chapter	**Aim**	**Publications**
Chapter 1 - Introduction	To introduce the social-ecological context of conservation	
Chapter 2. Fit challenges in conservation planning	To examine the relevance of the problem of fit to biodiversity conservation, and identify the fit challenges that can negatively affect conservation initiatives.	Guerrero, A. M., R. R. J. McAllister, J. Corcoran & K.A. Wilson (2013) "Scale Mismatches, Conservation Planning, and the Value of Social-Network Analyses" *Conservation Biology* 27(1): 35-44.
Chapter 3. Cross-scale collaboration to address scale mismatch	To investigate the specific fit challenge of *spatial scale mismatch.* And assess how collaborative approaches to conservation can address it.	Guerrero, A. M., R. R. J. McAllister & K.A Wilson (2015). "Achieving cross-scale collaboration for large scale conservation initiatives" *Conservation Letters* 8(2):107-117.
Chapter 4. A social-ecological approach to analysing the problem of fit	To explore research approaches that can incorporate social and ecological data to assess the problem of fit, and to conduct an empirical investigation of the potential of collaborative approaches in achieving social-ecological fit.	Guerrero, A.M, O.Bodin, R. R. J. McAllister & K.A. Wilson. “Achieving social-ecological fit through collaborative governance”. Submitted to *Global Environmental Change*.
Chapter 5. Analysing social-ecological interactions to determine opportunities for conservation	To demonstrate how the Social-ecological Systems framework be applied to guide the integration of relevant ecological and social data for determining opportunities and challenges for conservation.	Guerrero, A. M & K.A. Wilson "Informing implementation strategies for conservation using a social-ecological systems framework”. Submitted to *Biological Conservation*.
Chapter 6. Discussion	To present conclusions and future directions	

Chapter 2. Fit challenges in conservation planning

Published as: Guerrero, A. M., McAllister, R. R. J., Corcoran, J. and Wilson, K. A. (2013), Scale mismatches, conservation planning, and the value of social-network analyses. Conservation Biology, 27: 35–44. doi: 10.1111/j.1523-1739.2012.01964.x

2.1 Abstract

Many of the challenges conservation professionals face can be framed as scale mismatches. The problem of scale mismatch occurs when the planning for and implementation of conservation actions is at a scale that does not reflect the scale of the conservation problem. The challenges in conservation planning related to scale mismatch include ecosystem or ecological process transcendence of governance boundaries; limited availability of fine-resolution data; lack of operational capacity for implementation; lack of understanding of social-ecological system components; threats to ecological diversity that operate at diverse spatial and temporal scales; mismatch between funding and the long-term nature of ecological processes; rate of action implementation that does not reflect the rate of change of the ecological system; lack of appropriate indicators for monitoring activities; and occurrence of ecological change at scales smaller or larger than the scale of implementation or monitoring. Not recognising and accounting for these challenges when planning for conservation can result in actions that do not address the multiscale nature of conservation problems and that do not achieve conservation objectives. Social networks link organisations and individuals across space and time and determine the scale of conservation actions; thus, an understanding of the social networks associated with conservation planning will help determine the potential for implementing conservation actions at the required scales. Social-network analyses can be used to explore whether these networks constrain or enable key social processes and how multiple scales of action are linked. Results of network analyses can be used to mitigate scale mismatches in assessing, planning, implementing, and monitoring conservation projects.

2.2 Introduction

Scale mismatch, also referred to as the “problem of fit,” has emerged in the literature of natural resource management and refers to a mismatch between the extent and resolution of management actions and the ecological system of interest (Cumming et al., 2006; Lee, 1993; Young, 2002). The problem of scale mismatch in conservation settings occurs when conservation actions are undertaken at a scale that does not reflect the scale(s) required to solve a particular conservation problem. For example, scale mismatches are a common problem in the management of migratory species (e.g. Berkes, 2006) and when the relatively short time horizons of planners and politicians conflict with longer-term ecological and social changes (Folke et al., 1998a). Cumming et al. (2006) explored scale mismatch in the management of natural resources and explained its causes and consequences. They highlight that scale mismatches are generated by a wide range of social, ecological, and linked social-ecological processes and conclude that how to best resolve scale mismatches remains an open question. An understanding of how scale mismatches transpire and their likely consequences would be of value to conservation professionals because it would further the development of strategies to address problems of scale.

Conservation planning is evolving from being primarily concerned with the systematic identification of protected areas (Margules and Pressey, 2000) to a process of prioritising, implementing, and managing actions for the conservation of biological diversity and other natural resources, inside and outside protected areas (Wilson et al., 2009). Effectiveness of conservation planning is hindered by a lack of funding, support for only short-term projects, lack of consideration of ecological processes and dynamic threats that determine the persistence of biological diversity (Pressey et al., 2007), limited extent to which science and research results inform on-the-ground action (Balmford and Cowling, 2006; Pressey and Bottrill, 2009), unacknowledged diversity of human value systems (Van Houtan, 2006; Wondolleck, 2000), and unrecognised, opposing or conflicting goals that obstruct objective decision making (Biggs et al., 2011). Many of these challenges emerge as a result of scale mismatches, primarily because conservation problems often require multiple actions that are each associated with different ecological and management scales (Sarkar et al., 2006). The problem of scale mismatch lies not in fitting conservation action to match a particular scale. Rather, the multiscale nature of conservation problems needs to be understood and negotiated so that strategies and actions are developed and applied at appropriate temporal and spatial scales. Governance and management arrangements that have the capacity to alleviate mismatches across the range of actions are therefore required. However, there is often insufficient institutional structures or mechanisms to adapt to the multiscale nature of conservation problems and effectively manage across scales (Folke et al., 1998b; Wyborn, 2011; Young, 2002).

Conservation planning needs to include stages dedicated to understanding the social-ecological system in which conservation actions are to be implemented, including cultural, economic, and institutional contexts (Polasky, 2008; Pressey and Bottrill, 2008), and the norms, values, and other human factors that underpin opportunities for and constraints on effective conservation (Cowling and Wilhelm-Rechmann, 2007; Guerrero et al., 2010; Knight et al., 2010). The identification and involvement of stakeholders is key to effective conservation planning. It can facilitate the identification of new knowledge and opportunities for and barriers to implementation and engender trust and support for implementation (Knight et al., 2006b; Pierce et al., 2005; Pressey and Bottrill, 2009).

Network theory has been useful for explaining social phenomena across a diversity of disciplines (Borgatti et al., 2009). Social networks link organisations and individuals across space and time and hence are critical in determining the collective scale of conservation actions, which in turn underpins the magnitude of mismatch in scale. I sought to understand challenges mismatches of scale pose to the conservation-planning process. I explored this issue across scales associated with the different stages of conservation planning. I considered emerging conservation-planning approaches that may be useful in addressing scale mismatches and how social-network analyses (SNA) can be applied to the management of scale-mismatch problems.

2.3 Scale mismatches and conservation planning

Planning and implementing conservation actions (Figure 2.1) involves definition of the conservation problem, formulation of actions, and determination of how action will be implemented. Conservation problems are often complex and involve competing objectives, multiple actors, and a diversity of possible conservation actions. Decisions can be made at spatial and temporal scales that may not match the scale of the ecological patterns or processes relevant to the conservation problem, a situation that creates a scale mismatch. For example, actions and strategies may be formulated at a regional scale but the conservation problem also requires action at a finer scale (Briggs, 2001; Sarkar et al., 2006), or a plan may be formulated at an appropriate scale for action, but the operational capacity for implementation may be lacking.

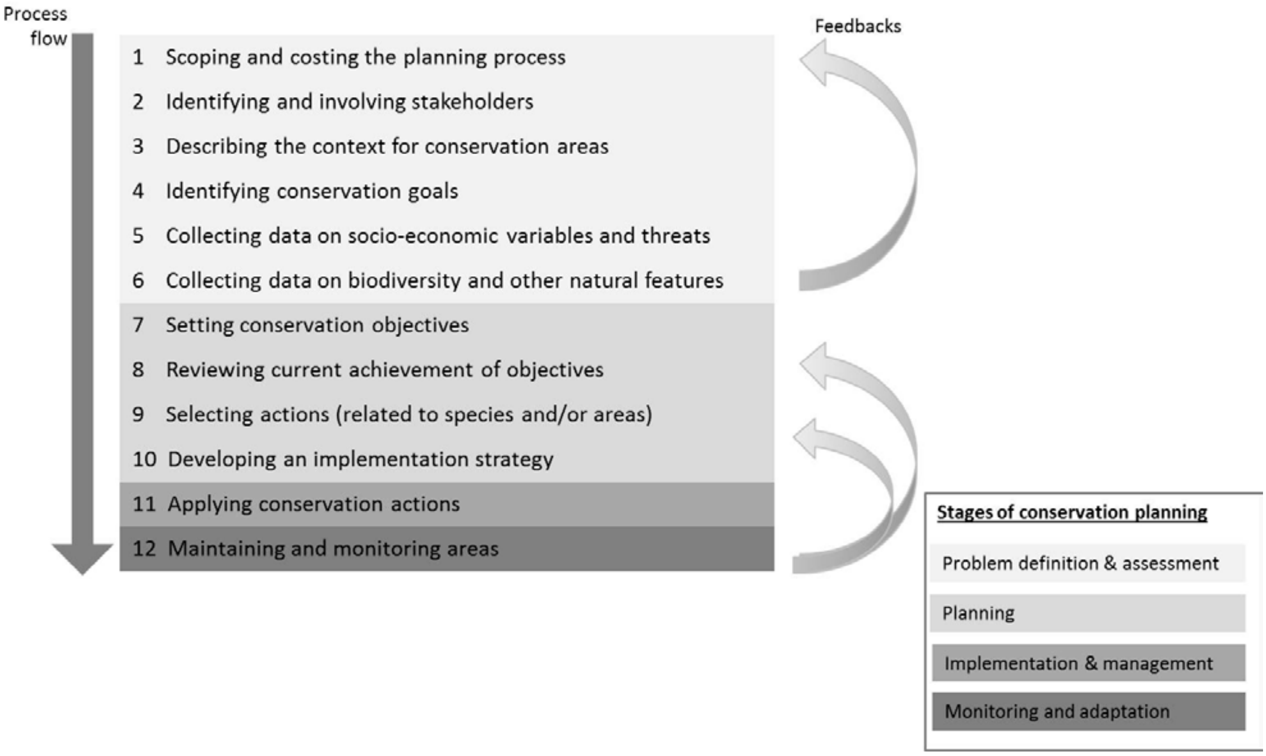

Figure 2.1. Representation of the process of conservation planning (adapted from Knight et al., [2006a] and Pressey and Bottrill [2009]).

I applied a modified version of Cumming et al.'s (2006) classification of scale mismatches (spatial, temporal, and functional scale mismatches) to show how scale mismatch manifests itself in diverse ways and at each stage in the conservation-planning process (i.e., problem assessment, strategy and action formulation, and plan implementation, evaluation, and adaptation; Table 2.1). Spatial scale mismatches occur when the geographic extent of the solution differs greatly from the geographic extent of the problem. For example, when conservation action is applied at a fine scale, such as vegetation patches, but the problem prevails at a broader scale, such as the landscape scale (Cash et al., 2006). Temporal scale mismatches relate to processes that occur over different time scales (Cash et al., 2006). Both temporal and spatial scales also have grain, which is the resolution at which observations are made (i.e., data resolution). Functional scale mismatches occur when the scope of processes considered for solving the conservation problem differs greatly from the scope of processes within the system associated with, or that affect, the conservation problem (Folke et al., 1998b; Lee, 1993). For example, a very narrow focus on a few ecological features compared with a broad focus on many ecosystem processes.

Table 2.1. Examples of how scale mismatches can manifest when planning for conservation.

	Problem assessment*	***Planning****	***Implementation and management****	***Monitoring and adaptation****
Spatial mismatch examples	The geographic extent of the planning region is not defined according to ecological boundaries but governance systems (e.g. state boundaries).	The different plans, actions, and strategies directed at the same ecosystem may be in conflict. No coordination among them may mean a lack of capacity for solving the conservation problem.	Operational scale of the implementing organisations may not be sufficient to cover the full extent of the conservation problem.	Monitoring undertaken at a scale at which involved organisations operate, which may not be representative of the full geographical extent of the conservation problem. Consequently, information for adaptation decisions can be misleading.
	The geographic resolution of data may not reflect the heterogeneity of the social-ecological system. Consequently decisions may miss important problem areas.	Actions and strategies are developed at a scale that does not reflect social-ecological system components (e.g. economic drivers, institutional barriers, cultural values), key for addressing the conservation problem, or that affect the success of conservation actions.	Implementation may not occur at an adequate scale; actions may be implemented too broadly or too narrowly to effectively address the issue.	Monitoring operations may not detect ecological changes that occur at wider or finer scales, which may limit the ability to respond and adapt to changes.

	Problem assessment*	***Planning****	***Implementation and management****	***Monitoring and adaptation****
Temporal mismatch examples	Limited data collection and quick assessments, driven by the time horizons of organisations and funding bodies, do not cover the social-ecological system in sufficient detail.	Actions are formulated for a short time horizon and do not address long-term ecosystem changes. Alternatively, actions and strategies take time to be formulated overlooking critical short-term ecosystem changes (e.g., climate change).	Actions are implemented at timeframes that do not reflect the timeframe of ecological change. Lack of continuity of personnel throughout the planning process can result in ineffective implementation of conservation actions.	Duration of monitoring activities is not enough to appropriately evaluate the effectiveness of conservation actions or is not scaled to the frequency of the event being evaluated.
Functional mismatch examples	The scope of the plan, objectives and targets, are determined restricted to the interests of funding bodies and their institutional frameworks, which can lead to a lack of accountability of features, processes, and threats to the ecological system key for addressing the conservation problem.	The actions formulated address only a limited subset of features, processes, and threats affecting the ecological system.	Actions outside of the scope of implementing organisations are not selected and result in a partly implemented plan.	Indicators chosen for monitoring activities do not provide a whole-systems view of the problem.

** Stages of project development and implementation (see Figure 2.1)*

2.3.1 Problem assessment

In conservation planning, one of the first tasks is to define the extent of the planning region. In some instances, regions are defined solely on the basis of institutional boundaries without accounting for ecological boundaries (Table 2.1). This can result in a plan that addresses only part or none of the conservation problem. For over 100 years the Murray-Darling Basin, one of the most important river systems in Australia, has provided water for irrigation, livestock, and industry, and domestic use across four Australian states. Increased water diversion fueled by the expansion of irrigation in the basin resulted in a 40% reduction in water flow (Cosler et al., 2010). As a result, ecosystems collapsed and native fishes, riparian vegetation, and wetlands of national significance have been negatively affected. Diverse but unconnected institutions (e.g., governments of different states) have attempted to repair water flow, and these efforts have led to a lack of effective governance of the basin as a whole. This is an example of a mismatch of spatial scale; the planning region did not reflect the boundaries of the ecological systems of the basin and instead encompassed areas of the basin occurring in each state. Linked to this spatial mismatch was a functional mismatch in which the full scope of features and ecological processes (e.g., patterns of river flow, condition of wetlands) occurring across the basin were not accounted for (Murray-Darling Basin Authority, 2011). More recently, attempts to manage these scale mismatches include creation of institutions operating at a federal level (e.g., Commonwealth Water Act of 2007) and formation of the Murray-Darling Basin Authority. The authority is responsible for the formulation of an integrated management plan to set water-diversion limits for the entire basin (Commonwealth Water Act of 2007) and for the development of specific conservation programs in conjunction with state governments (e.g., Rivers Environmental Restoration program). When identifying areas for conservation action, decisions about data resolution affect which and how many areas are selected (Pressey and Logan, 1995; Rouget, 2003). A spatial scale mismatch can occur when the resolution of the data used to understand the ecological and social setting fails to reflect the heterogeneity of the area (Table 2.1), which can limit the effectiveness of planning decisions (e.g. Rouget, 2003). The limited availability of fine-resolution data across a planning region and limited resources for acquiring new data (Margules et al., 2002) result in the use of coarse-resolution data (Mills et al., 2010).

Most spatial conservation-planning exercises involve representation of species diversity patterns, but relatively few consider ecological processes or dynamic threats to biological diversity (Pressey and Bottrill, 2009; Pressey et al., 2007). Lack of consideration of key ecological processes that sustain biological diversity at the assessment stage can lead to functional mismatches (i.e., failure of conservation actions prevent disruption of these processes; Pressey et al., 2007).

2.3.2 Planning: Formulation of actions and strategies

When conservation actions are not formulated at appropriate scales, the social-ecological components of the system that affect the success of conservation actions (e.g., institutional barriers, cultural practices, livelihood activities) may not be accounted for. An example of scale mismatch is when actions are formulated at a particular governance level, such as a state or county, but are applied to an ecosystem or ecological process that transcends governance boundaries. For instance, actions may be developed for species that migrate across countries but may not be developed for species that migrate within a country (e.g. Gilmore et al., 2007). In the United Kingdom, regulations on recreational use of inland waters are based on short-term behavioural responses of birds to disturbance that are averaged across sites and habitats (O'Connell et al., 2007). This generalised approach to planning does not account for site- and time-specific human disturbances and results in spatial and temporal mismatches. For example, human activity may only occur at particular times of the year or in specific locations and birds may use different lakes for different purposes (O'Connell et al., 2007).

Threats to biological diversity operate at diverse spatial and temporal scales. Therefore effective conservation planning requires the scheduling of multiple actions that can operate at these diverse scales. Some actions may need to be threat specific (Pressey et al., 2007; Salafsky et al., 2002) to address relevant ecological processes such as those associated with connectivity, population dynamics in fragments, and maintenance of patch dynamics (Carwardine et al., 2008) and thereby ameliorate the potential for mismatches of functional scale.

2.3.3 Implementation and management

The need for more effective implementation of conservation actions is recognised as a key challenge in conservation planning (Balmford and Cowling, 2006; Knight et al., 2008; Pressey and Bottrill, 2009). Many of the challenges of implementation stem from a disjointed planning process in which early stages are not integrated into a broader planning framework that focuses on implementation. This occurs, for example, when spatial prioritisation analyses do not account for implementation constraints and opportunities (Knight et al., 2008; Pierce et al., 2005) or when planning units used in the prioritisation of areas are dissimilar to areas where management will be implemented, which makes it difficult to translate plans into actions (Pierce et al., 2005).

Spatial scale mismatches in implementation lead to actions occurring at scales that do not resolve the conservation issue (Table 2.1). Spatial scale mismatch is sometimes driven by a lack of resources for implementation or occurs because key organisations or individuals were not engaged in the planning process (e.g. Waudby et al., 2007). Conservation of Australia's endangered bridled nailtail wallaby (*Onychogalea fraenata*; Commonwealth Environment Protection and Biodiversity Conservation Act of 1999) consisted of a centralised state program that was not effectively implemented at a local scale or over a long period; thus, subpopulations could not be maintained and the program failed to stop the decline of the species (Kearney et al., 2012).

Temporal scale mismatches at the implementation stage occur, for example, when funding does not match the long-term nature of ecological processes relevant to the conservation problem. This mismatch results in partially attained conservation objectives (e.g. Waudby et al., 2007). Temporal scale mismatches can also occur when actions are implemented at a rate that does not reflect the rate of change of the ecological system of interest, for example when actions are delayed due to political timeframes or in the pursuit of scientific certainty (e.g. Grantham et al., 2009).

Another temporal scale mismatch occurs when the same stakeholders cannot be involved throughout the planning and implementation processes (Pierce et al., 2005; Pressey and Bottrill, 2009; Walters, 2007). Implementation is an incremental and often lengthy process that requires the long-term participation of stakeholders so that plans can be adapted to reflect changes in ecological and social systems (e.g., changes in areas of interest, new data on threats and species diversity, changes in funding, and changes in interests of local communities; Grantham et al., 2010; Pierce et al., 2005). Consistent participation of stakeholders facilitates the adoption of plans into the regular activities of organisations responsible for planning and development (Pressey and Bottrill, 2009). Some conservation plans account for this temporal mismatch by ensuring long-term involvement of stakeholders (e.g. Green et al., 2009; Henson et al., 2009).

2.3.4 Plan monitoring and adaptation

Monitoring is key to evaluating outcomes, and it facilitates learning and adaptive management (Ferraro and Pattanayak, 2006; Field et al., 2007; Lindenmayer and Likens, 2010; Stem et al., 2005). Scale mismatches at the monitoring and adaptation stage of a plan manifest themselves when ecological changes occur at scales smaller or larger (or longer or shorter) than the scale of monitoring operations and thus go undetected (Table 2.1). Such mismatches limit ones' ability to respond to change, which can limit effectiveness of adaptive management.

To monitor conservation outcomes one must decide which ecological metrics to use, where to conduct monitoring, and the duration and frequency of monitoring (Lindenmayer and Likens, 2010; Spellerberg, 1994). These decisions can result in spatial, temporal, or functional scale mismatches. For example, choosing appropriate indicators (Carignan and Villard, 2002; Lambeck, 1997; Tulloch et al., 2011) is an uncertain process that may result in indicators that do not provide a whole-system view of the problem (Simberloff, 1998) and may not account for multiscale requirements of the species or ecological features for which the indicator is assumed to be a surrogate (Lindenmayer et al., 2002). Insufficient data, cost of monitoring, and the potential difficulties of applying the most appropriate indicator (Tulloch et al., 2011) are obstacles related to the problem of scale mismatch (Lindenmayer and Likens, 2010; Lindenmayer et al., 2002).

2.4 Addressing scale mismatch in conservation planning

2.4.1 Emerging approaches

For conservation planning to operate at diverse spatial, functional, and temporal scales, conservation practitioners need to apply tools that account for the multiscalar nature of conservation problems. Planning approaches that account for functional scale mismatches in the problem-assessment and plan-formulation stages are emerging. For example, Pressey et al. (2007) discuss approaches for planning for physical and biological processes that require management over large areas or of areas with unique topography. Such approaches include conservation areas that may change in extent and location over time, variable representation targets, and the use of specific design criteria (e.g. Briers, 2002; Leroux et al., 2007; Nicholson et al., 2006). Threats are also being considered in, for example, the scheduling of actions so that threatened areas or species are given priority and areas with non-abatable threats are avoided (e.g. Burgman et al., 2001; Game et al., 2008) and through the explicit consideration of the effects of multiple threats (e.g. Evans et al., 2011). New developments in conservation planning may address spatial and temporal mismatches inherent in more traditional planning methods, which account only for static views of the ecological, human, and social characteristics of an area. New methods balance divergent priorities at multiple spatial scales (Moilanen and Arponen, 2011) and prioritise actions over time in the face of dynamic threats, uncertainty, and changing costs of activities (Costello and Polasky, 2004; Meir et al., 2004; Wilson et al., 2006).

These new quantitative planning methods are useful for addressing scale mismatches that arise during problem assessment and action formulation stages of the planning process (Figure 2.1). During these stages species diversity and other biological data are compiled, conservation targets are set, and priority conservation areas or actions are identified (Margules and Pressey, 2000). However, scale mismatches at the implementation and monitoring and adaptation stages can still transpire. In addition, the need to embed quantitative planning methods in a social process that facilitates effective implementation is increasingly being recognised (Knight et al., 2006a; Pressey and Bottrill, 2009; Reyers et al., 2010) and acted on (Game et al., 2010; Knight et al., 2006b; Pierce et al., 2005). It is therefore timely to explore tools and approaches that can help deal with scale mismatches that impede effective implementation.

2.4.2 Social-Network Analyses

SNA may provide guidance on how implementation might be approached in the management of problems of scale mismatch. Some authors suggest integrating ecological assessments with social assessments of a region (Cowling and Wilhelm-Rechmann, 2007) to facilitate an understanding of the social-ecological effects on valued nature and of the opportunities for and constraints to implementation. Such social assessments could include an examination of the social networks that exist to determine key people affecting conservation outcomes (either through their involvement with conservation activities, or with economic, subsistent and other types of activities that have a direct effect on conservation outcomes); how the people involved are connected to each other through partnerships for action or other types of collaborations (e.g. Prell et al., 2009; Vance-Borland and Holley, 2011); and what spatial, temporal, or functional scales of operation or influence these partnerships have. Social-network theory can be used to characterise networks of collaborations and social relations and to facilitate multiscalar conservation.

For example, SNA can be used to determine the links between actors (individuals, groups, or organisations) that could be used to promote cooperation and coordination of key activities at particular and required scales of action (e.g. Gass et al., 2009).

I define conservation social networks as the networks of relationships that link actors involved in conservation activities across space. These networks are the basis of social norms and community learning; hence, they also link actors across time. Networks can be formal or informal. Informal networks are present where conservation action is to occur (e.g., a group of citizens concerned about specific issues; e.g. Newman and Dale, 2007; Vance-Borland and Holley, 2011) and take many forms, for example, farmer advice networks (e.g. Isaac et al., 2007). Formal networks (e.g. Carlsson and Sandstrom, 2008) are formed during the conservation-planning process through the establishment of formal agreements or partnerships, for example between nongovernmental organisations or government agencies, around a particular conservation objective (e.g. Bode et al., 2010). The different patterns of interactions among actors in a network give rise to different network structures (Borgatti and Foster, 2003) that can inhibit or enable social processes that are often needed in conservation planning, such as cooperation, knowledge generation, learning, and conflict resolution (e.g. Bodin and Crona, 2009; Hahn et al., 2006; Olsson et al., 2007). SNA are used to analyse the behaviour of actors in a network on the basis of its structure (i.e., pattern of relations; Emirbayer and Goodwin, 1994). For example, one can study the density of ties within a network (extent to which all actors are connected) to understand the capacity of integration and sharing of knowledge within that network (Bodin and Crona, 2009), whereas the level of fragmentation of a network (presence or lack of presence of distinct subgroups) can be useful for understanding capacity for collaboration within the network (Granovetter, 1973) and access to new knowledge (Bodin and Crona, 2009; Newman and Dale, 2007). Structural analyses of conservation social networks can help inform implementation strategies. For example, a network that is connected through a few key actors (Figure 2.2a) may indicate the best strategy is to engage with these few key actors so that they can then coordinate action through their own networks. Alternatively, a network that is highly fragmented (Figure 2.2b) may require engagement with many different actors and thus a greater financial investment at the implementation stage.

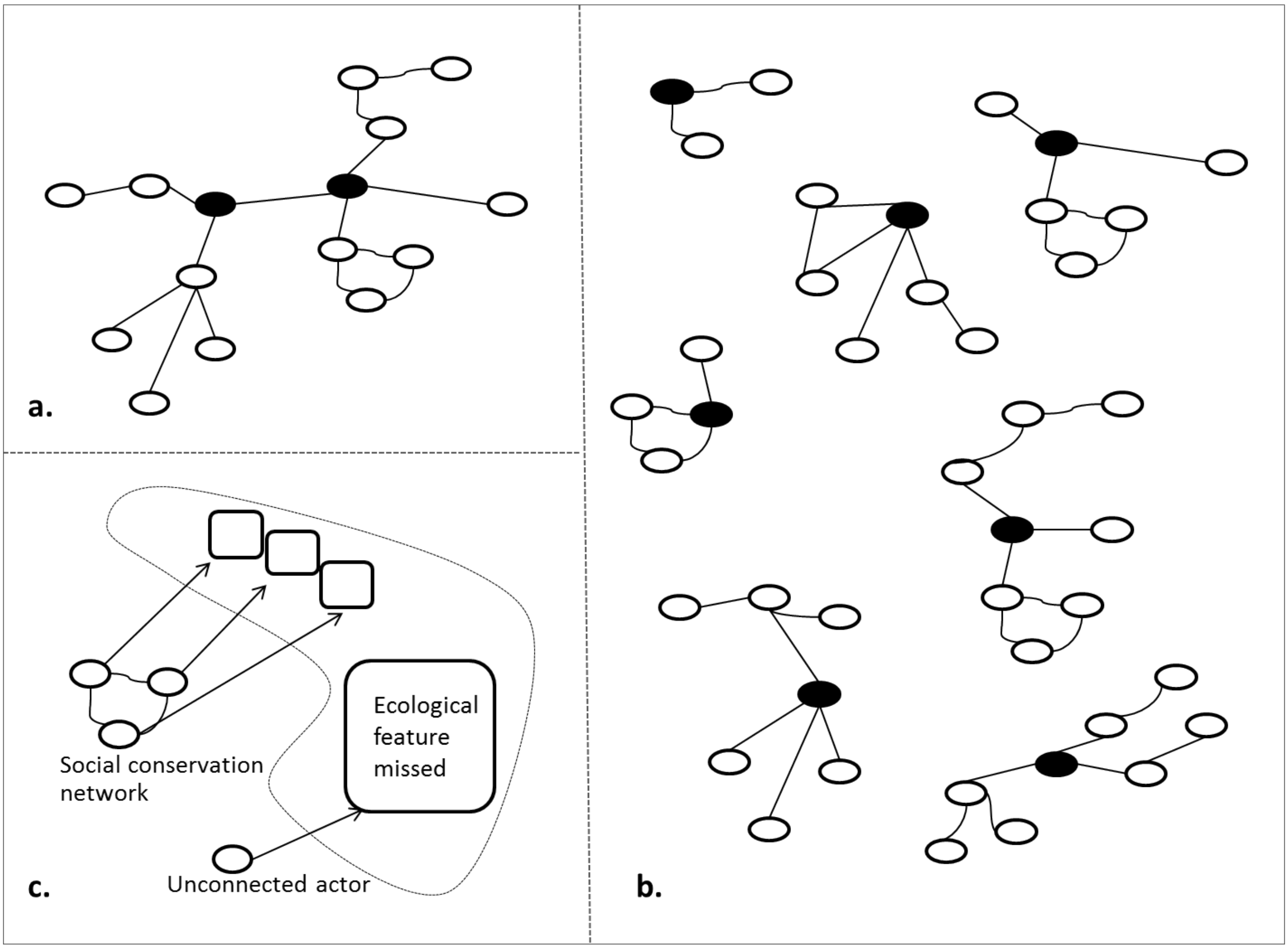

Figure 2.2. Examples of social-network structures (circles, actors [e.g., individuals, organisations]; solid circles, actors connecting the network or subgroups): (a) a network connected through a few key actors, (b) a fragmented network with an actor connecting actors in each subgroup, and (c) a network with 2 different subgroups, each involved with different ecological features (squares) of the ecosystem of interest (outlined area).

Analysing network structures can help in the understanding of the degree to which multiple scales of action are linked or being coordinated. For example, network analyses can be used to identify bridging actors (e.g. Olsson et al., 2007), or scale-crossing brokers, who link those operating at different scales who would otherwise be disconnected (Bodin et al., 2006). They can also help identify different subgroups of actors in the network that are related to particular required scales of action and thus could drive implementation at those particular scales. In work to recover the endangered Australian Glossy Black-Cockatoo (*Calyptorhynchus lathami*; Environment Protection and Biodiversity Conservation Act, 1999) a variety of agencies, community groups, landowners, and volunteers operating at different scales cooperated effectively to implement actions required for the persistence of this species (Waudby et al., 2007). Although, to my knowledge, a social-network analysis was not performed as part of this recovery plan, this example shows how identification and engagement of key groups associated with different scales of action could play a key role in the success of the project. The use of SNA to identify stakeholders allows for a targeted approach to stakeholder selection (Prell et al., 2009).

Results of SNA may be most useful when they are combined with other information about the social-ecological system. It is useful to understand not only how each actor relates to others, but also how they relate to the ecological features of interest (Fig. 2c; Janssen et al., 2006). For example, different fishers harvest different fish species at different fishing locations, and some of those species and locations will be of greater importance for achieving conservation outcomes. As well as identifying key actors who can connect to all other relevant actors—and other scales, it is also important to identify those actors connected to the most important ecological features. Such connections enable the targeting of actions to the most appropriate spatial scales.

There are other benefits of applying SNA in conservation planning. Engagement of stakeholders is an expensive process and SNA can help minimise costs by identifying either well-connected actors or actors linked to others who could prove difficult or costly to engage with directly (e.g. Prell et al., 2009). It can also be used to identify actors who could help maximise understanding of the system's complexity because of their connections to actors who hold different types of knowledge. It may also help uncover particular collaboration gaps that if addressed might connect key groups or actors who could collectively enhance conservation success (Vance-Borland & Holley, 2011).

Structural analyses of networks can provide insights into how social networks affect planned outcomes by enabling or constraining key social processes needed in the planning and implementation of conservation actions. However, acquiring a deep appreciation of the role of social networks likely requires not only an understanding of structural aspects, such as the presence or absence of links between two or more key actors or groups, but also information on the value or effectiveness of such links. For example, engaging an actor that is well connected to many other actors operating at different scales (a structural characteristic) may not be beneficial if that actor is not trusted by other actors (e.g. Gass et al., 2009), if the actor lacks legitimacy (Tyler, 2006), if the actor's presence in the network over time is uncertain (McAllister et al., 2008), or if cultural, institutional, and other contextual aspects affect the actor's willingness to participate (e.g. Bodin and Crona, 2008).

2.5 Conclusions

Strategic decisions made at the onset of a conservation project can be informed by an understanding of some of the challenges that can arise during the process of development and implementation of conservation actions, which include potential mismatches in spatial, temporal, and functional scales. I discussed how scale mismatches can manifest at each stage of the conservation-planning process and can lead to a plan that does not account for the threats, risks, constraints, and opportunities posed by the complexities and dynamics of the social-ecological system or a plan that cannot be implemented fully or partially. In addition, scale mismatches can also affect the adaptive capacity of conservation institutions during project development and implementation because the institution's ability to detect, and therefore learn from, ecological changes occurring at scales other than the scale of operation is impeded.

An understanding of how these scale mismatches manifest themselves at various stages of project development and implementation can be used to predict the likelihood of success of conservation initiatives. Anticipating the potential for scale mismatches can inform the conservation-planning process so that mismatch problems can be dealt with effectively. Strategies to avoid scale mismatches may involve a spectrum of alternatives, where at one extreme the mismatch is addressed and strategies and actions are developed and applied at temporal and spatial scales that are appropriate for the problem, and at the other extreme the mismatch is not addressed and the likelihood (however reduced) that some positive conservation outcomes may transpire is hoped for. Whether scale mismatches are addressed may depend on the resources available, on competing considerations that shape decisions about scale (Mills et al., 2010), and on the viability of strategies and actions that could address the mismatch.

The importance of social networks to solving conservation problems is associated with the nature of environmental problems. Environmental problems are multiscaled and constantly evolving. Thus, solutions to these problems require local actors to have connections to broad levels of society (and vice versa) and require a flexible and open process. Solving environmental problems also requires transdisciplinary processes involving experts, government, and local stakeholders (Newman and Dale, 2007). I have shown how SNA can be applied to conservation planning to improve the effectiveness of conservation action. Specifically I showed how it can be used to help conservation actions be applied at the required spatial, temporal, and functional scales.

Chapter 3. Fit challenge: cross-scale collaboration to address scale mismatch

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3.1 Abstract

Large-scale conservation requires the involvement of numerous stakeholders to plan for and implement a range of activities across multiple scales. This is necessary in order to minimise the mismatch between the scale of management and the scale of ecological processes. Establishing and sustaining the effective collaborations necessary to achieve this is a key challenge. Utilising data from a large-scale conservation initiative in the south west of Australia I characterise the interactions between stakeholders as a social network. I employ a novel network theoretical approach to assess the different forms of collaboration, including cross-scale collaboration. I find that the social network predisposes cross-scale collaboration for invasive animal control, an action where coordination of activities is necessary. I find that for revegetation activities there is little evidence of collaboration across scales, but this could be fostered by a subset of stakeholders acting in a “scale-bridging” role. Addressing this will likely improve the effectiveness of revegetation efforts and the outcomes of the broader conservation initiative.

3.2 Introduction

Over the past two decades large-scale conservation initiatives have gained momentum as scientists and practitioners recognise the need to move beyond the identification and management of single protected areas to account for the management of surrounding landscapes (Lindenmayer and Burgman, 2005). This new approach requires the consideration of ecological processes and threats that transcend protected area boundaries and determine the persistence of biodiversity in the wider landscape (Cowling et al., 1999; Rouget et al., 2006). Examples of large-scale conservation initiatives around the world, such as Yellowstone to Yukon (Y2Y) in North America and The Great Eastern Ranges in Australia, are characterised by multiple land tenures and jurisdictions, heterogeneous land uses and land covers, and numerous stakeholders with diverse, and potentially conflicting agendas (Fitzsimons et al., 2013a; Worboys et al., 2010). Large-scale initiatives typically involve a diverse array of activities that span multiple ecological and management scales. Interpretations of the concept of 'scale' vary and its definition is still contested across the social and natural sciences (Higgins et al., 2012; Manson, 2008). I use the term 'scale' to refer to the way the different interests of stakeholders participating in conservation initiatives fit along different spatial scales, from the property level to that of the supra-regional (Saunders and Briggs, 2002) .

Large-scale conservation initiatives can benefit from an overarching plan (Fitzsimons et al., 2013a; Pressey and Bottrill, 2009) although this can hide local level variation in the values, interests and rights of local stakeholders (Cash et al., 2006; Pajaro et al., 2010). Overarching plans can be complemented by numerous smaller scale plans that are tailored to specific contexts and aligned to local realities (Lambert, 2013; Marshall, 2007). Still, integration across scales, is not automatically enabled by this approach (Carr, 2013). Collaboration between the stakeholders of large-scale initiatives, including property owners, local communities, and government and non-government organisations, is required to enable adaptation of regional plans to local preferences, reconciliation of numerous plans, or scaling up of local actions (Henson et al., 2009; Lowry et al., 2009; Pajaro et al., 2010; Wyborn and Bixler, 2013). This will improve the fit between the scales for planning and implementing conservation actions and the scale of biophysical processes, thereby avoiding scale mismatch.

The establishment of relationships between individuals or organisations can lead to the coordination of activities, and can facilitate the formation of common goals and objectives (Jones et al., 1997; Robins et al., 2011; Snijders et al., 2006). Most large-scale conservation initiatives are underpinned by different collaborative arrangements, from short-term engagements, to longer-term collaborative partnerships, to fully amalgamated institutions (Sabatier, 2005; Wyborn, 2013). These arrangements can entail formal agreements around a particular objective (e.g. Bode et al., 2010; Carlsson and Sandstrom, 2008; Lauber et al., 2011), or develop informally around specific issues of conservation interest or establish over years of association, such as through the interactions between farmers when advice and information is shared (e.g. Isaac et al., 2007; Newman and Dale, 2007; Vance-Borland and Holley, 2011). I define these varied forms of relationships between the stakeholders of conservation initiatives as *conservation social networks*.

While essential to addressing scale mismatch, the level of collaboration between stakeholders in conservation social networks and the effective integration of planning and management across scales are key challenges for private land conservation (Kearney et al., 2012), marine resource management (Berkes, 2006; Lowry et al., 2009; Pajaro et al., 2010), and landscape scale ecological restoration (Wyborn, 2013). Initiating, and ensuring the longevity and sustainability of collaborative relationships, can be a lengthy process that can be impacted by budget limitations and funding cycles (Fitzsimons et al., 2013a), dominance of individuals or organisations, obstructing disagreements between partners (Young, 2006), or simply a lack of willingness to collaborate (Knight et al., 2010). Simply prescribing formal collaborative arrangements will not necessarily overcome these barriers and translate into greater capacity for or more effective collaboration (Carr, 2013; Lubell, 2004). Strategic approaches to the formation and support of effective collaborations for large scale conservation initiatives are needed.

The application of network theory to the study of collaborative processes and structures has been applied in diverse disciplines, including in natural resource management and conservation (Bodin and Crona, 2009; Cumming et al., 2010). The goals of analysing conservation social networks can range from understanding formal policy and institutional forms of governance (e.g. Sandstrom and Carlsson, 2008) to more informal modes of governance that characterise many conservation endeavours, including community-centred governance and collaborative conservation initiatives (e.g. Ernstson et al., 2010; Lauber et al., 2008; Vance-Borland and Holley, 2011). Typically, applications of social network theory in conservation link the structural characteristics of the whole network to theory about the social processes that underpin effective conservation governance (e.g. learning and innovation). Such characteristics can be explored using descriptive network statistics such as network centrality, cohesion and density metrics (e.g. Cohen et al., 2012; Isaac et al., 2007). Fewer studies have analysed whole networks by examining the sub-network structures exhibited by stakeholders who interact within the network (but see Robins et al., 2011).

It is possible that comprehensive analysis of the relationships between stakeholders could reveal options to enhance collaboration within and across scales of planning and management. The analysis of these collaborative structural patterns would preface an understanding of the degree to which plans and activities are being, or can be, coordinated across scales, to avoid scale mismatch (Guerrero et al., 2013). This knowledge could then be utilised to inform a strategic approach for the formation or support of a conservation social network that enhances coordination of activities and higher forms of collaboration such as those leading to the formation of common goal and objectives. This could be facilitated by identifying key actors in the network or the connections between stakeholders that would be of most benefit.

I seek to better understand how stakeholders interact in a large-scale conservation initiative in Western Australia through analysing the conservation social network. I statistically explore the different modes of interaction within and across scales for different types of activities (McAllister et al., 2014). I determine the propensity of the network to facilitate collaboration across scales to support multi-scale conservation and minimise scale mismatch.

3.3 Methods

3.3.1 Study region

The Fitz-Stirling, my case study region, is situated in Western Australia in one of the world's 34 global biodiversity hotspots (Figure 3.1). This region is part of the Gondwana Link large-scale conservation initiative, which aims to restore ecological connectivity across over 1,000 kilometres in south-western Australia (Bradby, 2013). The Fitz-Stirling covers over 240,000 hectares, it is bounded by two of the largest areas of intact natural habitats that remain in the broader hotspot – the Fitzgerald River and the Stirling Range National Parks – and consists mostly of private farm land (cropping and sheep grazing) with scattered remnants of vegetation.

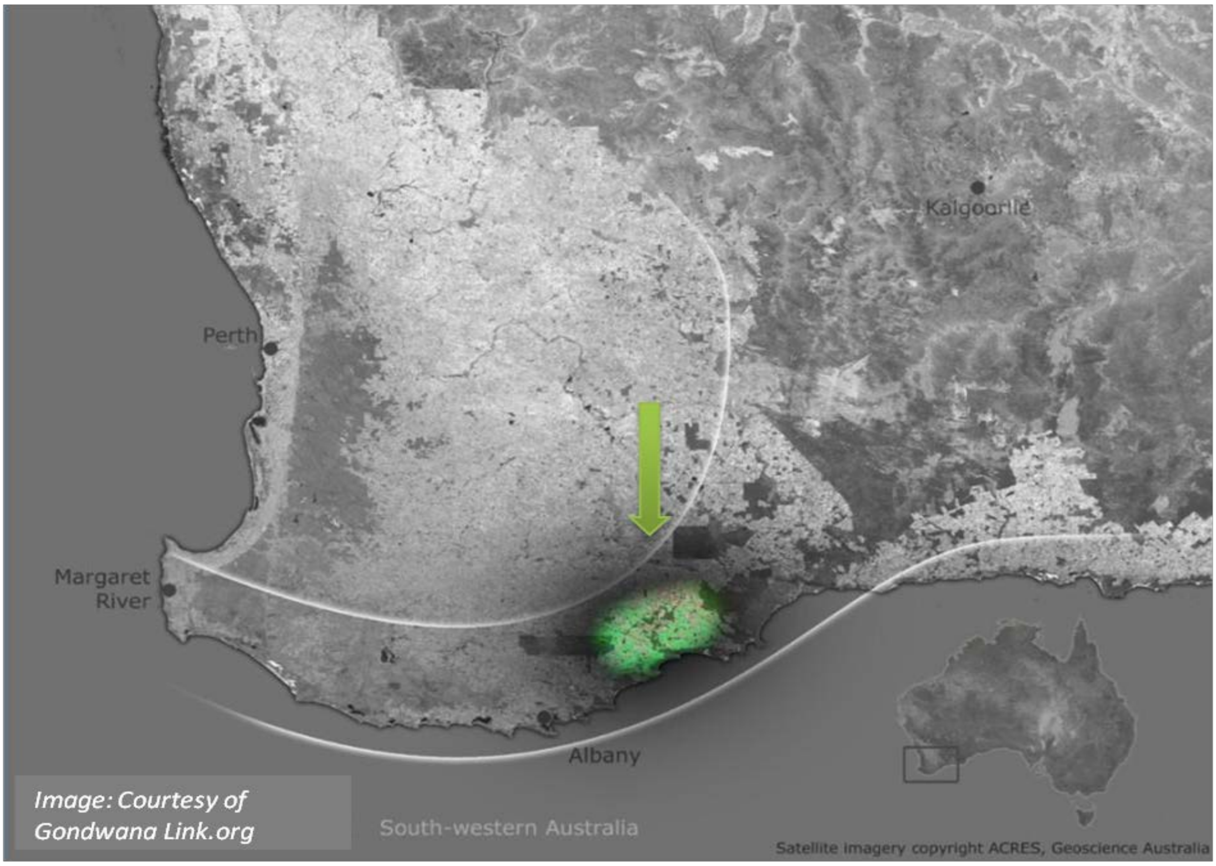


Figure 3.1. The Fitz-Stirling conservation region

3.3.2 Exploratory stage

This stage informed the design of the network study and involved 25 semi-structured interviews with stakeholders known to be involved in efforts to achieve conservation objectives for the Fitz-Stirling (Appendix A). I identified challenges to the implementation of conservation activities, including communication and coordination issues and the need for greater collaboration between different stakeholder groups. These findings were validated with quantitative methods (Appendix A, Figure A1) and pointed to the value of understanding how stakeholders interact to achieve conservation objectives for this region.

3.3.3 Network definition and data

The network was defined based on the collaborative interactions between stakeholders involved in conservation activities in the Fitz-Stirling region. An online survey was used to collect data on the people and organisations that each stakeholder collaborates with when performing different activities, including revegetation, protection of bushland and invasive species management (Table 3.1). A stakeholder was deemed part of the network on the basis of their involvement with conservation activities in the Fitz-Stirling region. The data were collected between October 2011 and July 2012, at which time the Fitz-Stirling conservation social network included four state and three local government agencies, one regional natural resource management group, seven NGOs, ten community groups, five university and research organisations, over 20 private organisations and independent contractors, and around 120 property owners. I coded stakeholders by scale of interest, as: property; sub-regional; and supra-regional level (Figure 3.2.). I obtained 38 completed online questionnaires (19 organisations and 19 landowners). With this data I were able to identify the full set of collaborative interactions for the 38 respondents plus partial information on the collaborative interactions for an additional 47 organisations who did not respond to the survey. (See Appendix A for a full description of data collection methods).

Table 3.1. Main types of conservation activities in the Fitz-Stirling.

Types of conservation activity	
Revegetation/restoration	Over 1750ha have been planted to date aided by the development and testing of large scale innovative restoration technologies, which include purpose built or modified machinery.
Livestock management	Activities such as fencing of bushland – to exclude cattle and sheep from sensitive areas are promoted by local community groups.
Weed management	Activities to control damaging weeds such as the South African lovegrass are promoted by local community groups.
Invasive animal control	Invasive animal control activities include state-funded but community-run programs such as the Red Card for Rabbits and Foxes program plus more localised activities led by non-government organisations or landholders.
Fire management	Landholders are required to ensure adequate firebreaks in their property to reduce the risk of wildfires. Bush fire brigades are coordinated by local government authorities and are comprised of hundreds of volunteer members (e.g. landowners). They assist in fire prevention and firefighting.
Land use planning	Local government organisations interact with diverse stakeholders when undertaking planning activities that affect the Fitz-Stirling region. These include development approvals and local strategic plans that contain conservation objectives.
Purchasing or setting aside land for conservation	Around 10,900ha of non-government conservation areas have been established in the Fitz-Stirling region since 2002 (Bradby, 2013).

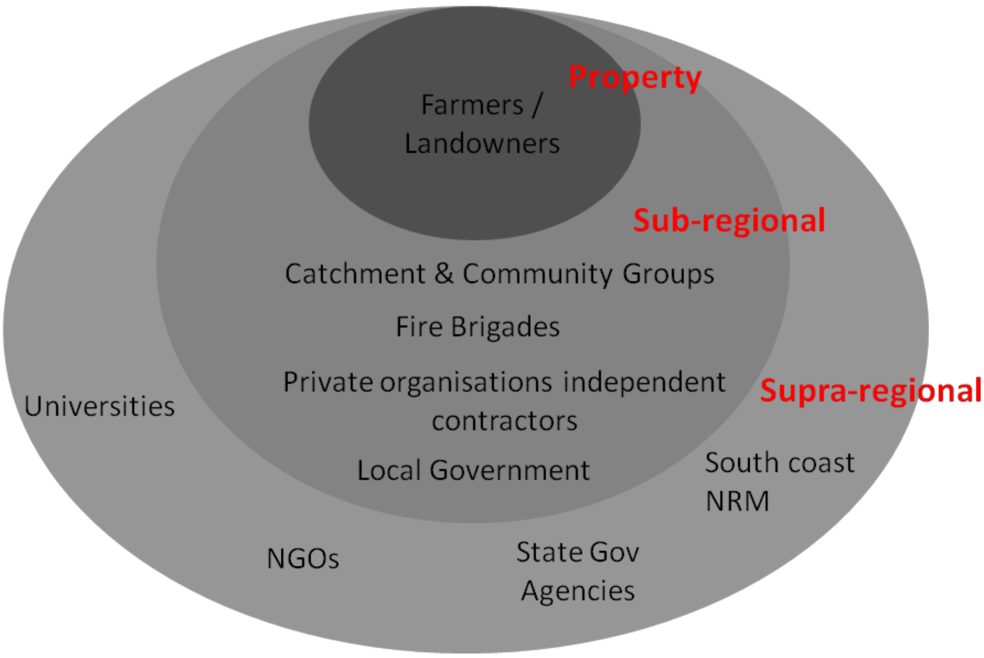


Figure 3.2. Fitz-Stirling stakeholders' scale of interest. At the property level a landowner decides whether they want to revegetate part of their land, protect remnants of bushland or manage threats such as invasive species. Community groups, local government and private organisations, make decisions at the sub-regional level, with an interest on particular areas (e.g. catchments). At the supra-regional level, state government and non-government organisations engage in state, national or international policy and projects, and provide funds or support for activities across the entire region. Universities and research organisations operate at a supra-regional level and also engage in projects which can influence and inform decisions across the entire region.

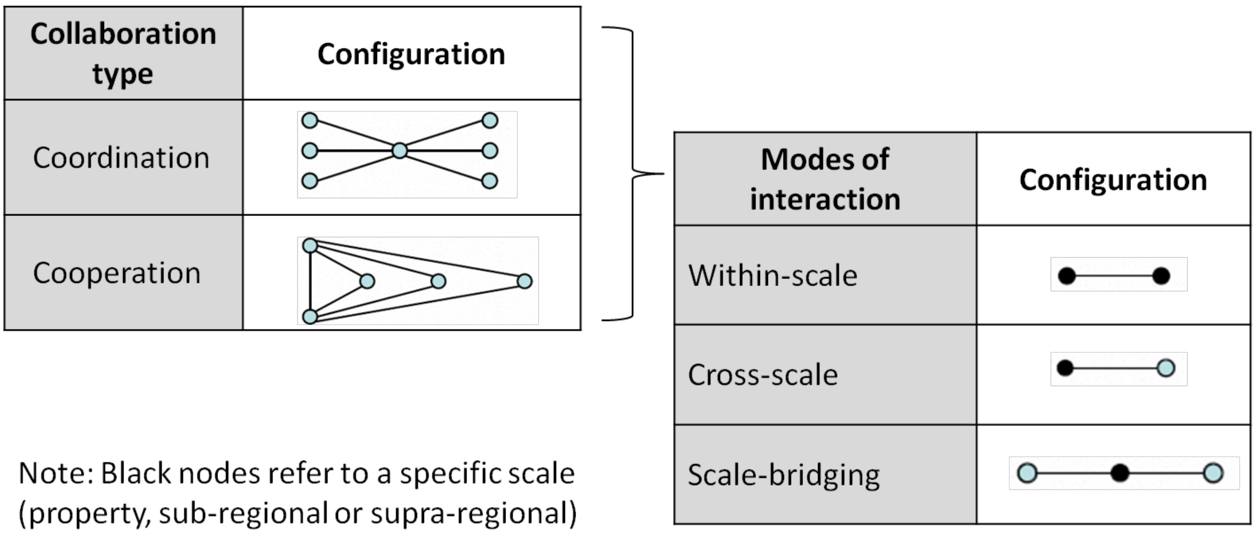


Figure 3.3. Conceptual framework. *Types of collaboration, modes of interaction, and associated configurations.*

3.3.4 Conceptual framework

My interest concerns the role of stakeholders that operate at multiple scales in a network. In particular I focus on how different types of interactions favoured by individuals and organisations contribute to whole of network collaboration, and collaboration within and across scales. I use the term 'collaboration' to refer to different types of interactions between stakeholders. Social processes such as the coordination of activities depend on such interactions being present (Granovetter, 1973; Olsson et al., 2007). The different roles of stakeholders manifest as observed variances in the types of sub-network interactions that they engage (McAllister et al., 2014). Such sub-network interactions are referred to as configurations, and these can be mapped to theoretical ideas beyond the limited set provided by a more conventional social network analysis, which is typically based on describing the detailed structure of entire networks (Bodin and Tengo, 2012; McAllister et al., 2014). To frame my analysis, I start by asking, what are the network configurations that I expect to see over- or under-represented for stakeholders engaged in activities that require collaboration within and across scales.

Social conservation networks can be structured in very different ways. For example, multiple stakeholders can tend to cluster around a single stakeholder, forming a 'star' (Figure 3.3). This network configuration connects stakeholders indirectly through a key stakeholder. In contrast, other configurations tend to be more tightly-bonded or 'closed', connecting stakeholders directly to each other (Figure 3.3). Within these general network configurations there can be different modes of interactions such as collaboration between stakeholders *within* the same scale; and collaboration *across* scales. There can also be *scale-bridging* configurations, where a stakeholder connects stakeholders from different scales (Ernstson et al., 2010).

3.3.5 Exponential random graph models and configurations

I utilise Exponential Random Graph Modelling (ERGMs) to identify the different modes of interaction that characterise the Fitz-Stirling network. Exponential Random Graph Modelling uses a statistical (regression) methodology to determine if certain configurations are more or less represented in an observed network than expected by chance alone (Snijders et al., 2006; Wasserman and Pattison, 1996). In this manner, and like other inferential statistics, ERGMs does not necessarily require that the whole network is measured (Robins et al., 2004). Observed frequencies of selected configuration are compared with frequencies derived from a large set of randomly generated networks to determine if these configurations are prevalent or rare in the network under study. In this way, statistical inferences can be drawn without the need for comparative networks. Importantly, ERGMs test for the prevalence of these configurations given the distribution of all other configurations included in the model. Some sub-network configurations are nested within higher order configurations, so two configurations might compete to explain an interaction observed in a network. Making inferences about a social process, such as collaboration, therefore requires analysis of nested configurations, allowing interpretation of an observed configuration relative to observations about all other configurations. In this way, ERGMs permit an understanding of the complex combination of social processes by which network connections are formed. I used the computer package pNet (Wang et al., 2009) to analyse three networks based on (1) all-activities (see Table 3.1), (2) revegetation, and (3) invasive animal control.

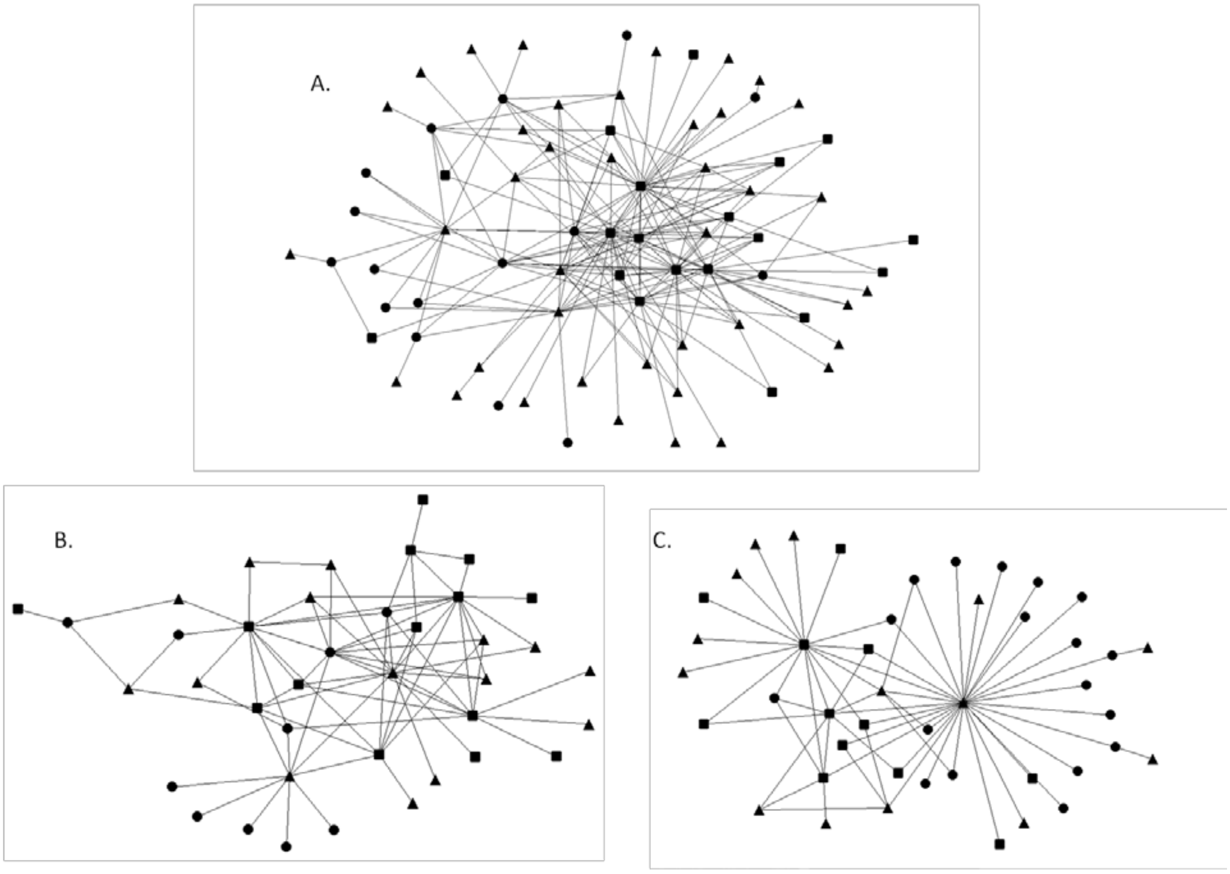


Figure 3.4. Conservation social network for the Fitz-Stirling. The all-activities (A), the revegetation (B), and the invasive animal control (C) networks. Nodes represent the different stakeholders and the links indicate collaborative interactions. The shapes of the nodes represent the scale of interest: property (circle), sub-regional (triangle), and supra-regional (square).

3.4 Results

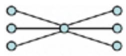
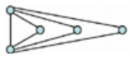
3.4.1 Exponential Random Graph Model

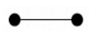
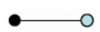
I parameterise a model for the all-activities, the revegetation, and the invasive animal control networks that included the 11 configurations shown in Table 3.2 (see section A2 in Appendix A for further modelling detail). For the all-activities network, 9 of the 11 configurations are statistically significant, indicating over- or under-representation of these configurations in the network (depending of the sign – or +; Table 3.2). My model contains parameters that account for the broad structural characteristics of the sampled network data ('star' and 'closed' configurations). These configurations are critical for providing a baseline that allows me to interpret the differences between the configurations of interest, namely *within* and *cross-scale* interactions.

My key interest concerns the role played by stakeholders operating at different scales across the network. The all-activities model shows significantly greater representation of *cross-scale* interactions for stakeholders at the property and sub-regional scales, suggesting their inclination to collaborate across scales. Conversely, the model shows fewer representations of *within-scale* interactions for stakeholders at the sub-regional scale, whereas they are over-represented for the supra-regional stakeholders. Lastly, there is evidence of scale-bridging roles suggested by the over-representation of *scale-bridging* interactions for all stakeholder categories.

For the revegetation activity network (Figure 3.4b), 7 of the 11 configurations are significantly over- or under-represented (Table 3.2). There is evidence of under-representation of *cross-scale* interactions, although scale-bridging roles are apparent. For the invasive animal control activity network (Figure 3.4c) the results show that *within-scale* interactions are under-represented for sub-regional stakeholders, whereas they are over-represented for the supra-regional stakeholders. *Cross-scale* interactions are represented greater than expected by chance for all stakeholder groups associated with invasive animal control activities.

***Table 3.2. Exponential Random Graph Model (ERGMs) model for the Fitz-Stirling conservation social network** for the (A) all-activities, (B) revegetation and (C) invasive animal control networks.* Estimated parameters and observed configuration counts are based on a model with a fixed density of 0.0574, 0.0218, and 0.0176 respectively.

		(A) All-activities network		(B) Revegetation network		(C) Invasive animal control network	
Form of collaboration	**Configuration**	**Parameter estimates**	**Observed counts†**	**Parameter estimates**	**Observed counts†**	**Parameter estimates**	**Observed counts†**
Coordination “bridging”		0.3965^	587	1.0822**	193	1.0958**	182
Cooperation		-0.1876^	256	-0.033	78	-0.3337	46

Mode of interaction	**Configuration**	**Parameter estimates**	**Observed counts†**	**Parameter estimates**	**Observed counts†**	**Parameter estimates**	**Observed counts†**
Within-scale (Property)		-0.275	5	N/A≠	0	N/A≠	0
Within-scale (Sub-regional)		-1.8257**	13	-0.9841	7	-3.4069**	5
Within-scale (Supra-regional)		0.9977**	49	0.6255	17	2.1195*	11
Cross-scale (Property)		0.3267**	73	-4.0711**	31	3.8099**	26
Cross-scale (Sub-regional)		0.7961**	142	-3.9261**	54	2.548**	53
Cross-scale (Supra-regional)		-0.0104	195	-4.1462**	71	0.8853*	47
Scale-bridging (Property)		0.0411**	283	0.088^	80	-1.8949**	9
Scale-bridging (Sub-regional)		0.0538**	514	0.1175**	175	-0.0392	459
Scale-bridging (Supra-regional)		0.0453**	1845	0.084*	271	-0.0454	180

^/*/** shows 90/95/99 % significance for the parameters. †T tests show no statistical difference between the observed configuration counts and simulation means. ≠One parameter could not converge as there are no instances of this configuration in the observed network.

3.5 Discussion

I have applied a novel method and developed an approach to analyse the ways in which stakeholders interact with one another to achieve conservation goals. I have applied this approach to a large-scale conservation initiative in Australia. The results suggest that in the Fitz-Stirling study region the coordination of plans and actions between stakeholders operating at a property or sub-regional scale is likely to present a challenge, given the under-representation of within-scale interactions at these levels. In particular, the coordination of invasive animal control activities in the Fitz-Stirling is less likely to occur at the sub-regional level, which is problematic given the prevalence of community-run programs for invasive animal control in the study region. The importance of coordinated efforts for the success of invasive animal control activities is well known (Coutts et al., 2013).

More central to my interest is the ability for collaborative conservation initiatives to enable the coordination of plans and actions across scales to minimise scale mismatch. Overall, cross-scale coordination in the Fitz-Stirling conservation social network can be facilitated by stakeholders at the property scale, and in particular, stakeholders at the sub-regional scale, who favour interactions across scales over within-scale interactions and show evidence of scale-bridging roles. Such roles have been observed in other studies. For example a study on farmer adoption of conservation practices suggests that sub-regional natural resource management bodies in Australia are better positioned than regional bodies to motivate cooperation from farmers (Marshall, 2009). This finding is particularly important for activities and programs devised at higher levels, and that are expected to be cascaded down to the property level for implementation. An example is invasive animal control programs originating from government agencies, such as the Red Card for Rabbits and Foxes operating in Western Australia since 2004, which provides government funding to local groups for the purchase of baits, and is mostly driven by local coordinators.

For the Fitz-Stirling region my results suggest that while capacity for within-scale coordination could be strengthened for invasive animal control activities, capacity for cross-scale coordination of this activity is strong. This contrasts with my finding for revegetation activities where coordination across scales is likely to present a challenge given the under-representation of *cross-scale* interactions. However, the results show that some sub-regional stakeholders are well positioned to facilitate cross-scale collaboration for revegetation activities, given the over-representation of *scale-bridging* interactions.

Making these stakeholders aware of their strategic position in the network, and supporting them in this role, may enhance coordination of revegetation activities across scales.

I demonstrate that network analysis can be used to determine the propensity for a network to support key social processes such as the coordination of key activities within and across scales. This information can then be used to identify ways that collaboration can be further promoted to minimise scale mismatch and inform future strategies and partnerships. I suggest that the Fitz-Stirling conservation social network should focus on developing new partnerships to strengthen relations within the sub-regional level, and provide further support to sub-regional stakeholders acting in a scale-bridging role.

My dataset is a partial sample of the complete network, which means network links are only observed around the organisations and landowners who responded to the survey. This means that the network data cannot be used to derive descriptive statistics about the network or make interpretations about individuals in the network. I have not attempted to do either. My analytical method treats network connections as a statistical sample, and hence robust conclusions can be draw from partial networks. A complete dataset reduces the standard errors and there are emerging technical approaches for ERGMs to reduce standard errors in the context of missing data (Koskinen et al., 2013). However, in my analysis statistical patterns were observed which tells us that my dataset is a sufficient sample from which to analyse the patterns observed. While more data are certainly better, network data can be expensive and time consuming to collate, often with substantial logistical considerations. I demonstrate that even with incomplete data it is possible to undertake a practical analysis for a real conservation problem, and in doing so I have also contributed to the development of new, cutting edge approaches for analysing social processes that are critical for the effectiveness of conservation actions.

The importance of collaboration for achieving conservation outcomes is well-known (Bode et al., 2010; Mazor et al., 2013), especially when multiple geographic scales are involved (Fitzsimons et al., 2013a; Wyborn and Bixler, 2013). I provide an approach for determining the potential for a network to support multi-scale conservation and demonstrate its utility in a complex system that involves multiple stakeholders that undertake diverse activities across scales. Targeted approaches for the development and support of collaborative relationships can reduce the complexity that characterises large-scale conservation initiatives. Specifically, it can avoid the inefficiencies that can result from comprehensive overarching approaches to specifying collaborative partnerships and governance arrangements. A targeted approach to understanding and enhancing collaborative relationships, such as the one I demonstrate, can improve the effectiveness of conservation initiatives by identifying and nurturing key stakeholders and important relationships.

Chapter 4. A social-ecological approach for assessing the problem of fit

Submitted to *Global Environmental Change* as “Achieving social-ecological fit through bottom-up collaborative governance: an empirical investigation”

4.1 Abstract

The management of ecosystems is thought to be enhanced by governance systems that adequately capture the extent and complexity of the biophysical system under management, thereby averting problems of social-ecological fit. Different forms of collaborative approaches to governance are often proposed as a way to increase the fit and enhance the effectiveness and efficiency of environmental management. I employ new theory and methodological approaches underpinned by interdisciplinary network analysis to investigate three key challenges associated with social-ecological fit. Specifically, I identify social-ecological network configurations that capture the hypothesised ways in which collaborative arrangements could overcome these challenges. Using social and ecological data from a large-scale biodiversity conservation initiative in Australia I determine how well the observed patterns of stakeholder interactions reflect these configurations. I find that co-management occurs when stakeholders manage the same spatially defined ecological resource, but not when they manage different yet interconnected ecological resources. This implies that the conservation initiative lacks capacity to detect the effects of management actions that could affect outcomes beyond the ecological resource being managed. In addition, I find that the collaborative arrangement is structured in a way to enable management across different levels (local, regional, supra-regional) thereby promoting adequate spatial scale matching. These findings corroborate with qualitative information obtained through semi-structured interviews of project stakeholders. My study provides empirical support for how collaborative approaches to governance can address the problem of fit, but also reveals that collaborative approaches do not necessarily solve all challenges associated with social-ecological fit. My approach also provides an avenue to incorporate both social and ecological data into evaluations of environmental projects, with the results of these evaluations able to inform the design of future initiatives better able to achieve social-ecological fit.

4.2 Introduction

The rapid loss of biodiversity, increasing pressure on natural resources, and the loss of ecosystem services are environmental problems of global significance. Of particular challenge are environmental problems that extend political jurisdictions (e.g. the conservation of migratory species; Runge et al., 2014) or both jurisdictions and policy sectors (e.g. ocean and water management; Crowder et al., 2006; Sabatier, 2005); those that are characterised by uncertainty and rapid cumulative changes (e.g. the management of invasive species; Hobbs, 2000); and those for which there is a strong interplay between social and ecological systems (e.g. land-use change; Lambin et al., 2001) and spatial and temporal scales (e.g. climate change; Wilbanks and Kates, 1999). The ability to address many of these complex challenges is contingent upon the degree to which a governance system fits (or aligns with) the characteristics of the biophysical system (Ekstrom and Young, 2009; Epstein et al., 2015; Folke et al., 2007; Young, 2002). This is referred to as the problem of social-ecological fit. A lack of fit is thought to reduce the effectiveness and efficiency of environmental management (Dallimer and Strange, 2015).

Significant progress has been made in conceptualising the problem of social-ecological fit, and interest in the topic spans the social and natural sciences (Cumming et al., 2006; Epstein et al., 2015; Folke et al., 2007; Galaz et al., 2008; Pelosi et al., 2010; Vatn and Vedeld, 2012; Young, 2002; Young et al., 2008). Some studies have approached the topic from a policy and institutional perspective (e.g. Cosens, 2013; Ekstrom and Young, 2009; Morrison, 2007; Nagendra and Ostrom, 2012; Ostrom, 1990); others have focussed on multi-stakeholder governance processes (e.g. Meek, 2013; Olsson et al., 2007; Wyborn, 2014); and some have taken a structural approach focusing on the interactions between governance actors (Bergsten et al., 2014; Bodin et al., 2014; Guerrero et al., 2015; Treml et al., 2015). Other studies have highlighted the problem through a managerial lens, identifying instances where management actions are not suited to the biophysical system of interest (Hobbs et al., 1993; Saunders and Briggs, 2002). A core recommendation that has arisen from this body of work is that diverse actors (e.g. government, non-government and community groups, and individuals) should coordinate the planning and implementation of environmental management actions (Brondizio et al., 2009; Carlsson and Berkes, 2005; Folke et al., 2005; ÖSterblom and Bodin, 2012; Ostrom, 2010; Walker et al., 2009; Young, 2002).

Environmental problems often extend large geographic areas and require management over extended periods of time; as such governance involves actors across multiple scales (local, regional, supra-regional) and across distinct geographic yet ecologically connected areas. Global intergovernmental regimes seek to provide a coordinated approach to identifying solutions to environmental problems and are an example of forums for international decision-making (e.g. the Convention on Biological Diversity and the Intergovernmental Panel on Climate Change; Newell et al., 2012). Decision-making forums at national and regional scales then formulate specific actions that have societal relevance (e.g. national climate change policy, regional investment in threatened species management, and national conservation initiatives). Local forms of governance can then promote engagement in conservation practices and the required behaviour change of resource users and beneficiaries (e.g. consumption preferences, adoption of policies, conflict resolution). Interaction between these governance actors that operate at different scales is thought to enhance the formulation, refinement and coordination of actions that are locally implemented but have global consequence (e.g. Galaz et al., 2014; Meek, 2013).

Collaborative forms of governance (as opposed to command-and-control e.g. Holling and Meffe, 1996) are increasingly regarded as essential to accomplish the critical interactions needed to achieve coordinated action across scales (e.g. Ansell and Gash, 2008; Emerson et al., 2012; Lubell, 2015; Provan and Kenis, 2008). Collaborative governance can be accomplished through the integration of knowledge systems and by bringing together the resources, scientific capacity, and commitment needed to facilitate critical learning processes and understand the complexity of systems (Armitage et al., 2012; Armitage et al., 2009; Folke et al., 2005; Olsson et al., 2007; Ostrom, 2010; Pahl-Wostl, 2009). Although collaborative governance is typically thought of as a managed process (e.g. Sørensen and Torfing, 2007), it largely rests on the assumption that various actors will seek to engage in collaborative activities in order to collectively solve problems (Lubell et al., 2014).

While there is increasing theoretical support for collaborative approaches to governance as a mechanism for addressing the problem of social-ecological fit, empirical research to support this prescription is lacking (Huitema et al., 2009; Newig and Fritsch, 2009). In practice, the many costs associated with establishing and maintaining collaborative arrangements can interfere with the capacity to effectively deal with ‘fit challenges’ (e.g. Wyborn, 2014). High transaction costs, conflicting mandates, trust issues, competition and overly complex governance arrangements are some of the factors that can reduce the efficacy of collaborative approaches to governance (Ansell and Gash, 2008; Aswani et al., 2013; Lazer and Friedman, 2007; McAllister and Taylor, 2015; Newig and Fritsch, 2009; Visseren-Hamakers and Glasbergen, 2007).

In this chapter I take an empirical approach to exploring the potential of collaborative approaches to governance in achieving social-ecological fit. I employ new network theory and methodological tools to analyse both social and ecological data from a large scale conservation initiative in Australia (Bodin et al., 2014; Bodin and Tengo, 2012) and combine this approach with new statistical models of multi-level networks (Wang et al., 2013). I demonstrate the value of this novel research approach for the evaluation and future design of collaborative environmental initiatives.

4.3 Theoretical background: Specific social-ecological fit challenges and the potential role of collaborative governance

Problems of fit arise from challenges related to the connectedness and interdependence between ecological and social systems (Galaz et al., 2008). From a social-ecological system perspective, elements of the social system (governance systems, organisations, resource users, civil society) acting at different levels (local, regional, supra-regional), interact with elements of the ecological system through a set of actions ranging from those related to land and resource use through to management and conservation (Gunderson and Holling, 2002; Liu et al., 2007). As the spatial and/or temporal scale of environmental problems increases, fit challenges can arise from these interactions (or lack of interaction). In this section I introduce three types of social-ecological fit challenges and the hypothesised ways in which collaborative approaches to governance could address them. These fit challenges constitute the focus of my analysis.

A social-ecological fit challenge can arise when two or more actors have an interest or stake in the same biophysical resource. Acting independently of each other's decisions can in some circumstances lead to overexploitation of resources or ineffective management (Ostrom, 1990). This governance challenge is specifically relevant for environmental problems that extend across jurisdictions and national borders. For example, watershed management can be ineffective if management is confined to individual jurisdictions without coordination between jurisdictions (Sabatier, 2005). Collaborative governance can enable co-management of biophysical resources so that planning and implementation of management actions can be coordinated amongst multiple actors (e.g. O'Keefe and DeCelles, 2013).

Social-ecological fit also relates to the interconnected nature of biophysical systems (Galaz et al., 2008). For example, a fit challenge arises when ecological resources are connected (e.g. vegetation patches on farming land or along wildlife corridors) (e.g. Yellowstone to Yukon wildlife corridor; Chester, 2006) but management responses are applied to distinct ecological resources in isolation. When ecological resources are interdependent adverse effects can spread beyond the domain of a managing actor (e.g. the dispersal of invasive species and the depletion of fish stocks in river systems), reducing the effectiveness of management. From a natural science perspective, the need to account for the interconnected nature of biophysical systems to inform solutions to environmental problems has long been recognised (Beger et al., 2010; Christensen et al., 1996). It is increasingly common for land management policy and on-ground programs to seek to enhance the functional connectivity of landscapes to reduce the risk for species extinctions (e.g. Saura and Pascual-Hortal, 2007), address the rate of invasive species spread (e.g. Chades et al., 2011), and protect the conservation values of interdependent areas (Iwamura et al., 2014; Martin et al., 2007). Notably, interconnected, or "boundary-spanning", biophysical systems have been identified as an important consideration for investigating environmental governance and the problem of fit (Galaz et al., 2008; Young et al., 2008). Through collaborative governance, the "spillover" effects resulting from managing one of the ecological units can be fed back to the managing actor through sharing of information and expertise with other actors managing adjacent resources, therefore directly experiencing these spill-over effects. Collaborative governance can therefore increase opportunities for tightening feedback loops between actions and outcomes in the management of interconnected systems and enable the internalisation of system-level management costs and benefits.

In addition, social-ecological fit challenges can arise from issues associated with scale. The definition of scale is contested and interpretations differ across the social and natural sciences (Higgins et al., 2012; Sayre, 2005; Termeer et al., 2010). Here I focus on spatial scale, and use the term ´scale´ to refer to the spatial dimension at which diverse ecological and management processes occur – which vary along a continuum from local to broad levels of ecological and social organisation (Cash et al., 2006; Sayre, 2005). Scale challenges arise due to a multitude of reasons. First, the management of environmental problems tends to be planned at diverse spatial scales (from a global scale to that of individual property) while actions are commonly implemented at local scales (e.g. farm paddock; Saunders and Briggs, 2002). Second, biophysical systems are underpinned and affected by multiple ecological and anthropogenic processes that operate simultaneously at numerous spatial scales (from nanometres to tens of thousands of kilometres), and the dominant patterns and relationships observed depend on the spatial (or temporal) scale of observation (Cumming et al., 2014; Levin, 1992; Poiani et al., 2000; Saunders and Briggs, 2002; Wiens, 1989). Scale mismatches can therefore occur if management is applied only at one scale (Cattarino et al., 2014; Kearney et al., 2012). For example, broad-scale actions implemented across extensive areas can fail to have a positive effect at a local level (e.g. Wilson, 2006). Minimising scale mismatches could be important for responding to problems such as climate change (Ostrom, 2010), controlling invasive species (e.g. McAllister et al., 2015) and delivering large-scale restoration efforts (e.g. Guerrero et al., 2015). Through collaborative governance, diverse actors operating at multiple scales (corresponding to local, regional, supra-regional levels) can generate and share the knowledge required to coordinate responses to threats across different levels and implement actions at the most appropriate scales.

Although the concept of social-ecological fit is conceptually quite well developed, empirical research on the capacity for collaborative governance to address the problem of fit is in its infancy. There are few studies that simultaneously examine both social and ecological dimensions in investigations of the problem of fit (Cumming et al., 2010; Pelosi et al., 2010), or that link theories and insights on governance approaches to the features of the system under investigation (but see Bodin et al., 2014). In addition, traditional research and analytical methods within the social sciences are not designed to theoretically capture and account for the ecological system in detail. Thus, new analytical methods that can integrate social and ecological data and that permit analysis of the connections between social and ecological systems are needed (Bodin and Tengo, 2012). In this chapter I make use of social and ecological case study data together with a new theoretical and methodological framework (Bodin et al, in review) to theorise specific social-ecological fit challenges and empirically test whether a collaborative conservation initiative is able to address these challenges.

4.4 Material and methods

4.4.1 Analytical framework

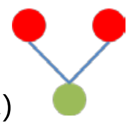
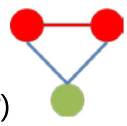
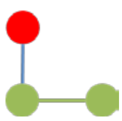
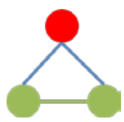
A network perspective can been used to describe and analyse diverse patterns of social-ecological interactions (Janssen et al., 2006), where nodes can be used to describe actors and ecological components, and ties can be used to describe social connections, ecological connections, and social-ecological connections (e.g. due to management or resource use). Together these different types of interactions can be used to describe a social-ecological system as a social-ecological network. Bodin and Tengo (2012) propose a framework for analysing social-ecological systems where specific patterns of social and social-ecological interactions are characterised by particular node configurations (referred to herein as building blocks) and can be theoretically linked to specific governance challenges. Using this framework I identify building blocks that are consistent with the hypothesised ways in which interconnected ecological systems could be managed (i.e. possible collaborative arrangements) so as to enhance the social-ecological fit (Table 4.1). I expand this building block approach to include considerations about scale (Guerrero et al., 2015).

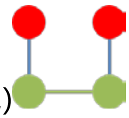
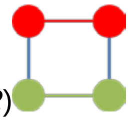
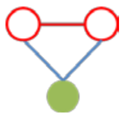
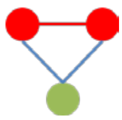
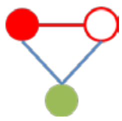
The first social-ecological fit challenge relates to the management (or use) of shared ecological resources (e.g. vegetation patches, catchments, species; Table 4.1A). This is illustrated by building block a1 (Table 4.1) where two red nodes (governance actors) are connected to the same green node (ecological unit) but are not connected to each other (i.e. they do not collaborate). The ability of a governance system to deal with this challenge is likely to increase when actors collaborate to coordinate activities (as is illustrated by building block a2 in Table 4.1).

The second social-ecological fit challenge relates to the management of interconnected ecological resources (Table 4.1B). This can occur when an actor has an interest in an ecological resource that is directly or indirectly connected with another ecological unit, or two connected ecological units are managed independently by two different actors. This is illustrated by building blocks b1 and bb1 (Table 4.1) where two green nodes (ecological units) are connected to each other, but a red node (governance actor) is only connected to one of them (b1 in Table 4.1) or the two connected green nodes (ecological units) each have a connected red node (governance actor) but these red nodes are not connected to each other (i.e. they do not collaborate; bb1 in Table 4.1). The ability of a governance system to overcome this challenge is likely to increase when a governance actor is involved in the management of each connected ecological resource (b2 in Table 4.1). The governance ability also improves when actors managing two connected ecological resources collaborate (bb2 in Table 4.1).

The last type of social-ecological fit challenge concerns the matching between the scale of ecological processes and the scales of management (Table 4.1C). This is illustrated by building block c1 where two red nodes (governance actors) connected to a green node (ecological unit) are associated with the same scale of management. The ability of a governance system to overcome this challenge is likely to increase when actors associated with different scales of management collaborate to coordinate actions across different levels, therefore increasing their joint ability to address key cross-scale ecological dynamics (local, regional, supra-regional; c2 in Table 4.1; Cash and Moser, 2000; Guerrero et al., 2013; Young et al., 2008).

Table 4.1. Conceptualising fit challenges through a social-ecological 'building block' approach.

Type of fit challenge	**Social-ecological building blocks#**		**Examples**
	misfit	*fit*	
A. *Management of shared ecological resources* When more than one actor uses and/or manages the same ecological resource. Collaboration (red line) could prevent overexploitation or ineffective management.	(*a1*) 	(*a2*) 	Bodin et al., 2014 Bergsten et al., 2014
B. *Management of interconnected ecological resources* Management of connected resources (b2) or collaboration	(*b1*) 	(*b2*) 	

Type of fit challenge	Social-ecological building blocks[#]		Examples
between actors (bb2) could permit detection of adverse effects beyond the domain of the managing actor and negative feedback loops (e.g. when the effects of management actions involving one ecological resource negatively affect outcomes in connected ecological resources).	(*bb1*) 	(*bb2*) 	Bodin et al., 2014 Bergsten et al., 2014
C. Cross-scale collaboration to maximise spatial scale matching Maximising the scale matching between the scale of management and the scale of ecological processes requires coordination of a range of actions across different levels.	 or  (*c1*)	(*c2*) 	Guerrero et al., 2015 McAllister et al., 2015

[#]Actors (red circles) are connected with each other (red lines) and with ecological units (green circles). Red and white circles in building blocks in *c1* and *c2* differentiate between actors associated with different scales of management.

4.4.2 Study region and data

My case study region, the Fitz-Stirling, is situated in Western Australia. This case study region confronts, and is an illustrative case of, several global environmental issues, including extensive deforestation, salinisation, incidence of wildfire, propagation and persistence of invasive species, and the compounding and uncertain effects of climate change. The Fitz-Stirling is situated in one of the world's 34 global biodiversity hotspots (Figure 4.1), covering over 240,000 hectares consisting mostly of private farm land (cropping and sheep grazing) with scattered patches of vegetation, and is bounded by two of the largest areas of intact natural habitats that remain in the broader hotspot – the Fitzgerald River and the Stirling Range National Parks.

The Fitz-Stirling is part of the Gondwana Link large-scale conservation initiative, which aims to restore ecological connectivity across over 1,000 kilometres in south-western Australia (Bradby, 2013). The initiative is founded upon principles of providing venues to facilitate and support collaboration between different actors including government and non-government organisations. As such, the Fitz-Stirling represents a case of collaborative governance for biodiversity conservation.

Semi-structured interviews and an online survey was used to collect data on the collaborative interactions between the actors who undertake conservation and management activities in the region – including revegetation, protection of remnant vegetation, fire management and invasive species management. Information on the geographic locations where they performed these activities was also collected. Actors were coded by scale of interest, as: property; sub-regional; and supra-regional level. The present analysis includes interactions across sub-regional and supra-regional scales (see Appendix A for a full description of data collection methods).

I used publicly available data on the distribution of native vegetation in the Fitz-Stirling region. Over 2000 distinct vegetation patches (ecological resources) were identified. My survey method required respondents to indicate on a map the vegetation patches in which they applied their conservation and management activities. To make this feasible, the vegetation patches were clustered based on a 0.5 km dispersal threshold. This threshold was chosen since many bird and mammal species would experience any two patches within a 500 meter range to be well connected and therefore effectively seen as one coherent area of habitat, or a “meta-patch” (Sutherland et al., 2000; Zetterberg et al., 2010). This resulted in 80 vegetation clusters.

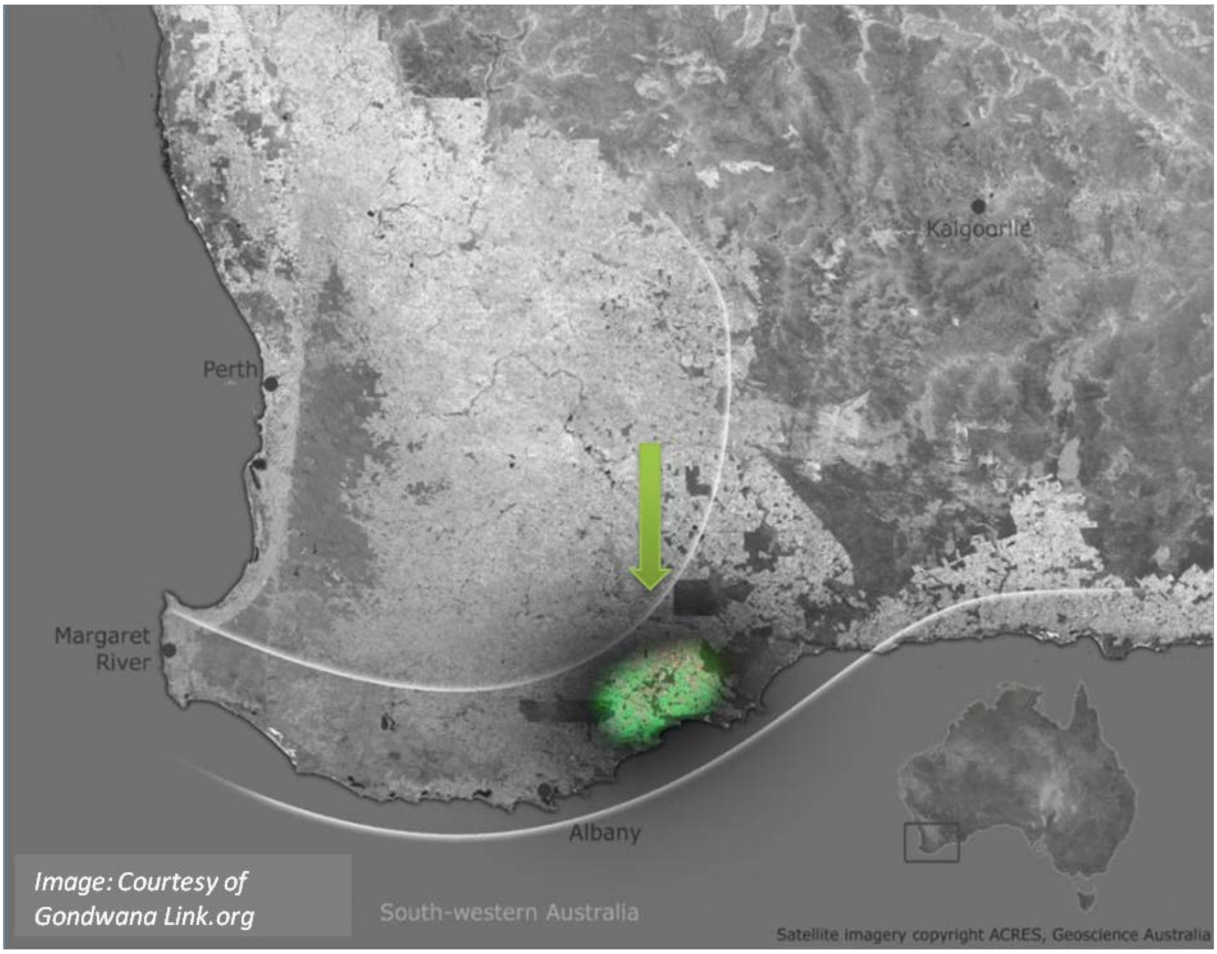


Figure 4.1. *The Fitz-Stirling conservation region.*

4.4.3 Characterising and analysing the Fitz-Stirling social-ecological network

I characterised actors, vegetation clusters, and their interactions, as a social-ecological network, where actors are the social nodes and vegetation clusters the ecological nodes. The social connections were defined based on collaborative interactions between actors when performing key activities in the Fitz-Stirling region. These comprised revegetation, invasive species control, livestock management, weed management, fire management, setting aside land for conservation, and land use planning activities. The ecological connections between vegetation clusters were defined based on a species maximum dispersal threshold of 1km (Figure 4.2). The maximal dispersal distance corresponds to the distance that a species will not be able to, in most cases, exceed while dispersing between different habitat areas. The threshold is ultimately a species specific measure. However, I choose a 1 km threshold since then I am able to describe the landscape's level of connectivity for a fairly broad range of species. The selected threshold is relevant for many bird species, as well as for several mammals and amphibians species (Bergsten et al., 2013; Sutherland et al., 2000; Zetterberg et al., 2010). But small mammal and insects species would require smaller thresholds that better reflect their dispersal across the landscape, as would larger mammals typically call for larger thresholds (Sutherland et al., 2000). Interactions between the two networks (i.e. the social-ecological connections) characterise the interest that actors place on the different locations (vegetation clusters) for performing their conservation activities. The resulting Fitz-Stirling social-ecological network consists of 15 social and 80 ecological nodes and their ecological, social, and social-ecological connections.

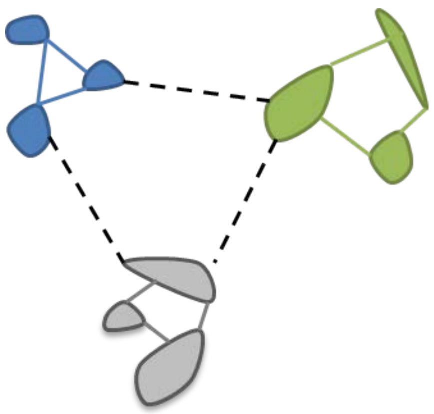


***Figure 4.2. Exemplification of clusters and their connections*.** Vegetation patches (blue, green and grey) were grouped into clusters based on 0.5 km (blue, green and grey lines) and the clusters were considered ecologically connected if they were located within a distance of 1 km (dotted lines).

Utilising novel statistical network methodologies (section 4.4.4), and following my analytical framework (outlined in section 4.4.1), I analyse data describing the Fitz-Stirling social-ecological network, to test for the statistical representation of building blocks that theoretically characterise each social-ecological fit challenge. If the collaborative approach followed by the Fitz-Stirling conservation initiative has the capacity to address the fit challenges outlined in my analytical framework, I would expect for building blocks a2, b2, bb2 and c2 in Table 4.1 to be represented in the Fitz-Stirling network data more than would otherwise be expected by chance. For instance, for the challenge associated with the management of interconnected ecological resources I would expect a high incidence of triangles (b2 in Table 4.1), or 4-cycles (bb2) relative to the occurrence of other building block structures across the network.

4.4.4 Analytical methods: Multilevel Exponential Random Graph Modelling (MLERGM)

I integrate my analytical framework (section 4.4.1) with a methodological approach developed within the social sciences. This approach is a recent extension of a class of stochastic network models called Exponential Random Graph Models (ERGM; Frank and Strauss, 1986; Snijders et al., 2006; Wasserman and Pattison, 1996). ERGM account for multilevel networks where two-layered networks are connected through cross-level ties (Wang et al., 2013), as it can occur in a social-ecological network. These multilevel networks can contain multiple network configurations (e.g. different patterns of connected social and ecological nodes). ERGM tests the prevalence of selected configurations relative to the distribution of all other configurations in the network and accounts for nested configurations – when a configuration contains one or several other configurations. In this way the ERMG approach facilitates more precise interpretations of an observed configuration than approaches that assume independence of configuration observations. Unlike other network analysis approaches, ERGM simultaneously evaluate a large set of network configurations to determine a selected (modelled) set of configurations' relative importance in explaining the structure of a social-ecological network. Hence, it builds on an analytical approach where the researcher can test if there are some specific configurations that can explain the observed structure of the whole network. This feature makes it an ideal analytical tool for testing hypothesis about the processes, expressed as specific configurations (building blocks) that gave rise to the observed network. Multilevel ERGM (MLERGM) has the ability to account for connections between multiple layers of a network (Wang et al., 2013). For example, it can be identified if the way ecological units are connected is associated to the way actors relate to each other, or the way actors are connected to ecological units (i.e. through management). In addition to network configurations, attributes of the nodes (ecological units or actors) can be included in the analysis to establish if they have significant structural effects. For instance, it can be tested if an actor exhibiting a particular attribute (e.g. a particular scale of management) is associated to the way they connect to ecological units (e.g. through the types of management decisions).

The integration of this advanced statistical network methodology with the 'building block' approach to analysing social-ecological systems was recently proposed by (Bodin et al., in review). Through the Fitz-Stirling case study I further elaborate and provide the first application of this novel research approach for the analysis of a collaborative initiative's ability to accomplish social-ecological fit. While current stages of software development prevented me from including all configurations from my analytical framework in my MLERGM model (c1 in Table 4.1), I demonstrate how the theoretical basis provided by the social-ecological building block approach can guide interpretation of the outputs of MLERGMs, and how this interpretation can lead to valuable insights on how the structure of governance arrangements can be improved so as to increase their level of social-ecological fit.

4.5 Results

I fitted a MLERGM model to data describing the Fitz-Stirling social-ecological network that included the eleven configurations shown in Table 4.2. Six of these configurations (a - f) are baseline configurations that help explain the overall social-ecological network structure. These configurations are critical for providing a baseline to interpret the results for the configurations of interest, namely those that suggest social-ecological fit (configurations g - k in Table 4.2). The parameter estimate for configuration c is significant and negative in the MLERGM. This suggests that the Fitz-Stirling social-ecological network contains fewer of these configurations than would otherwise be expected by chance, implying that actors do not give preference to a location already being managed by other actor(s). The significant and positive parameter estimate for configuration d suggests that some actors tend to undertake their activities in many different locations compared to an average actor. Likewise, configuration e is significantly positive, suggesting that locations chosen by actors tend to be connected with other locations. In contrast, a significant and negative parameter estimate for configuration f suggests that actors do not favour collaboration with other actors working in a location different to their own.

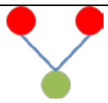
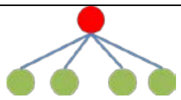
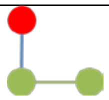
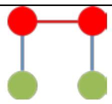
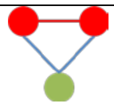
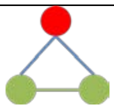
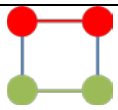
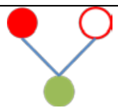
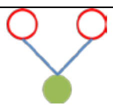
The remaining configurations in Table 2 relate to social-ecological fit, and reflect the social-ecological building blocks described in my conceptual framework (Table 4.1). However, a key feature of the ERGM method is that interpretation of results can be enhanced by simultaneously considering results of configurations that are nested within others. Thus, in the results that follow my interpretations for the configurations that relate to social-ecological fit also consider the results for some of the baseline configurations already discussed.

Configuration g relates to the fit challenge of *management of shared ecological resources*. The parameter estimate for this configuration is significant and positive. Interpretation of this result can be enhanced by simultaneously considering the parameter estimate for baseline configuration c. This is because configuration c is nested within configuration g. The consideration of parameter estimates for c and g suggests that even though two actors do not tend to work in the same location (given the negative parameter estimate for baseline configuration c), when they do work in the same location (i.e. the same ecological unit) they tend to collaborate (given the positive parameter estimate of configuration g).

Configurations h and i relate to the fit challenge of *managing interconnected ecological resources*. The result for configuration h considered together with the result for baseline configuration e suggests that actors tend to work in a location that is connected to another location (given the significant and positive parameter estimate for configuration e), and that they have a strong propensity towards working in these other locations as well (given the significant and positive parameter estimate for configuration h). Configuration i depicts situations where two actors each manage two different, yet connected, ecological resources. The parameter for this configuration is significant and negative, suggesting that there are fewer of these configurations in the Fitz-Stirling social-ecological network data than would otherwise be expected by chance. This result considered together with the result for baseline configuration f, implies that actors do not favour collaboration with others working in different locations to where they work (given parameter estimate of baseline configuration f) and even less so when these locations are connected (given parameter estimate for configuration i). From a governance perspective a negative parameter for configuration f can be seen as desirable, as spending time collaborating with other actors whose locations are not the same as your own is potentially inefficient. However, this result is less desirable when considering the result for configuration i, since when the locations are connected, collaboration between actors could improve the coordination of management activities (e.g. invasive species management, revegetation of areas to facilitate wildlife movement).

The last two configurations depict situations where locations are being managed at one scale (configuration k) or at different scales (configuration j). The negative and significant parameter estimate for configuration k suggest that actors managing at the same scale tend to avoid managing the same locations (and this over and above the general avoidance of sharing locations as observed through c). Conversely, the positive and significant parameter estimate for configuration j suggests that actors managing at different scales tend to manage the same locations (albeit the general tendency to avoid sharing locations still applies). These results imply that even though actors tend to avoid locations managed by others, this tendency is reduced if the other actor is managing at a different scale. While I was not able to test whether the actors managing at different scales tended to collaborate or not (Table 4.1, c), the general tendency to collaborate with actors sharing a resource still applies (Table 4.2, g).

Table 4.2. Multilevel Exponential Random Graph Model (MLERGM) for the Fitz-Stirling social-ecological network. Estimated parameter and observed configuration counts are based on a model where the ecological network component has been fixed. This reflects my interest to test whether connections between ecological units are associated to the locations chosen by actors (the social-ecological connection), and who they chose to collaborate with (the connection between actors).

	Parameter estimates (t-stat)		Standard error	Observed counts (t-stat#)	
Baseline configurations					
a. 	-0.0126	(-0.05)	0.232	60	(0.09)
b. 	-1.4323	(-1.62)	0.885	253	(0.08)
c. 	-2.929	(-6.83)*	0.429	311	(0.08)
d. 	1.1565	(3.80)*	0.304	467	(0.09)
e. 	0.3663	(2.56)*	0.143	431	(0.00)
f. 	-1.1752	(-4.98)*	0.236	45	(0.06)
Configurations associated with social-ecological fit					
g. 	1.2367	(6.37)*	0.194	289	0.07
h. 	1.9868	(10.03)*	0.198	165	0.02
i. 	-0.2702	(-5.87)*	0.046	600	0.06
j. 	1.408	(3.23)*	0.436	301	0.07
k. 	-0.4945	(-2.91)*	0.17	130	0.02

* shows 99% significance for the parameters. #T tests show no statistical difference between the observed configuration counts and simulation means.

4.6 Discussion

I demonstrate a robust empirical approach to analyse a challenge associated with managing the world´s most pressing environmental problems: the problem of social-ecological fit. I do this through the application of a novel multilevel network analytical method to a specific collaborative initiative for environmental governance in south west Australia. Using this approach I am able to integrate social and ecological considerations in evaluating the capacity for a conservation initiative to address three governance challenges related to the problem of fit: management of shared ecological resources, the management of interconnected ecological resources, and spatial scale mismatch. This ability is an important component in devising evidence-based policies for maintaining or revising collaborative arrangements.

My results suggest that the collaborative approach of the Fitz-Stirling conservation initiative is able to deal with some but not all of the governance challenges related to the problem of fit described in Table 4.1. For instance, collaboration is favoured when actors share the same locations, but not when managing connected locations. This indicates that this initiative can enable the coordination of diverse management actions for particular locations (management of shared ecological resources), but may not support coordinated management actions across different locations (management of interconnected ecological resources). This result is supported by data on the perceptions held by Fitz-Stirling actors on the effects of collaborative relationships on the performance of on-ground activities. For example, 83% of collaborations were perceived as delivering some, very good, or results above expectations to the particular activity performed at particular locations (Appendix A, Figure A2). In contrast, insufficient communication and coordination was highlighted as an important barrier to successfully carrying out conservation activities across the region (Appendix A, Figure A1). Qualitative data captured through in-depth interviews suggest that this is especially a problem when diverse management actions need to be coordinated so as to reduce undesirable feedbacks between activities. For example, fire management strategies can negatively affect revegetation outcomes when they do not consider revegetation activities being carried out in nearby locations. Likewise, revegetation plans that ignore fire management plans can reduce the effectiveness of fire management activities across locations.

While I was unable to directly test the particular social-ecological network configuration related to scale mismatch challenges (i.e. whether actors managing at different scales tended to collaborate or not; Table 4.1c), I found evidence to suggest that the Fitz-Stirling conservation initiative is promoting cross-scale collaborative management (co-management Armitage et al., 2007; Berkes, 2009; Carlsson and Berkes, 2005). While I found a general tendency to avoid locations managed by others, I found that actors operating at different scales tended to share locations, and also that actors sharing a location in general tend to collaborate. These results demonstrate that the Fitz-Stirling conservation initiative is a true empirical example of co-management. This finding is illustrated by invasive species management activities undertaken as part of the Red Card for Rabbits and Foxes program, which have involved diverse stakeholders from regional coordinators to local landholders (Tulloch et al., 2014). However results suggest that, for this case, co-management has not been able to solve all the challenges associated to the problem of fit.

In this chapter I show that the consideration of social-ecological interdependencies can lead to more thorough assessments of the propensity for collaborative approaches to address fit challenges in social-ecological systems. In Chapter 3 I found evidence indicating cross-scale management. However, in that study the ecological system was not considered and an assessment of cross-scale management was limited to the governance system. By considering social-ecological interdependencies in this chapter I was able to assess how the governance structure aligns to the ecological system and whether connections between ecological units are associated to collaboration between actors at different scales of management.

By linking the results from the social-ecological network analysis with qualitative data gathered through interviews, I provide tentative support for my hypothesised relationships between the level of fit (between the observed structure of collaborative relationships and the structure of the ecological system) and governance capacity to deal with the associated fit challenges (Table 4.1). I conclude that the collaborative approach of the Fitz-Stirling conservation initiative shows the capacity to deal with the fit challenges associated with the shared management of ecological resources, however, it lacks capacity to detect the effects of management actions applied in particular locations but that affect outcomes at connected locations. The ability to deal with this challenge would likely be enhanced by improving collaboration amongst actors working in connected locations (Table 4.1, bb2). This would allow them to increase opportunities for detection of feedback loops between actions and outcomes, thereby increasing the likelihood that the governance system can respond to the effects of adverse actions occurring beyond the ecological resource being managed.

Overall, results of this empirical research support the idea that approaches promoting actors to engage in collaboration with various others can create governance systems able to deal with the problem of fit. This framework could be expanded to classify, capture and explicitly define a greater diversity of fit challenges in social-ecological network terms. For example, diverse social-ecological network configurations can be used to theorise when a governance arrangement is better structured to address temporal mismatch challenges (Cumming et al., 2006). Temporal dimensions could be captured by assigning them as attributes to the social and ecological components of networks (i.e. nodes). Doing so would provide a theoretical basis for future testing, and facilitate future systematic research on temporal dimensions of the problem of fit. In addition, current developments on Exponential Radom Graph Modelling methods are limited to a subset of social-ecological network configurations. The incorporation of more elaborate social-ecological configurations would allow fit challenges to be captured in a more comprehensive way. For example, the effect of a threat occurring at a particular scale could be significantly different to the effects of the same threat occurring at a different scale (e.g. the effects of broad scale versus localised grazing on habitat connectivity; Cattarinno et al., 2014). To analyse such issues configurations are required to describe different governance levels, and the connections between different management and ecological scales. Ongoing development of analytical approaches is required so that a wide diversity of fit challenges can be explored. The results of such analyses can then provide an evidence-base for developing management recommendations to address fit challenges.

My approach can be replicated and applied in different contexts and thus similar studies in other areas and contexts are possible. In addition, combining this approach with qualitative assessments of the governance process can elucidate how other factors can affect the effectiveness of collaborative governance approaches. This can include assessments of the effect of costs and barriers to collaboration on the ability to align governance structures to ecological systems (e.g. Wyborn, 2014), and factors such as the quality of relationships (Lauber et al., 2011), the qualities of key individuals (Harrington et al., 2006; Keys et al., 2009; Shackleton et al., 2009), and other aspects such as power imbalances and trust issues (Adger et al., 2005; Hahn et al., 2006) that affect the effectiveness of diverse governance arrangements.

4.7 Conclusion

Social-ecological systems are inherently interconnected, so assessments of the performance of conservation programs should not be limited to only a subset of the system (i.e. only social or ecological dimensions). Furthermore, responses to global environmental problems require effective governance approaches that can respond to interdependencies that exist between social and ecological systems. I expand a recently proposed framework and methodological approaches for quantitatively analysing these interdependencies. I demonstrate their application through an empirical assessment of the degree to which a governance system fits (or aligns with) the characteristics of the biophysical system. By permitting analyses of the interdependencies between social and ecological systems, I provide a more complete and accurate assessment of collaborative approaches to governance. The approach employed and results derived from my study offers great potential for improving the structure of governance arrangements such that the effectiveness of environmental programs is enhanced. For my study region, the promotion of collaboration between actors working in ecologically connected areas would likely improve the effectiveness of on-ground management actions.

Chapter 5. Analysing social-ecological interactions to determine opportunities for conservation

Submitted to *Biological Conservation* as: Guerrero, A. M. & K.A Wilson. "Informing implementation strategies for conservation using a social-ecological systems framework".

5.1 Abstract

One of the key determinants of success in biodiversity conservation is how well conservation planning decisions account for the social system in which actions are to be implemented. Understanding elements of how the social and ecological systems interact can help identify opportunities and barriers to implementation. Utilising data from a large-scale conservation initiative in the south west of Australia I demonstrate how a social-ecological system framework can be applied to guide the integration of a variety of contextual factors that influence the opportunities for conservation. I identified areas that could benefit from different implementation strategies, from those suitable for immediate engagement to areas requiring implementation over the longer term in order to increase on-the-ground capacity and identify mechanisms to incentivise implementation. The systematic consideration and integration of ecological and social data can inform the translation of priorities for action into implementation strategies that account for the complexities of conservation problems in a focused way.

5.2 Introduction

The past two decades have seen an increase in the application of systematic techniques for informing decisions about better ways to reduce biodiversity declines and protect and conserve natural values. These techniques are applied within a conservation planning framework (Margules and Pressey, 2000; Pressey and Bottrill, 2009), to inform the selection of priority actions (associated with species and/or areas), or the allocation of resources (e.g. amongst multiple actions), to enhance objectivity, transparency, and scientific defensibility, and maximise the outcomes achieved with limited financial resources (e.g. Murdoch et al., 2007; Wilson et al., 2006). It is also increasingly recognised that to be effective, conservation decisions must be cognisant of the social and institutional context in which actions are to be implemented (Armitage et al., 2012; Cowling and Wilhelm-Rechmann, 2007; Robinson, 2006). Factors such as competing social values and objectives, political agendas, social norms, organisational and governance processes and technological and financial constraints can all facilitate (or inhibit) the implementation of conservation programs but are not commonly considered in conservation plans (Bryan et al., 2011; Mascia et al., 2003; Pannell et al., 2006).

While the importance of the social and institutional context of conservation is acknowledged in the conservation planning literature (Cowling and Wilhelm-Rechmann 2007; Knight et al., 2006b), the mechanisms for accounting for this context reflect an ad hoc collection of technical solutions (Table 1.1). Most studies focus on the use of spatial data related to threats or costs for the identification of priorities (Armsworth 2014; Naidoo et al., 2006). A few studies have considered social characteristics to identify areas of conservation feasibility – areas where conservation actions are more likely to be successful – on the assumption that prioritising these areas will increase the effectiveness of conservation investments (e.g. Adams et al., 2014; Guerrero et al., 2010; Knight et al., 2010; Mills et al., 2013; Sewall et al., 2011; Tulloch et al., 2014; Whitehead et al., 2014). Less attention has been given to utilising social data to inform implementation strategies, which has been focused on identifying where the values of the community align (or otherwise) with scientifically defined ecological values (Bryan et al., 2011). Studies that adopt a systems approach, accounting for interactions between social and ecological factors are increasingly found in the natural resource management literature but are only starting to appear in the conservation planning literature (e.g. Ban et al., 2013; Mills et al., 2013; Palomo et al., 2014). The integration of conservation planning with a social-ecological systems framework has been proposed (Ban et al., 2013), but practical examples of how this can be achieved are not available.

The social-ecological system (also referred to as the human-environment system) is a complex and perpetually dynamic system defined by several spatial, temporal, and organisational scales (Redman et al., 2004). In a social-ecological system, elements of the social system, including actors (e.g. government and non-government organisations, resource users, civil society) and institutions (e.g. rules and regulations, formal and informal procedures, policy instruments) interact with one another and with elements of the ecological system to regulate a continual interchange of inputs (e.g. land/resource use, management actions) and outputs (e.g. harvest, cultural or biodiversity values and other ecosystem services; Berkes et al., 2003; Redman et al., 2004).

In this chapter I explore an option to capture the social-ecological system in conservation planning. I demonstrate how the Social-Ecological Systems framework (SES) can be applied to inform the development of implementation strategies for conservation programs, specifically through the identification of opportunities for conservation (Figure 5.1). Conservation opportunity encompasses different biophysical, social, political and economic factors (Moon et al., 2014). I focus on implementation capacity (the capacity to put into practice and activity or program) and ecological data reflecting the importance of vegetation patches for enhancing connectivity from a large-scale conservation initiative in Western Australia.

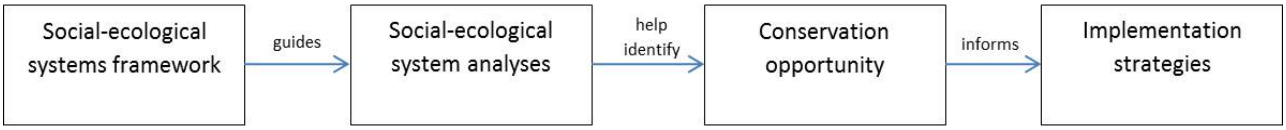


Figure 5.1. Informing implementation strategies. Implementation strategies can be informed by social-ecological analyses through the identification of conservation opportunity.

5.3 The Socio-Ecological Systems Framework and its relevance to conservation planning

There are a number of prominent frameworks used for conceptualising social-ecological systems. These include the Social-Ecological Systems framework (SES), the Press–Pulse Dynamics framework (PPD), and management strategy evaluation framework (MSE) (Bunnefeld et al., 2011; Collins et al., 2010; Ostrom 2007). These frameworks differ in terms of their disciplinary background and their applicability, but not many explicitly account for social-ecological interactions and their dynamics (Binder et al., 2013).

Here I apply the SES framework (McGinnis and Ostrom 2014; Ostrom 2007), which explicitly accounts for social-ecological interactions. The SES framework has been developed after decades of investigation on the key components of social-ecological systems, and the critical relationships amongst these, that are relevant to explaining outcomes in natural resource management. This has resulted in an extensive multi-tiered hierarchy of variables aimed at providing a common language for the analysis of social-ecological systems (Figure 5.2). I propose this interdisciplinary framework can be useful for guiding the systematic identification of social and ecological factors to be included in conservation planning studies. Specifically, the framework can help organise the conservation planning task by directing attention to the variables affecting key social-ecological interactions and outcomes in the social-ecological system of interest.

The SES framework organises potential relevant social and ecological factors into six distinct components (Figure 5.2). The resource system (RS) and the resource services and units (RU) include variables that describe the ecological system under study. The actors (A) subsystem include variables associated with the stakeholders that influence outcomes (O) (e.g. sustainability, livelihood and biodiversity outcomes) in the social-ecological system of interest. The governance system (GS) includes variables that describe the formal and informal mechanisms in place for management of natural resources and the conservation of biodiversity values. The two external subsystems include variables related to the external social, economic and political setting affecting outcomes in the focal system (e.g. market incentives, government policies) and potential wider ecosystem variables of relevance such as climate patterns. The SES framework proposes that variables in these different subsystems interact in an action situation to produce observed social-ecological outcomes. This action situation involves interactions (I) between actors who jointly affect outcomes, and interactions between the social and the ecological system, which are specified by the range of activities in which actors are engaged (e.g. harvesting and monitoring activities; McGinnis and Ostrom, 2014). I propose that there are action situations involving interactions related to conservation and management activities that also influence social-ecological outcomes (e.g. restoration activities, invasive species management activities, species reintroductions).

The hierarchical organisation of the SES framework permits different degrees of specificity in the analysis of social-ecological systems, which constitutes an advantage of this framework over other frameworks (Binder et al., 2013). The multi-tiered hierarchy of the SES framework can also facilitate the identification of the key social and ecological factors affecting biodiversity outcomes in a particular system, and the selection of key variables for analysis. As such, the SES framework can be used to help understand and unpack the complexity of social-ecological systems for integration with broader conservation planning processes.

Ban et al. (2013) suggested different ways in which the SES framework can benefit systematic conservation planning, and three dominant ways are outlined in Figure 5.3. First, both social and ecological data can be used to conceptualise the natural and human aspects of conservation problems (and conservation solutions) as a single complex system. This can enhance the identification and articulation of conservation objectives, and facilitate the explicit consideration of the needs and values of different stakeholders. Second, it can enhance prioritisation analyses through the identification of key factors that influence conservation outcomes but are not commonly considered. For example, governance factors, such as the level of political stability and corruption (McCreless et al., 2013) and stakeholders' location, attitudes and collaboration capacity (Bode et al., 2010; Guerrero et al., 2015; Knight et al., 2010). Third, and the focus of this chapter, is through the identification of areas that represent distinctive opportunity for implementing conservation programs to inform the development of implementation strategies.

Implementation strategies are an essential, yet uncommon, component of conservation plans (Cowling et al., 2008; Knight et al., 2006a; Knight et al., 2006b; Pressey and Bottrill 2009). While conservation prioritisation analyses identify what we should conserve and the required conservation actions (e.g. land protection, revegetation, invasive species control), implementation strategies identify how we can execute those actions (e.g. through direct engagement, political or financial support, collaboration strategies, education campaigns, marketing and communication strategies, financial and market-based incentives, or a combination of these). Implementation strategies reflect the available local resources and local modes of operation, are of direct relevance to stakeholders, and link directly to the activities of implementing organisations (Del Campo and Wali, 2007; Pierce et al., 2005). For example they can inform the timing of actions, and target implementation effort (Bryan et al., 2011). They can also target stakeholders that are well informed about, and aligned with, key aspects of the ecological system (Guerrero et al., 2013; Prell et al., 2009; Vance-Borland and Holley 2011). They can involve communication strategies that emphasise the benefits for people's livelihoods rather than conservation benefits alone, such as financial benefits, for example through job creation and tourism activities (e.g. human-wildlife conflict mitigation programs; Henson et al., 2009). It is only through analysing the connections between ecological and social factors, and not through analysing individual subsystems (e.g. ecological or the social) that opportunities (and barriers) for implementation of conservation actions can be identified; and this represents one of the main utilities of a social-ecological systems framework for conservation planning.

5.4 Applying the Social-ecological Systems framework to inform implementation strategies for conservation

5.4.1 The Fitz-Stirling case study

The Fitz-Stirling region covers over 240,000 hectares and is part of a large-scale conservation initiative that aims to restore ecological connectivity along a 1,000 kilometre corridor in south-western Australia (Bradby, 2013). Multiple stakeholders are involved in efforts to achieve conservation objectives for the Fitz-Stirling including property owners, state and local government agencies, regional natural resource management groups, non-government organisations, community groups, university and research organisations, private organisations and independent contractors (Guerrero et al., 2015). These stakeholders engage in diverse activities, including revegetation, protection of bushland, invasive species management, livestock management, fire management and land use planning.

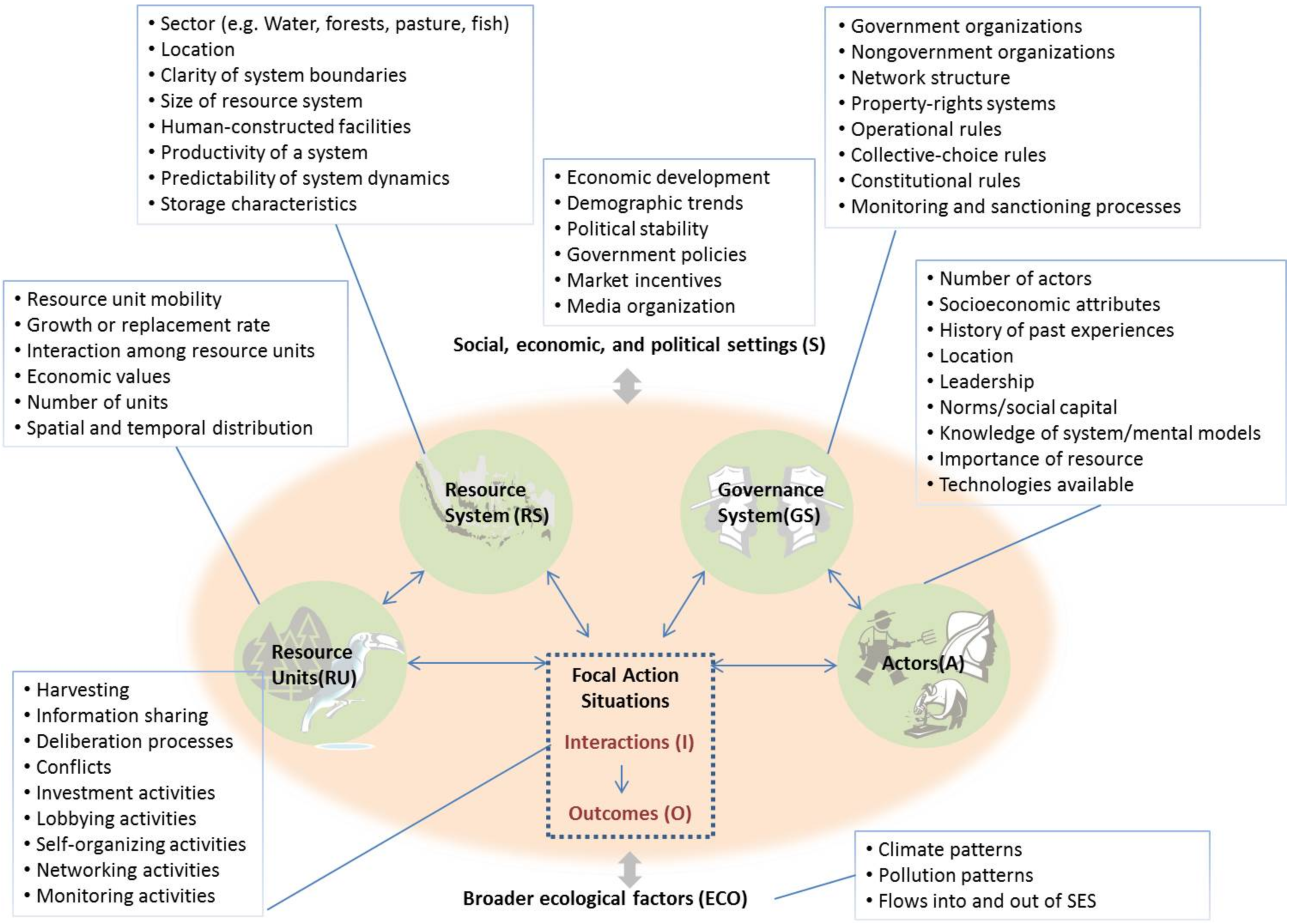


Figure 5.2. General framework for analysing a social-ecological system. Boxes depict the social and ecological factors that can affect sustainability, livelihood and biodiversity outcomes, at multiple ecological scales (e.g. habitat, landscape) and socio-political scales (e.g. local, regional, national, global). Arrows depict how the different subsystems (RU, RS, GS and A) interact in a focal action situation. Interactions (I) influence different types of outcomes (O) - including biodiversity outcomes. Figure is based on Ostrom (2007, 2009) and McGinnis and Ostrom (2014).

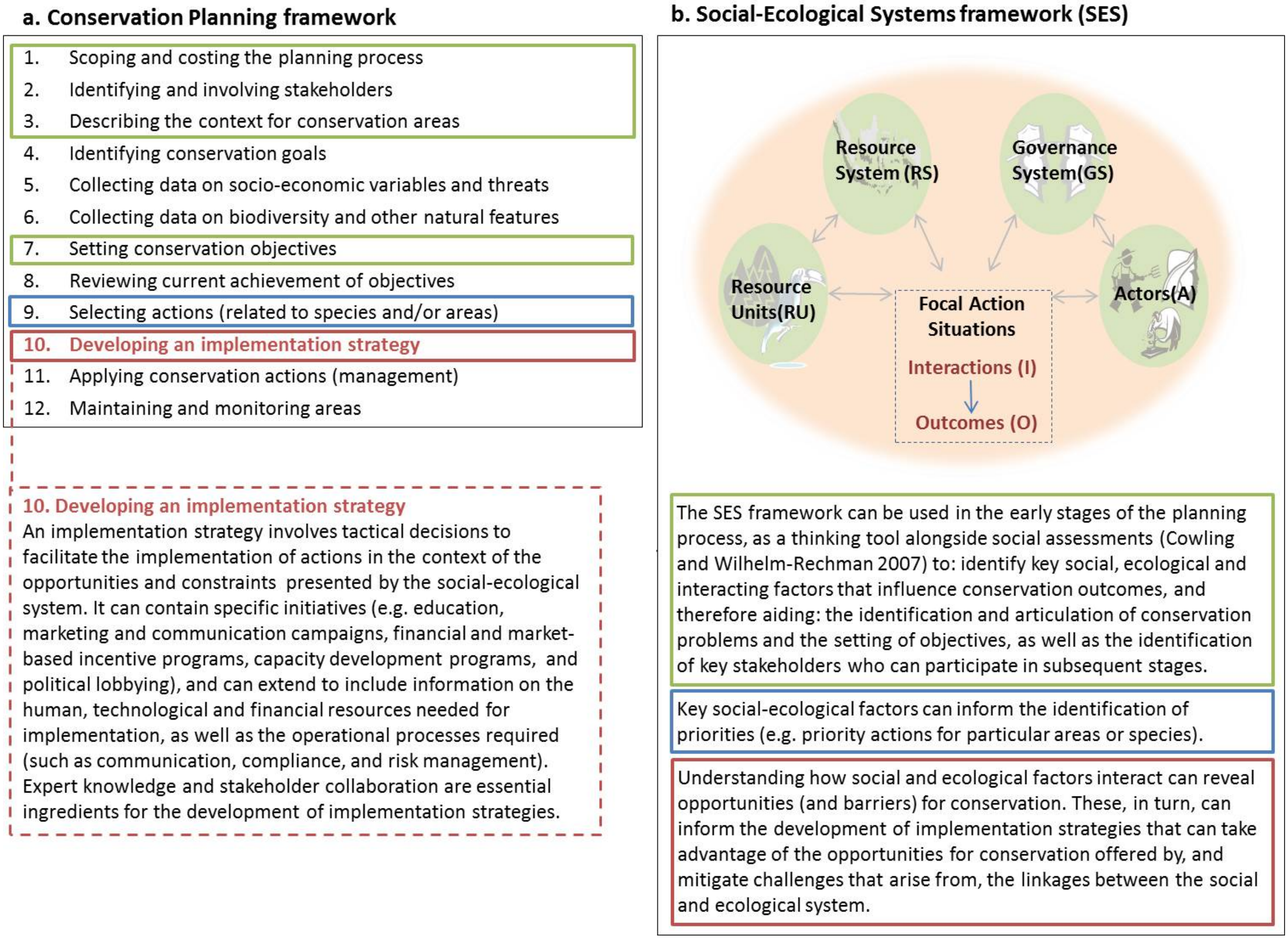


Figure 5.3. Three uses of a social-ecological systems framework in systematic conservation planning. (a) has been adapted from Pressey and Bottrill (2009) to include a new stage (stage 10), based on Driver (2003), Pierce et al. (2005), Knight et al. (2006a), Knight (2006b), and Cowling et al. (2008). See Figure 5.2 for a detailed view of (b).

5.4.2 Applying the Social-Ecological Systems Framework

I conceptualised the Fitz-Stirling region as a social-ecological system, where the focal system of analysis is the broader landscape (i.e. matrix of remnant vegetation plus agricultural land). The resource system (RS) is defined as the entire Fitz-Stirling region, consisting of land used for agriculture and livestock production and land designated for conservation purposes, and the resource services and units (RU) are the remnants of native vegetation scattered through the landscape (totalling approximately 24,000 hectares). The actors (A) subsystem is composed of the stakeholders that influence conservation outcomes through land use and conservation activities in the Fitz-Stirling region, including landholders, government and non-government organisations and other stakeholder groups. The governance system (GS) includes the governance networks that relate to land use management and conservation (specifically, organisational partnerships or community group collaborations). Finally, my focal action situation is the implementation of conservation activities such as revegetation, protection of bushland and invasive species management.

The first step in applying the SES framework to a particular case is the selection of the variables in the SES framework (actors, governance systems, resource system and units, Figure 5.2) that interact in the focal action situation and are the most relevant to the particular conservation problem and the objective of the analysis being undertaken (McGinnis and Ostrom, 2014). Here I seek to identify opportunities and challenges for the implementation of conservation activities in the Fitz-Stirling region.

5.4.2.1 Ecological system assessment

There are at least two variables in the resource services and units (RU) sub-system that can be deemed important for guiding the implementation of conservation activities in the Fitz-Stirling region. These are the spatial distribution of the vegetation patches (ecological units) in the landscape and their connection with other vegetation patches. These two variables can be measured using different metrics, including how well remnants are connected to facilitate dispersal of species across the landscape (Moilanen and Nieminen, 2002), otherwise referred to as functional connectivity (Saura and Rubio, 2010).

I used publicly available data on the distribution of native vegetation in the Fitz-Stirling region. Over 2000 distinct vegetation patches (ecological units) were identified. My methods required survey respondents to indicate on a map the vegetation patches in which they applied their conservation and management activities (Appendix A, Table A5). To make this feasible, the vegetation patches were clustered based on a 0.5 km dispersal threshold. This threshold was chosen since many bird and mammal species would view any two patches within a 500 meter range to be well connected and therefore effectively seen as one coherent area of habitat, or a “meta-patch” (Sutherland et al., 2000; Zetterberg et al., 2010). This resulted in 80 vegetation clusters.

Functional connectivity was analysed using a 1km threshold. The threshold is ultimately a species-specific measure. However, I selected a 1 km threshold since then I am able to describe the landscape’s level of connectivity for a fairly broad range of species. The selected threshold is relevant for many bird species, as well as for several mammal and amphibian species (Bergsten et al., 2013; Sutherland et al., 2000; Zetterberg et al., 2010). But small mammal and insect species would require smaller thresholds that better reflect their dispersal across the landscape, and larger mammals would require greater thresholds (Sutherland et al., 2000). Based on the methods of Saura and Rubio (2010) a probability of connectivity metric (PC) was calculated for each of the 80 vegetation clusters to quantify their relative importance to overall habitat connectivity.

5.4.2.2 Social system assessment

I conducted a social system assessment to identify aspects of the actors (A) and governance (G) sub-systems that could guide the identification of opportunities for implementing conservation actions in the Fitz-Stirling region. I focused on identifying aspects related to the capacity of an organisation or group of individuals to put into practice an activity or program (here on referred to as implementation capacity).

The factors that are thought to determine implementation capacity vary across different natural, social and health sciences disciplines, and are a combination of institutional, psychological, economic and organisational factors (Brown 2008; Katsuhama and Grigg 2010; Martin et al., 2009; Mountjoy et al., 2014). For conservation programs, implementation capacity is often associated with political and institutional support, access to financial and human resources, and the extent of collaboration amongst implementing organisations – particularly across sectors and scales (Fitzsimons et al., 2013; Guerrero et al., 2015; Hall 2008; Jameson et al., 2002; Pasquini et al., 2011). Collaboration also increases social capital and thus the ability to harness both resources and support (Cramb 2006; Njuki et al., 2008; Pretty and Ward 2001). Addressing complex problems – such as those associated with conservation programs – not only requires multiple stakeholders to contribute to implementation (i.e. contribute human resources) but it also requires coordinated action. Collaboration enables coordinated action to occur (Brondizio et al., 2009; Lubell et al., 2002; Weiner, 2009). Importantly, for conservation problems transcending jurisdictional and ecological boundaries, successful programs require that actions are coordinated across management scales (Dallimer and Strange 2015; Epstein et al., 2015; Ostrom 2010; Young et al., 2008).

Using semi-structured interviews and an online questionnaire I collected qualitative and quantitative information to identify key aspects influencing implementation capacity in the Fitz-Stirling region (see Appendix A for further detail). The importance of collaboration in this region is supported by data on the perceptions held by Fitz-Stirling stakeholders on the factors that influence on-the-ground activities. For example, insufficient communication and coordination was highlighted as an important barrier to successfully carrying out conservation activities across the region (Appendix A, Figure A2). Nonetheless, of the collaborative relationships identified 83% were perceived to deliver results to the particular activity performed at particular locations (Appendix A, Figure A3). Results also suggest that accessibility to human resources is a key aspect of implementation capacity in the Fitz-Stirling region (Appendix A, Figure A2). In addition, the perceived need for greater communication and coordination to support implementation appears to be particularly important across organisational levels (e.g. "between government agencies and NGOs and landholders"). This is further supported by perceptions of the value of collaboration with government agencies, with only 9% of non-government stakeholders identifying such collaborations to be of low or no value (Appendix A, Figure A4).

These results point to at least three variables in the in the Actors (A) and Governance (G) sub-systems that influence the implementation of conservation activities in the Fitz-Stirling region. These are the proportion of stakeholders working in each vegetation remnant (i.e. the human resource aspect), the level of connectedness of stakeholders (i.e. the collaboration aspect), and their scale of management. To obtain a measure of each of these variables I asked survey respondents to indicate on a map the vegetation patches in which they applied their conservation and management activities, this gave me an indication of stakeholder presence in each vegetation patch. I also asked who they collaborate with for performing different conservation activities and using the degree centrality metric (Borgatti et al., 2009) I measured their level of connectedness (Appendix A for further details). By coding stakeholders by scale of interest (Appendix A, Figure A1) I was able to identify areas associated with multiple scales of management.

5.4.3 A social-ecological analysis for the development of implementation strategies

I utilised the results of the ecological system assessment (Section 5.4.2.1) and the results of social system assessment (Section 5.4.2.2) to conduct a social-ecological analysis. Specifically, I sought to identify areas of varying conservation opportunity to inform the development of implementation strategies (Figure 5.4). Areas of high ecological importance and high stakeholder presence (A in Figure 5.4) represent areas of existing opportunity (Moon et al., 2014) where conservation actions can be implemented in the near term and can thus benefit from immediate action, engaging with existing stakeholders to coordinate required conservation actions (e.g. revegetation, invasive species management). This might require the provision of financial, technological or knowledge to support current activities, or the development of partnerships and agreements to enable coordination of on-the-ground actions and support management of areas across multiple scales (red circles in A, Figure 5.4). Areas of low ecological importance and high implementation capacity (B in Figure 5.4) are areas that can lead to inefficiencies in efforts to achieving outcomes, and can thus benefit from communication and education strategies, and the sharing of knowledge to increase stakeholders' awareness of areas of higher ecological importance where they can redirect their efforts. Areas of low ecological importance and low implementation capacity (C in Figure 5.4) are currently areas of low priority and unlikely to require immediate attention. Finally, areas of high ecological importance and low implementation capacity (D in Figure 5.4) represent areas of potential conservation opportunity (Moon et al., 2014) where conservation outcomes could be difficult to achieve in the near term. These areas can benefit from a longer-term implementation strategy aimed at increasing on-the-ground capacity combined with incentives for conservation. This might entail diverse activities such as education campaigns to promote the ecological importance of the areas, forming or enhancing organisations capable of implementing actions, and the development of conservation incentive instruments (e.g. covenants). This might be achieved by harnessing the social capital of areas displaying high levels of social connectedness (larger circles in D, Figure 5.4). In addition, the success of conservation efforts could be maximised by focusing efforts on those areas associated to multiple scales of management (green circles in D, Figure 5.4).

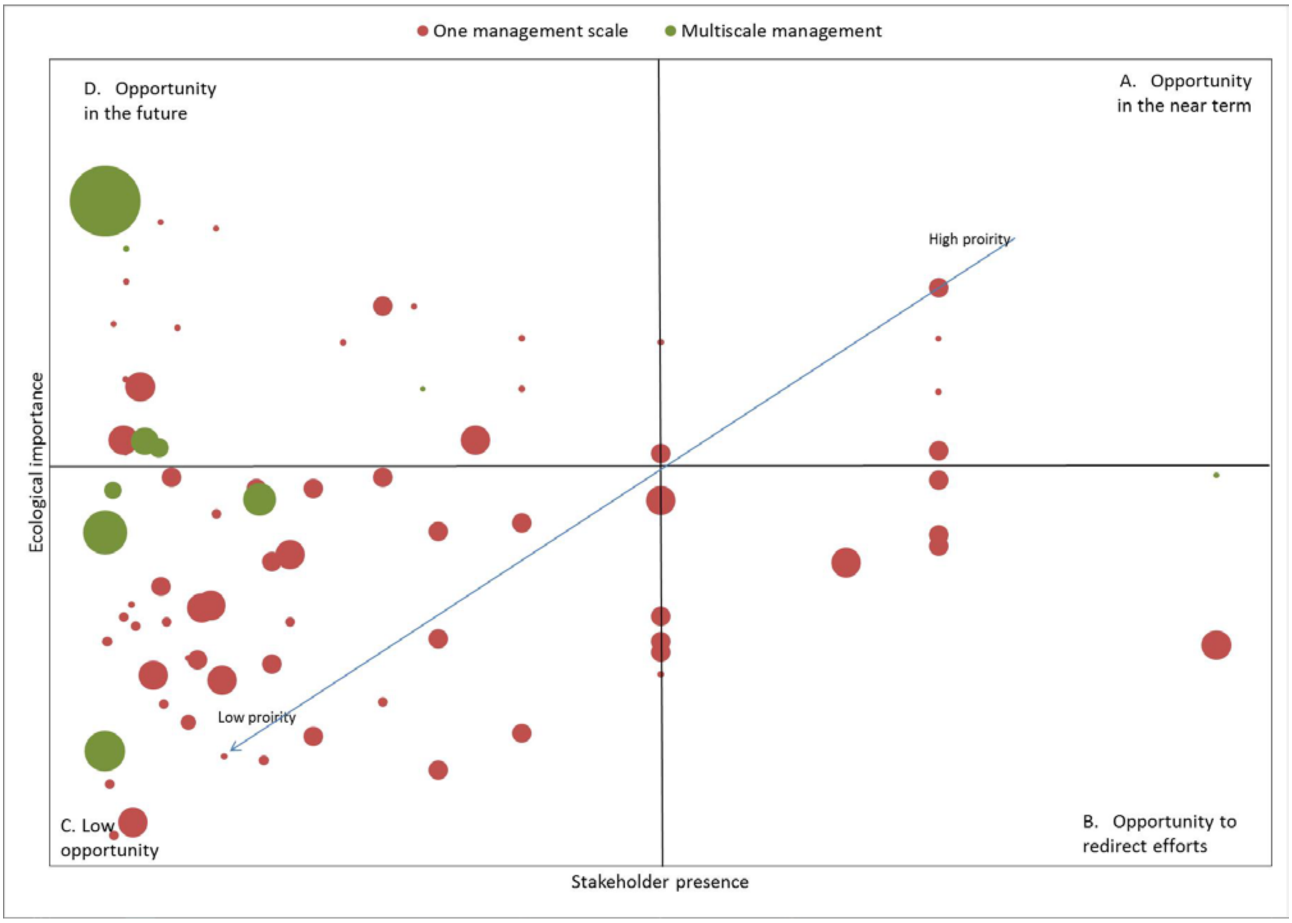


Figure 5.4. Areas of distinctive conservation opportunity. Circles represent vegetation clusters with different levels of ecological importance and implementation capacity. Implementation capacity metrics include stakeholder presence (proportion of stakeholders working in the area), social connectedness – measured by a degree centrality metric (total number of collaboration relations pertinent to each cluster), and the scale of management. Size of circles denotes the social connectedness associated to each area, from low connectedness (small circles) to high connectedness (big circles).

5.5 Enhancing current conservation planning approaches

The systematic consideration and integration of social and ecological data can enhance current approaches to conservation planning, by explicitly accounting for interactions between the social and ecological systems in the identification of conservation opportunities. For the Fitz-Stirling region, a standard approach may result in the prioritisation of areas of high ecological importance but some of these areas may also have low capacity for implementation (D in Figure 5.4) – which could affect the effectiveness of conservation efforts. In addition, a poor understanding of the interactions between the social and ecological system would likely result in inadequate implementation strategies that fail to respond to the opportunities and challenges identified. For the Fitz-Stirling this could result in, for example, the provision of financial or other type of resources to implement activities in areas where the implementation capacity is currently limited. This could include areas where stakeholders are not ready or lack commitment to achieving conservation outcomes, which could in turn lead to failed implementation, delays and inefficient use of valuable resources.

5.6 Limitations and future directions

The main objective of this study was to illustrate how the SES framework can be applied to conservation planning, specifically to extend the use of social and ecological data from identifying conservation priorities alone, to identifying conservation opportunities to inform the development of implementation strategies. My social and ecological assessments focused on identifying indicators of conservation opportunity. In order to undertake similar analyses in other socio-ecological systems or focal action situations it is likely that different ecological and social assessment approaches would be required. For example, in the Fitz-Stirling region there are likely to be variables in the actor (A) or governance (GS) subsystems that influence conservation outcomes and would necessitate a different form of social assessment and use of different indicators to the ones I have employed. For instance, in the actor subsystem it is likely that the willingness of landholders to participate in a stewardship program would be a key indicator of conservation opportunity and the resultant social assessment would aim to identify areas of implementation readiness (Knight et al., 2010). Situating conservation analyses into a social-ecological framework, as I do here, helps guide the identification of the most relevant indicators given the nature of the conservation problem being explored.

It would be feasible to incorporate other variables of the resource services and units (RU) sub-system (Figure 5.2) such as more detailed biodiversity and ecosystem data than included in my example. Likewise, assessments of implementation capacity could include variables of the actors and governance sub-systems not considered in my example if these were found to be relevant to the particular social-ecological system being studied (e.g. leadership, access to financial resources and technology). In addition, variables from the related ecosystems (ECO) sub-system such as the impact of climate change on the components of the socio-ecological system could be captured if such dynamics were perceived to be important for designing implementation strategies (e.g. Faleiro et al., 2013). Application of the social-ecological framework with a focus on social-ecological interactions enables the conservation researcher or practitioner to disentangle the complexity of conservation problems to focus on the aspects that warrant closer investigation.

5.7 Conclusions

Implementation strategies are an essential, yet uncommon, component of conservation plans (Knight et al., 2006a; Reyers et al., 2010). I show how ecological and social data can be combined to identify priorities, opportunities, and potential challenges to conservation and how this information can inform the formulation of implementation strategies. I identified areas requiring different implementation strategies through integrating conservation planning with the knowledge and analyses from each component of a social-ecological systems framework. I reveal the potential to increase the power of information drawn from ecological or social data. The result of this approach is the identification of areas that can be targeted in different ways, from high opportunity areas that are suitable for immediate engagement in conservation activities, to areas at risk of failure requiring a mix of strategies in order to facilitate conservation over a longer-term.

Chapter 6. Discussion

This doctoral thesis addresses a critical gap in the field of conservation science: accounting for the social-ecological context of conservation problems. Social-ecological research is increasingly considered a vital component of the research agenda for environmental problems (Ostrom et al., 1999; Rapport et al., 1998; Rockstrom et al., 2009). Understanding how stakeholders, use, manage, influence, and are influenced by different elements of the natural environment and by their own interactions can reveal opportunities for conservation and inform the design of effective governance arrangements. Thus, while knowledge on ecological systems is essential to the formation of *ideal* solutions to conservation problems, formulating *viable* and *feasible* solutions depends also on knowledge of the social-ecological context. Social-ecological analyses can lead to important insights that can improve the effectiveness of conservation efforts.

This thesis addresses several gaps. First, the number of conservation planning studies incorporating social factors is growing, but studies that account for social-ecological interdependencies and evaluate governance arrangements are rare. Second, the conceptual and analytical mechanisms for accounting for this context remain underdeveloped. And third, given the diversity of social factors that affect the effectiveness of conservation efforts, new frameworks and methodological approaches are needed to aid understanding of the social-ecological system and facilitate the selection of appropriate variables for conservation planning analyses.

The research that I have undertaken advances the field of conservation science by furthering understanding of key challenges to conservation that can only be understood from a social-ecological system perspective – the ‘fit’ between the governance system and the biophysical system of interest (Chapter 2); identifying how collaborative governance approaches to biodiversity conservation can address some of these challenges (Chapters 3 and 4); demonstrating research approaches to investigate different aspects of ‘fit’ and the potential for collaborative governance to overcome fit challenges (Chapters 3 and 4); and demonstrating how implementation strategies for conservation can be informed by social-ecological analyses (Chapter 5). In this section I synthesise the main results drawn from one or more chapters and identify and discuss priorities for future research.

6.1 Understanding and addressing the problem of 'fit'

Important challenges arise when governance systems do not fit the biophysical system of interest when addressing complex environmental problems such as the loss of biodiversity. An example is when the complexity and scale of the problem (e.g. the dispersal of invasive species, climate change) requires multiple stakeholders to collaborate so that management decisions can match the scale(s) of key biophysical processes. These challenges are common, but are not usually acknowledged in conservation planning studies. This thesis shows how a greater understanding of 'fit challenges' can inform strategic decisions at various stages of the planning and implementation process to ameliorate 'misfit' (Chapter 2). I identify ways that fit challenges can affect conservation outcomes, highlighting that not addressing them can result in ineffective actions, partial solutions that do not embrace the full extent of the problem, or solutions that simply fail to be implemented.

Social network analyses can be useful for assessing how networks of governance actors in a conservation setting can address fit challenges. However, some of the methods available used techniques with significant limitations in practical application (Carrington et al., 2005; Mills et al., 2014). For example, most social network studies in the conservation and natural resource management literature apply descriptive statistics, such as measures of centrality (Bodin and Crona, 2009; Cohen et al., 2012; Garcia-Amado et al., 2012) or frequency distributions (e.g. Bodin et al 2013; Bergsten et al 2014). Such methods require complete network datasets that sample the entire population, which can be difficult to achieve. Using partial network data can significantly reduce the stability of centrality measures, introduce biases that increase the perceived centrality of surveyed individuals, and lead to central hubs of network actors being overlooked (Costenbader and Valente, 2003; Kossinets, 2006). In contrast, statistical methods such as the one applied in this thesis (i.e. Exponential Random Graph Models; Robins et al., 2007), treat network connections as a statistical sample, and thus represent a robust way to deal with partial network data. Some studies suggest that data collection methods that allow respondents to nominate several connections can enhance the accuracy of networks, and that snowball sampling can mitigate biases (Lee et al., 2006). Both these data collection methods were also applied in thesis.

A framework proposed by Bodin and Tengo (2013) permits explicit representations of how social and ecological systems interact (in social-ecological network terms), and make it possible to connect these representations to theory of diverse governance challenges, including fit challenges (Bodin et al., 2014). This framework is a structured way to describe and organise different elements of a social-ecological system, with particular focus on their interconnections (Bodin and Tengo, 2012). The integration of this framework with recently developed models for multilevel networks (Wang et al., 2013) makes it possible for these theories to be empirically tested. This integration is explained by Bodin and others (in review), and Chapter 4 of this thesis is the first detailed study applying this approach to investigate the problem of fit.

Using these novel network research methods, I demonstrate how social network analyses can be used to understand the propensity for governance arrangements (networks of governance actors) in a conservation initiative to address fit challenges. Chapters 3 and 4 demonstrate how the structure of their interactions across spatial scales (Chapter 3) and their alignment to the ecological network in a landscape (Chapter 4) can be assessed. The results reveal how the particular governance arrangement studied is well structured to address some fit challenges (e.g. to enable invasive species management across scales) but not others (e.g. management of interconnected ecological units). Knowledge on the propensity for governance arrangements to address fit challenges provides an evidence base to inform future collaborative partnerships. For example, the information could be used to identify the key partnerships to maintain and the partnership gaps to be addressed, so as to facilitate the implementation of solutions at the required scales.

Future directions

There is a need to account for fit challenges when devising and implementing solutions to conservation problems. However, further theoretical and methodological developments are required to facilitate this. While the conceptual foundation has been considerably progressed in the last few decades (Cumming et al., 2006; Folke et al., 1998b, 2007; Galaz et al., 2008; Young et al., 2008), greater clarity is needed on what constitutes a fit challenge. Different types of fit challenges need to be more explicitly and clearly defined and evaluated. For example, scale mismatch is one aspect of the problem of fit, yet terms are often used interchangeably. While classification of spatial, temporal and functional scales have helped in conceptualisation of fit challenges (Cumming et al., 2006) it is still unclear what constitutes a fit challenge related to scale (e.g. management actions applied at either too broad or fine geographic scales) and how these differ from other types of challenges related to scale, for example those related to geographic extent (e.g.

management actions not covering the full extent of a river catchment). This lack of clarity may be partly driven by the varied interpretations of the concept of scale across the natural and social sciences (Higgins et al., 2012; Sayre, 2005).

Theoretical frameworks being developed for the analysis of social-ecological systems, such as the one proposed by Bodin and Tengo (2013), can be used to further research on the problem of fit. While chapter 4 demonstrated the use of this framework for the analysis of specific fit challenges, this framework could be expanded to classify, capture and explicitly define a greater diversity of fit challenges in social-ecological network terms. For example, diverse social-ecological network configurations can be used to theorise when a governance arrangement is better structured to address functional mismatch, or temporal mismatch challenges. Here the functional and temporal dimensions could be captured by assigning them as attributes to the social and ecological components of networks (i.e. nodes). Doing so would provide a theoretical basis for future testing, and facilitate future systematic research on the diverse aspects of problem of fit.

Exponential Radom Graph Modelling methods have advantages over more commonly used network analysis methods (i.e. descriptive methods), but are limited to a subset of social-ecological network configurations (e.g. Bergsten et al., 2014; Bodin and Tengo, 2012; Ekstrom and Young, 2009). The incorporation of more elaborate social-ecological configurations would allow fit challenges to be captured in a more comprehensive way. For example, the effect of a threat occurring at a particular scale could be significantly different to the effects of the same threat occurring at a different scale (e.g. the effects of broad scale versus localised grazing on habitat connectivity; Cattarinno et al., 2014). To analyse such issues configurations are required that describe different governance levels, and the connections between different management and ecological scales. Ongoing development of analytical approaches is required so that a wide diversity of fit challenges can be explored. The results of such analyses can then provide an evidence-base for developing management recommendations to address these challenges.

Understanding the propensity for governance arrangements in a conservation initiative to address fit challenges (Chapter 3 and 4) could inform a pre-emptive diagnosis of the likelihood of success of conservation efforts. This information could then be employed in prioritisation analyses as estimates of the likelihood of success of actions, which could guide the allocation of funds for conservation to particular locations or species. Conservation priority setting that accounts for the propensity of a governance arrangement to manage fit challenges could improve the efficient use of the limited available funds for conservation, and ensure that allocated funds result in effective actions.

6.2 Collaborative approaches to environmental governance

Polycentric, collaborative forms of decision-making are increasingly promoted over command and control approaches for the governance of environmental problems (Armitage et al., 2012; Folke et al., 2005; Ostrom, 2010b; Young et al., 2008), yet quantitative empirical research assessing this potential rarely link theories and insights on governance approaches to the features of the system under investigation (Huitema et al., 2009; Newig and Fritsch, 2009). This thesis provides the first quantitative empirical test of the potential for collaborative approaches to address governance challenges related to fit. Chapters 3 and 4 examined the potential for a contemporary conservation initiative, characterised by a collaborative governance arrangement, to address specific fit challenges. Results from these studies suggest the governance arrangement is well structured to deal with some fit challenges but not others, providing support to the idea that collaborative approaches such as co-management can increase the capacity of governance systems to deal with the problem of fit. Because this research is limited to one case study system, similar studies in other areas and contexts are needed to deliver generalised conclusions. The findings of this study however provide empirical support, rarely found in the literature, of the benefit of collaborative approaches to the governance of conservation programs in particular, and natural resources more generally.

Future directions

Further research is needed to progress understanding of the benefits of collaborative governance to conservation. While collaborative governance has the potential to facilitate the attainment of conservation goals, it can be difficult and costly to achieve (e.g. Wyborn, 2014). There is a need to understand the challenges related to establishing and maintaining collaborative arrangements, including the transaction costs associated with the time and effort required to engage with many different stakeholders (e.g. time spent on meetings, opportunity costs of participating stakeholders, travel costs), information costs (e.g. sharing of information) and decision-making process related costs (such as conflicting mandates, trust issues, dominant individuals and un-clear procedures; Blore et al., 2013; De Cremer and Stouten, 2003; Enengel et al., 2014; Marshall, 2013; McCann et al., 2005; Reeson et al., 2011; Widmark and Sandstrom, 2012). Measuring these costs and integrating them in assessments of governance arrangements would help determine the feasibility and worth of adopting a collaborative approach (Carr, 2013; Wyborn, 2014).

The most effective approaches to measuring and accounting for the costs associated with collaboration would first require exploration. Stakeholder analyses techniques, such as interviews or analysis of past records, are one avenue for identifying and deriving estimates of the costs associated with collaboration. Institutional scoping methods (e.g. scoring matrices, and power analysis tools; Maxwell, 2010; Mayers, 2005) currently used to capture the level of interest, capacity, effectiveness and role of prospective partners could be expanded to include a specific section on the likely costs associated with collaborating. Once measured, data on costs can be integrated into stakeholder selection processes, with the goal of ensuring that the net benefit delivered by collaborations is maximised.

Maximising the collaborative ties between all possible stakeholders does not necessarily lead to better solutions (Lazer and Friedman, 2007). Thus there is a need to go beyond simple prescriptions of governance arrangements that promote collaboration, and determine the type and extent of collaboration required for different tasks. For instance decision-making tasks (e.g. elicitation of objectives, selection of actions and policy decisions) may require different collaboration capacities to tasks related to the generation of knowledge (e.g. capturing of data), or implementation tasks (e.g. invasive species control activities, revegetation activities). This will likely require studying the array of collaboration types in a conservation setting (e.g. networking, cooperating, coordinating, collaborating, partnering) that can be placed along a continuum from those requiring less interaction and preservation of individual goals, to those requiring higher levels of interaction and shared goals and responsibilities (McNamara, 2012; Patton, 1994).

Determining the benefit of different types of collaboration to different decision-making and implementation tasks would benefit from social network research approaches. For example, the different collaboration types could be described in network terms, where nodes could represent different stakeholders, and the ties, the interactions between them. Different network measures could be used to characterise each collaboration type. For instance, density and frequency measures could be used to characterise collaboration types requiring high levels of stakeholder interaction (e.g. *partnering*) and centrality measures could be used to characterise *coordinating* types (Bodin and Crona, 2009; Freeman, 1977). Density measures could also be used to characterise the extent to which shared goals can be formed in different collaboration types (Coleman 1990). Two-mode networks could also be used to characterise different collaboration types, where the extent and type of participation of actors (first type of node) could be described for different decision-making forums (second type of network node; e.g. Berardo, 2014; McAllister et al., 2014). This network characterisation of collaboration types can then be used to investigate the types of collaborations needed for ensuring the success of different decision-making and implementation tasks. This could be informed by the analysis of network data for case studies where particular decision-making, knowledge generation or implementation tasks were proved successful.

6.3 Governance structure and performance

Assessing the performance of governance arrangements is an important emerging research area in biodiversity conservation (Rands et al., 2010), and related fields such as sustainability science and natural resource management (Henry and Vollan, 2014). This thesis has demonstrated how structural analyses can be used for assessing the performance of governance arrangements in terms of addressing fit challenges for whole programs and for particular conservation actions, and can be replicated and applied in different contexts. This methodological contribution extends beyond the conservation planning field, to research in the policy science field that study governance systems through a network lens (Carlsson and Sandstrom, 2008; Lubell et al., 2014; McAllister et al., 2014; Sandstrom and Carlsson, 2008), examine multi-scalar aspects of governance (Marshall, 2007; Morrison, 2007; Wyborn, 2014), and study the social and political processes by which governance structures are formed (Berardo and Scholz, 2010; Henry et al., 2010). In addition, by focusing on specific governance challenges (i.e. the coordination of actions across management scales and the alignment of actions with ecological systems) and by taking a structural perspective, this thesis complements research on environmental governance that emphasise the importance of non-structural aspects on performance, such as the quality of relationships (Lauber et al., 2011), the qualities of key individuals (e.g. leaders; Harrington et al., 2006; Keys et al., 2009; Shackleton et al., 2009), and other aspects such as power imbalances and trust issues (Adger et al., 2005; Hahn et al., 2006).

Future directions

Future assessments should consider structural aspects of governance in conjunction with non-structural aspects to enable a more complete assessment of the performance of governance arrangements in conservation settings. This would likely entail combining the network research methods employed here with qualitative data analysis techniques that capture the non-structure-related aspects of networks. This could include information on the characteristics of governance actors (e.g. leadership, authority) and contextual aspects of governance such as the resources available, and the level of credibility and perceived legitimacy of actors (Armitage et al., 2012; Newell et al., 2012). This might require combinations of in-depth analysis with quantitative network data, or careful design of survey questionnaires that can capture both structural and qualitative data. Qualitative data could be captured as attributes of network nodes (e.g. type of knowledge held by a governance actor, organisation type and role, gender, attitudes towards conservation, perceptions about others etc.) and analysed to identify their effects on performance relative to network structure.

An understanding of how the network structure affects performance can inform strategies for the development of governance arrangements and stakeholder engagement when designing conservation programs. But investigating this relationship requires understanding of what makes a governance arrangement effective. To do this, it can help to differentiate between governance-related outcomes and conservation-related outcomes (e.g. improved probability of species persistence). Conservation-related outcomes can be measured via biologically based indicators but often it is not feasible or cost-effective to deliver timely information (Gerber et al., 2000; Salafsky and Margoluis, 1999). Governance-related outcomes are easier to measure and can be further differentiated between outcomes at the program level or the individual level. For example, a program-level outcome could be defined as the overall level of collaboration among stakeholders. On the other hand, a node-level outcome can be a farmer participation in a voluntary conservation program or their adoption of conservation practices. Future studies on governance structures should focus on measuring effectiveness more explicitly, at either the program level or individual (node) level.

Research on how network structure affects performance can also be aided by understanding and accounting for the different types of costs (and expected benefits) associated with the establishment of governance arrangements (see section 6.2).

Accounting for these costs would enable research to determine for example, the most cost-effective governance structures given the costs involved in establishing and maintaining collaborative ties and the characteristics of the particular conservation challenge. Such research might benefit from experimental approaches given the difficulties related to obtaining social network data. For example agent-based modelling techniques could be employed to simulate and test the effects of different network structures on outcomes.

6.4 The social processes that affect conservation outcomes

As the field of biodiversity conservation moves beyond the management of single protected areas into the management of surrounding landscapes (Lindenmayer and Burgman, 2005) a wide range of political, economic, social and institutional factors add a new layer of complexity to an already complex science. When problems and solutions extend across multiple jurisdictions and heterogeneous land uses, and involve stakeholders and decision makers with diverse interests and objectives, planning for conservation becomes a *social process*. Learning, knowledge generation, collaboration and competition, and negotiation are some of the social processes that take place and affect decision making. These processes are thus critical determinants of the effectiveness of conservation actions. This has implications for conservation scientists and practitioners beyond the obvious but vague instruction to 'embrace' the social sciences (Mascia et al., 2003; Reyers et al., 2010), or incorporation of social factors into limited aspects of conservation planning and prioritisation (e.g. Adams et al., 2014; Guerrero et al., 2010; Ng et al., 2014; Whitehead et al., 2014). I believe that in order to improve the science and practice of conservation it is critical to understand the role that the various social processes play in the successful planning and implementation of actions, and how these facilitate or hinder conservation outcomes. Chapters 3 and 4 demonstrate how collaboration processes can enable the coordination of actions across different management scales (Chapter 3), and can ensure that management actions are aligned with the ecological system (Chapter 4). Thus this thesis contributes to understanding of how social processes of collaboration affect conservation outcomes.

Future directions

In addition to furthering the research on collaboration processes (see section 6.2) there is a need to understand how other social processes affect decision making in conservation settings. There is a large volume of published research in the psychology, education, public policy, business and management fields from which to base studies on social influence and social learning processes in conservation research (e.g. Cialdini and Goldstein, 2004; Senge and Suzuki, 1994; Zimbardo and Leippe, 1991). Learning as a social process has been extensively researched in the psychology, education and management literature (Bransford et al., 1999; Senge and Suzuki, 1994), and while it is increasingly considered as a requisite for the effective management of environmental problems (Beratan, 2007; Mostert et al., 2007; Muro and Jeffrey, 2008), it is still little understood in the natural resource management literature (Newig et al., 2010; Reed et al., 2010). Developments in the field of diffusion research has advanced understanding of how ideas spread through social interaction and learning, leading to new norms of appropriate behaviour and behaviour change within and between groups (Rogers, 2010; Valente, 2005; Valente and Pumpuang, 2007). The theories and insights offered by diffusion research, used in conjunction with network methods, can be used to progress understanding of the role of social influence in conservation (Deroïan, 2002; Keys et al., 2009).

Social processes do not occur in isolation. Difficulty in disentangling the effects of one social process from another, make it difficult to assess the prevalence of a particular social process in a conservation planning situation. Social influence and social learning are facilitated by social interactions, and thus can be examined from a social network structural perspective. Network research approaches such as the one applied in this thesis could be utilised to analyse the effects of social processes relative to other social processes taking place. The application Exponential Random Graph modelling methods enable analysis of multiple social processes and their relative effects on network structure formation. Thus it can be determined, for particular conservation settings, the dominant processes (e.g. social influence, learning, collaboration) that affect decision making.

6.5 Social-ecological research in conservation planning and beyond

Understanding the social-ecological complexity that characterises most global environmental problems is one of our greatest challenges if we are to effectively respond to biodiversity decline, climate change and the depletion of key natural resources. The interplay between different social and ecological factors makes it difficult for researchers to disentangle the impact they have on conservation outcomes, and thus, to devise effective solutions. For example, this interplay is evident when the impact that land use change has on an ecosystem varies depending on the scale of change, or when the presence of an institutional element might interfere with the efforts of another. Therefore, there is a need for research approaches to help disentangle this social-ecological complexity (Young et al., 2008). These understandings can then be fed into conservation strategies so as to increase the effectiveness of conservation efforts. Theoretical approaches that provide a foundation for empirical analyses of social-ecological systems have recently started to emerge in the natural resource management literature (e.g. Bodin and Tengo, 2012; Ostrom, 2009), and new research approaches are being developed (Bodin et al., in review). Chapters 4 and 5 demonstrate how social-ecological analyses increase the power of information drawn from ecological or social data, by explicitly accounting for the interconnections between the social and ecological systems in assessments of the effectiveness of governance arrangements (i.e. networks of governance actors; Chapter 4) and to inform the development of implementation strategies (Chapter 5). If social-ecological interconnections are not accounted for then conservation assessments might be incomplete, planning decisions might be unable to take advantage of opportunities for action, have low feasibility, or be inefficient.

There are a diverse set of frameworks available for conceptualising social-ecological systems (e.g. Collins et al., 2010; Milner-Gulland et al., 2010; Ostrom, 2009) but not many explicitly account for social-ecological interactions and their dynamics, or the social and ecological systems are not captured with equivalent depth (Binder et al., 2013). Applications of the social-ecological systems framework (SES) developed by Elinor Ostrom and colleagues (McGinnis and Ostrom, 2014; Ostrom, 2009), and applied in Chapter 5, have been primarily directed to the identification of key characteristics of social-ecological systems that are relevant to explaining or predicting outcomes. As demonstrated in Chapter 5, this framework can be used to guide the selection of social-ecological factors to be included in conservation planning studies for the identification of conservation opportunities.

Future directions

There is increasing acceptance of the need to incorporate social factors into prioritisation studies and in developing modelling tools that include both social and ecological components (Bunnefeld et al., 2011; Knight et al., 2010; Milner-Gulland, 2012; Stephanson and Mascia, 2014). But without a sufficiently comprehensive framework to help guide these efforts, researchers will be 'shooting in the dark' as the complexity of interactions between multiple system components, the numerous stakeholders and diverse institutions, make it difficult to identify the most important factors determining the effectiveness of conservation actions. While the usefulness of the SES diagnostic approach and multi-tiered collection of concepts and variables is often highlighted, issues have been raised in relation to the lack of formalised relationships between variables and concepts and representation of dynamics in the framework (Hinkel et al., 2014). On the other hand, the Management Strategy Evaluation framework (MSE) incorporates system dynamics and permits the evaluation of different management alternatives under uncertainty, but is limited to only a few ecological and social components, and assumes that management options can be implemented without accounting for the behaviour of individuals (e.g. resource users, land managers; Bunnefeld et al., 2011; Nuno et al., 2014). Finding an adequate social-ecological framework for conservation planning might entail answering questions such as: what are the current limitations of the conservation planning approach? What are the benefits and challenges of applying the most prominent social-ecological frameworks to conservation? Are there particular social-ecological frameworks that are more beneficial than others in the context of different conservation problems and settings?

Social-ecological research is also likely to benefit from continued and improved research in interdisciplinary environments. It is now widely acknowledged that interdisciplinary research that considers both social (e.g. governance and human aspects) and ecological aspects is of critical importance in addressing environmental problems (Daily and Ehrlich, 1999; Reyers et al., 2010; Rockstrom et al., 2009). While interdisciplinary studies are on the rise, and interest in interdisciplinary environmental education is rapidly growing (Vincent, 2010), research is still in infancy (Fitzgerald and Stronza, 2009), and further challenges remain. For example, addressing the disconnect that exists between scientific disciplines (Fitzgerald and Stronza, 2009), greater support and funding for the development of rigorous interdisciplinary education and research that is adequately integrated (Focht and Abramson, 2009; Nissani, 1997), the development of interdisciplinary positions and programs within conservation organisations (Mascia et al., 2003), and greater and improved cross-disciplinary collaboration (Campbell, 2005; Marzano et al., 2006). Addressing these challenges would advance social-ecological research and improve our understanding of how the social-ecological complexity can be accounted for when addressing environmental problems

6.6 Final Conclusion

Understanding how social and ecological systems interact is now recognised as essential for devising effective responses to conservation and environmental problems (Ban et al., 2013; Berkes et al., 1998; Collins et al., 2010; Folke et al., 2005; Liu et al., 2007; Milner-Gulland, 2012; Ostrom, 2009). In this thesis I develop and demonstrate approaches for accounting for the social-ecological context in conservation planning. I focus on aspects of governance, multi-stakeholder collaboration processes, and the analysis of social-ecological interactions to inform implementation strategies and to assess the benefits of collaborative approaches to governance.

This thesis advances research on a governance challenge associated with most environmental problems: the *problem of fit*. By advancing recently proposed frameworks and methodological approaches for analysing social-ecological systems, this thesis provides the first empirical test of the potential for multi-stakeholder collaborative approaches to governance to address the problem of fit. By theoretically linking the problem of fit to the capacity of a governance arrangement to address fit challenges this body of research also constitutes an important step towards understanding how the performance of governance arrangements can affect biodiversity conservation outcomes.

This research gives empirical support to the benefit of collaborative approaches to the governance of conservation and natural resource management problems. The approaches demonstrated can be replicated and applied in different contexts. Future research should also include investigations of collaboration processes in conjunction with other social processes that also affect environmental decision-making. This research direction is needed if we are to formulate and support governance arrangements and management decisions that are fit for addressing conservation and other environmental challenges.

This thesis contributes to research on governance networks, and this contribution extends beyond the field of conservation science. Approaches to investigate how structural characteristics of networks relate to governance performance have been highlighted as an important research direction in disciplines such as policy science, natural resource management and sustainable development (Bodin and Crona, 2009; Henry and Vollan, 2014). This thesis demonstrates such an approach by relating key components of network structures (i.e. network configurations or building blocks) to key governance challenges (i.e. the coordination of actions across management scales and the alignment of management actions to the biophysical system). The ability to assess the performance of governance arrangements, in relation to key governance challenges, is an important ingredient in devising evidence-based policies for new collaborative arrangements.

Conservation planning is in need of a social-ecological framework that can guide the selection of actions and identification of implementation strategies. This thesis has shown how the utilisation of social-ecological framework in conservation planning can lead to the development of implementation strategies by accounting for the opportunities and constraints offered by the connections between social and ecological elements. The framework applied in this thesis can be further explored and tested against other frameworks. This research contributes to the ongoing development of the systematic conservation planning framework to account for the social-ecological interdependencies influencing conservation outcomes.

Effective responses to environmental problems require that the interdependencies that exist between the social and ecological systems can be taken into account when devising solutions, and that governance approaches are able to respond to the challenges posed by those interdependencies. This thesis addresses both these requirements. The analytical and methodological approaches demonstrated in this thesis can be applied to further advance social-ecological research, be used to improve implementation strategies and inform new (and improve ongoing) collaborative arrangements.

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Appendices

Appendix A – Supplementary Material Chapters 3, 4 and 5

A1. Social assessment methodology and data

Exploratory stage

The objective of the exploratory stage was to understand the aims, history and context of the Gondwana Link large-scale conservation initiative, and to identify and generate data about the relevant stakeholders and gain a general understanding about their objectives, agendas, challenges, influence and relations. A further objective of this stage was to determine the feasibility of the social network study.

Data were gathered through semi-structured interviews, internal documents, strategic plans and public material (i.e. websites, reports). Semi-structured interviews can provide valuable insight into the perceptions of selected study participants. It was ensured that stakeholders with diverse views were included in the interview process. The questions that guided the semi-structured interview process are shown in Table A1. The interviews were used to understand the complex social environment surrounding conservation in the Fitz-Stirling area (and adjacent areas). The particular aims were to 1) identify the stakeholder playing a key role in the conservation of the Fitz-Stirling, 2) identify the main activities carried out related to conservation of the Fitz-Stirling, 3) gain a sense of the diversity of values driving conservation in the Fitz-Stirling, 4) determine the feasibility of gathering social network data and determine an effective method for doing so, and 5) Identify an initial list of stakeholders to be approached for completing the next research stage: the quantitative stage (social network data gathering).

Stakeholder analysis methods were applied (Reed et al 2009, Varvasovszky and Brugha 200, Brugha & Varvasovszky 2000). Specifically, the ‘reputational approach’ or ‘key informant approach’ was used to generate an initial, preliminary, list of stakeholders. Informants were selected on the basis of their position and association with the Gondwana Link initiative. It was ensured that all known stakeholder categories were represented in the informant group. To complete the list of stakeholders (to be used in the quantitative stage) interview participants were presented with the preliminary list and asked to add stakeholders to which they had relationships that were relevant to the study (Marsden, 2005). When a new stakeholder category was identified through this process they were approached to complete an interview. This was done until no new stakeholder categories could be identified. A total of 9 key stakeholder categories were identified (Figure A1). The completed stakeholder list was used in the next research stage: the quantitative stage. This list comprised 48 organisations.

Quantitative stage

Data collected in the exploratory stage informed the research design of the quantitative research stage. This research stage had several objectives: to confirm perceptions around challenges to conservation action implementation identified in the Qualitative stage, to generate the data for the social-ecological analysis (including social network data), and identify perceptions around collaboration. The questionnaire was conducted online using the software *Checkbox*. To increase completion rate, additional contact (through email and phone) preceded and followed the online questionnaire. Survey data were collected between October 2011 and July 2012.

A stakeholder was deemed part of the network on the basis of their activities in the Fitz-Stirling region. A stakeholder would be considered part of the network if they stated that they had been involved, directly or indirectly, with any of the conservation activities listed in Table A2, within the previous two years (see Table A3 for the specific question used).

Survey respondents were presented with the completed list of 48 organisations and asked to recall if within the previous 2 years they had collaborated with any of the organisations listed. They were also asked if they had collaborated with any landowners within the same time period (see questions used in Table A4).

Survey respondents were asked to indicate on a map the vegetation patches in which they applied their conservation and management activities (See Table A5) for the specific question used). This data was used to determine the proportion of stakeholders working in each vegetation cluster.

A total of 38 completed online questionnaires were obtained, representing the 9 key stakeholder categories identified during the qualitative stage (Figure A1). A total of 19 of the 48 identified organisations responded to the survey (40% response rate). The other 19 respondents were landowners. While 29 identified organisations did not respond to the survey some of their collaborative relationships were identified through survey respondents. The analytical method used treats network connections as a statistical sample, and hence robust conclusions can be draw from partial networks. Overall, the network comprised a total of 205 collaborative relationships based on all activities considered (Table A2).

Descriptive data

Figures A2 and A3 show the perceptions held by Fitz-Stirling stakeholders, on factors influencing implementation of on-the-ground conservation activities (Figure A2), and on the effects of collaborative relationships on the performance of on-ground activities (Figure A3). Figure A4 show the perceptions held by Fitz-Stirling stakeholders of the value of collaboration with government agencies. These results were used in the identification of implementation capacity metrics (in Chapter 5).

Figures A5 to A9 show information describing the people who responded to the survey. Half of the people who completed the survey where farmers; most of them dedicated to a mix of livestock and crop production (74%), while others only did crop production (11%) or only livestock (5%), and others stock agistment (5%) (Figure A5). All other stakeholder categories where represented in the sample. The majority of respondents (67%) had been working in natural resource management or conservation related activities in the Fitz-Stirling for more than 5 years (Figure A6), and over half of those respondents who farmed had done so for over 20 years (Figure A7). At the time of data collection, a third of the respondents where between 46 to 55 years of age and only 10% of them where over 60 years of age (Figure A8). Around half of the respondents where affiliated to one or two community groups, and 16% had no community group affiliations (Figure A9). Apart from bush fire brigades, the Fitzgerald Biosphere Group was the most popular form of membership, followed by Friends of the Fitzgerald River National Park and the Mallefowl Preservation Group.

Figures A10 to A12 show information on the activities stakeholders are involved with. According to the survey results, conservation of the Fitz-Stirling is pursued through diverse activities with fox control and revegetation amongst the most popular (Figure A10). In addition, most stakeholders are involved with more than one type of activity, and the average across all respondents was 4 activities (Figure A11). Some of the stated activities under the "Other Activity" category included salt control, river health monitoring and research on particular threatened species. There are diverse ways in which stakeholders get involved conservation activities (Figure A12). The most common type of involvement across all activities is in planning, management or coordination roles (Figure A12). In addition, over half of the stakeholders are involved with on the ground implementation of activities and a third with education or public awareness (Figure A12).

***Table A1**. Semi-structured interview guiding questions*

1. *What kind of activities do you undertake in relation to conservation in the Fitz-Stirling e.g. habitat protection, revegetation.* 2. *What are the geographic area(s) where you undertake those activities?* 3. *What are you trying to achieve through your involvement with this activities? Why are you doing it? What are the challenges?* 4. *Who are you collaborating with in undertaking such activities, and the nature of that collaboration (e.g. formal agreements, funding, informal etc.)* 5. *Roughly how many people/organisations do you currently collaborate with?* 6. *How many more if you think about the last 2 years?* 7. *Can you give a rough indication of how many rangers/farmers/NGO personnel/government personnel are working on habitat protection/revegetation/other conservation activities in the Fitz-Stirling region?* 8. *What do you understand by revegetation?* 9. *What do you understand by habitat protection? What kind of activities do you think fall into habitat protection?* 10. *Who do you think the key players are in relation to conservation activities such as revegetation and habitat protection (those who contribute the most to revegetation/habitat protection outcomes)* 11. *What are your perceptions of the collaboration network (pattern of relationships) that currently exists around key conservation activities like revegetation and/or habitat protection. Is it well connected or highly fragmented? Are there clearly defined groups? If so, which? Are there opposing agendas? What are them?* 12. *What kinds of collaborations occur between individuals and organisations with regards to activities like revegetation and/or habitat protection (or setting aside land for conservation, fencing off vegetation). For example*

a. Individuals/organisations might coordinate the activities they are doing or planning to do with other individuals/organisations, in terms of location, timing, resources etc.

b. Others might enter into formal and informal agreements e.g. partnerships where one party contributes money and another one the knowledge or on-ground resources, or a landowner allowing an individual/organisation to come into their property to do some kind of research or conservation activity.

c. How else can two individuals/organisations work together with regards to revegetation activities?

d. How else can two individuals/organisations work together with regards to habitat protection activities?

e. How else can two individuals/organisations work together with regards to other conservation activities?

13. Who engages with planning activities for revegetation / habitat protection in your organisation? Who does coordination/implementation?

14. Is it safe to assume that most landholders that do habitat protection/revegetation activities on their land belong to a member based organisation such as "friends of" groups and catchment groups?

15. Do you belong to any natural resource/conservation related groups/committees? How many? Which ones?

16. Would you be willing to respond a survey either online or face-to-face in relation to this subject? It will take around 15 min to complete. What would be your preference (online, phone, face-to-face)?

Table A2. Conservation activities in the Fitz-Stirling

Types of conservation activity
Revegetation/restoration Livestock management Weed management

Feral animal management
Fire management
Land use planning
Purchasing or setting aside land for conservation

Table A3. Survey question: network boundary question

"Over the <u>past two years</u>, have you been involved in any of the following activities in relation to the Fitz-Stirling bushland? (Please tick all that apply)"* ** Your involvement may include, but is not restricted to: planning, coordination, fieldwork/on-the-ground implementation, fundraising, provision of funding, research, education, monitoring, survey work, or lobbying for any of these activities.*
Revegetation or habitat restoration
Stock exclusion from bushland
Fencing off bushland and/or revegetation areas
Weed management (in bushland)
Erosion control (in bushland)
Fire management
Fox control
Other feral animal control
Purchasing or setting aside land for conservation
Land use planning and/or management
Other activity that relates to the conservation of Fitz-Stirling bushland

Table A4. Survey question: collaboration network

"*This section relates to the people you have collaborated with in relation to the conservation activities you have been involved with (in the Fitz-Stirling region)*". *"Over the <u>past two years</u>, have you collaborated* with any of the following organisations when performing the activities you were involved with in the Fitz-Stirling region? Select all that apply."*

**Collaboration might include: Sharing information or advice, coordinating activities, working together, receiving or providing assistance, goods, services, technological or financial resources, etc.*
"Over the past two years, have you collaborated with any landowners when performing any of the activities you have been involved with in the Fitz-Stirling region?"* **Collaboration might include: Sharing information or advice, coordinating activities, working together, providing assistance, goods, services, technological or financial resources, etc.*

Table A5. Survey question: location of activities

"This map below shows "areas of interest" of this research study. *If you undertake any activities in any of these areas please indicate so below by clicking on the relevant options on the list shown.* *Tick all that apply"* <PICTURE OF MAP> *Please tick on all areas that apply to your past (in the last two years) or current activities.* □ *Area a* □ *Area b* □ *Area c*

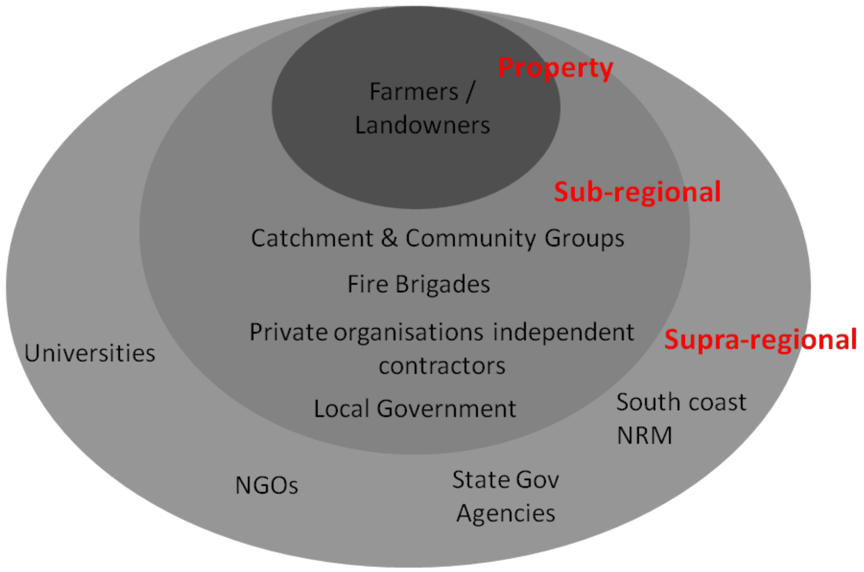


Figure A1. Fitz-Stirling stakeholder categories and their scale of interest.

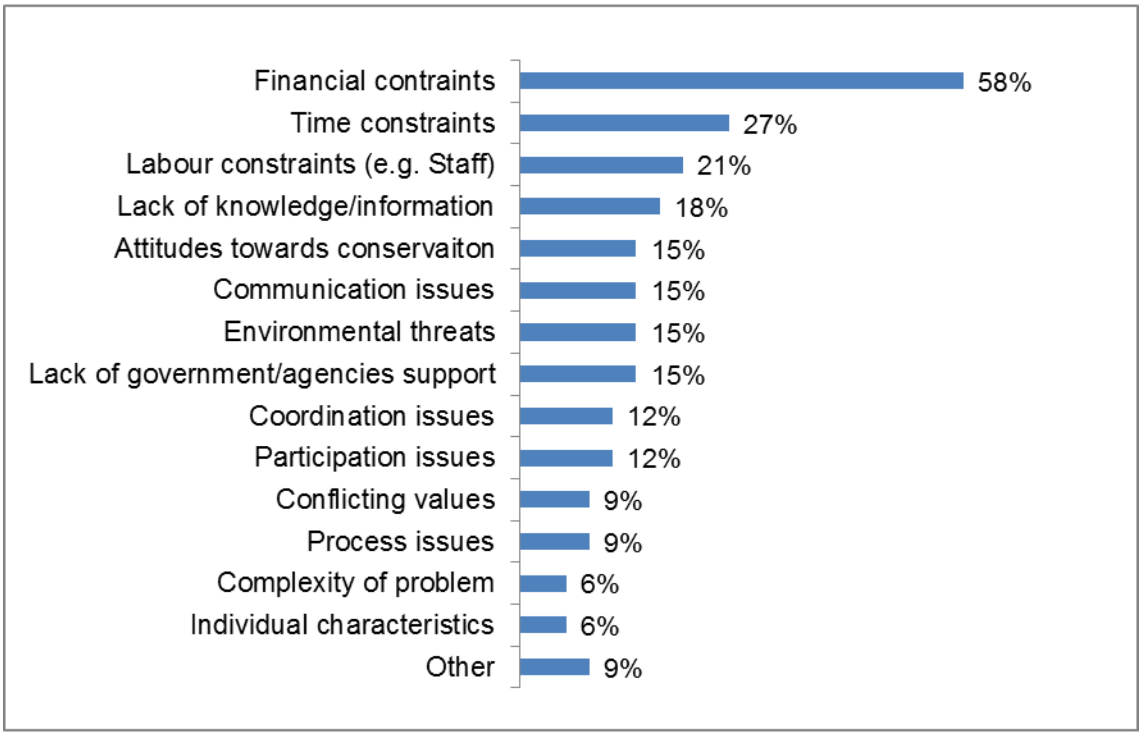


Figure A2. Perceived barriers to conservation action implementation including communication and coordination issues. Proportion of respondents who mentioned each barrier - unprompted (n=33). Where time constraints barriers were mentioned, these were strongly associated to workforce constraints (e.g. "*Time - we do the work ourselves so have to fit it in with normal farming work*").

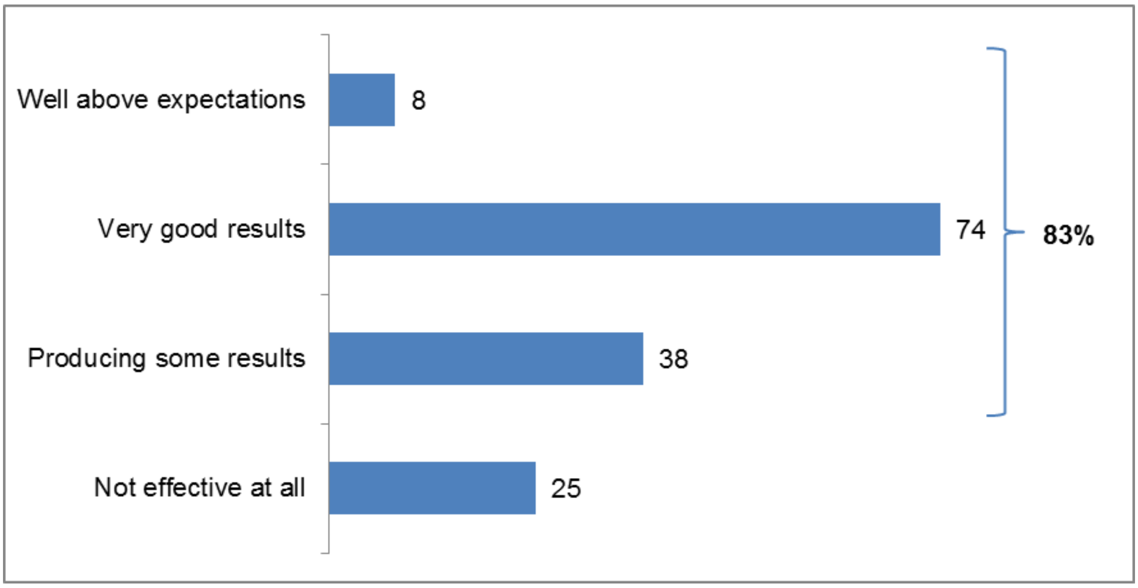


Figure A3. Perceptions on collaborations. *Number of collaborations perceived as delivering some, very good, or results above expectations to the particular activity performed at a particular location* n = 145.

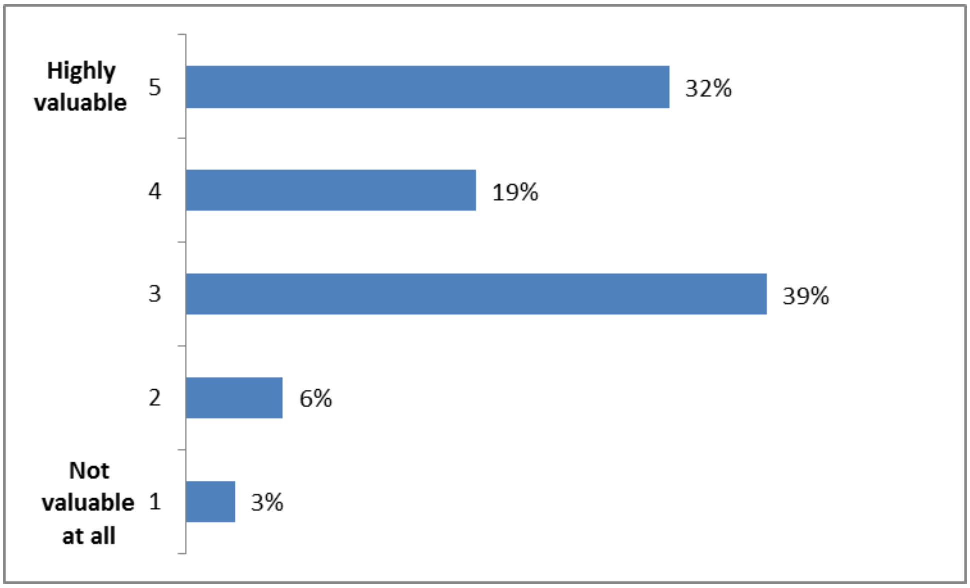


Figure A4. Perceptions on the value of collaboration with government agencies n = 31.

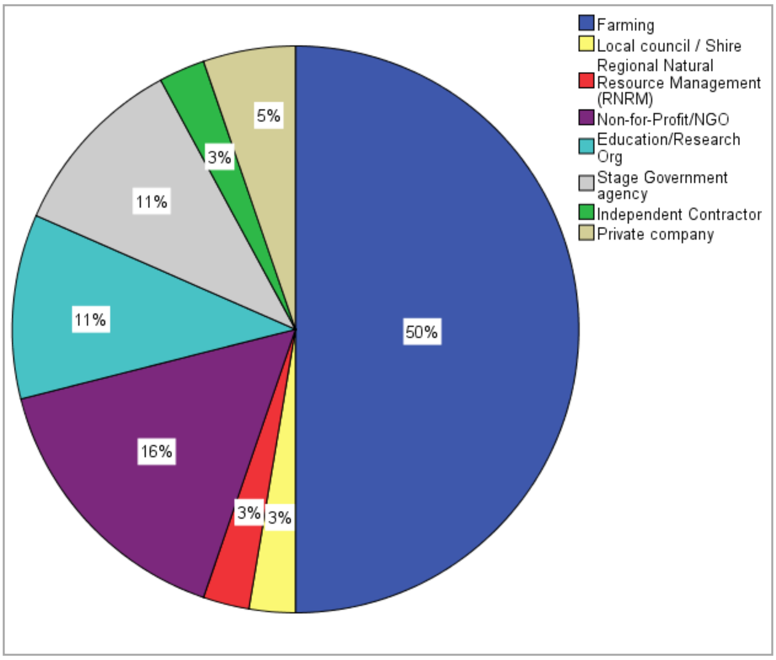


Figure A5. Respondents' organisational affiliation

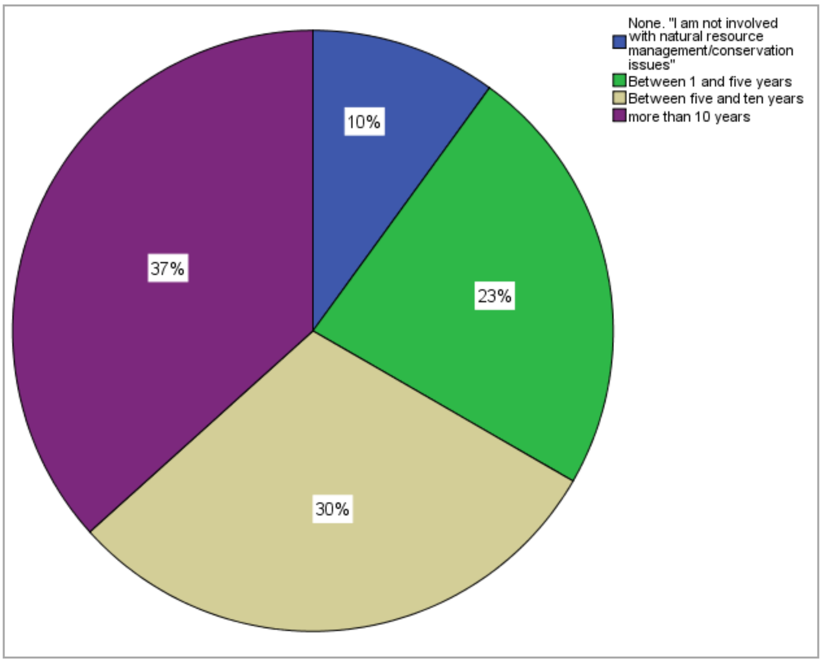


Figure A6. Respondents years of involvement in natural resource management or conservation related activities in the Fitz-Stirling

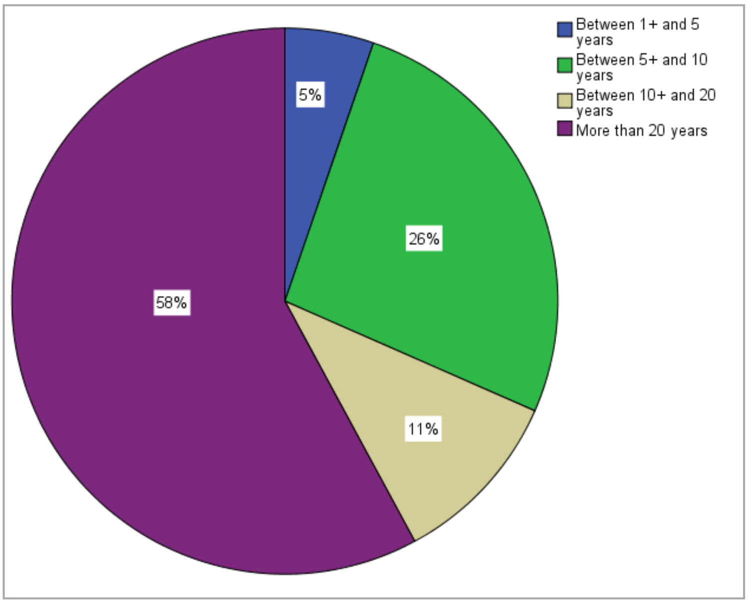


Figure A7. Respondents' years of farming (Farmer category only)

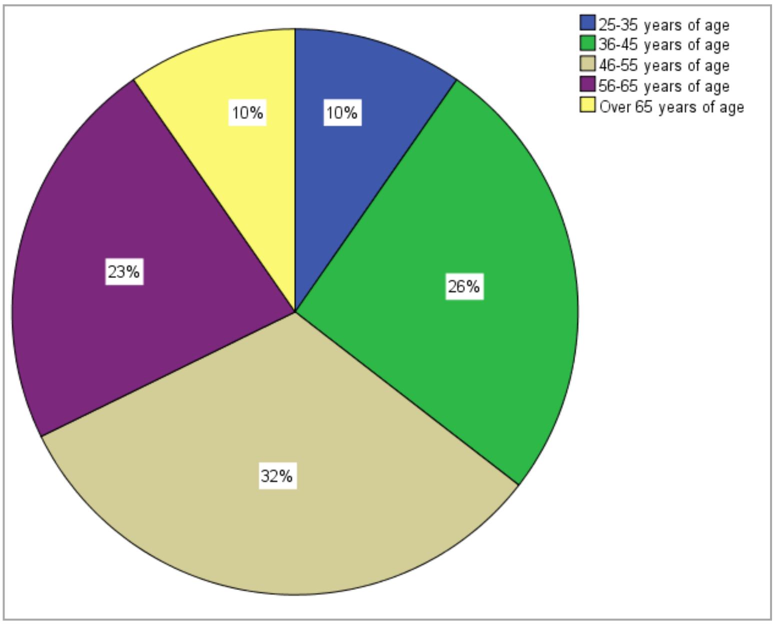


Figure A8. Respondents' age

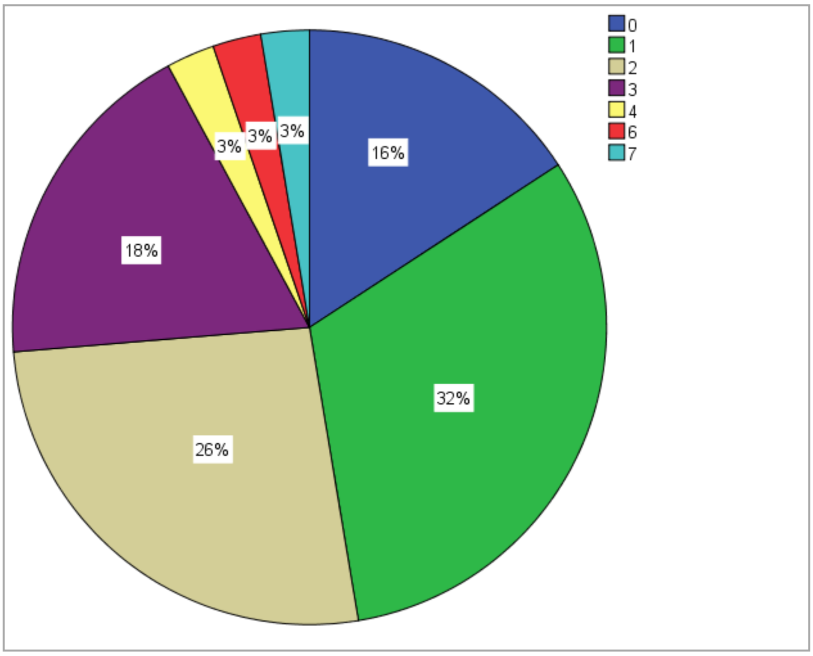


Figure A9. Respondents' number of community group memberships

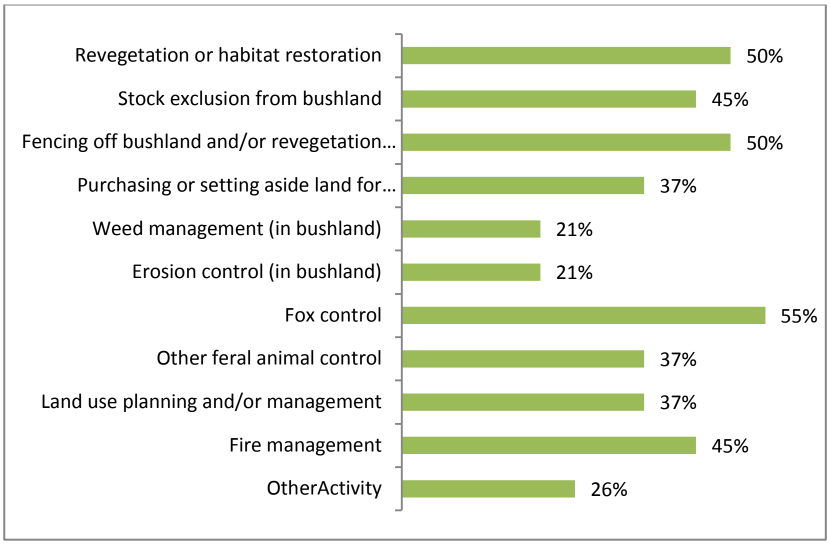


Figure A10. Percentage of stakeholders involved with each of the activities related to the conservation of the Fitz-Stirling Bushland

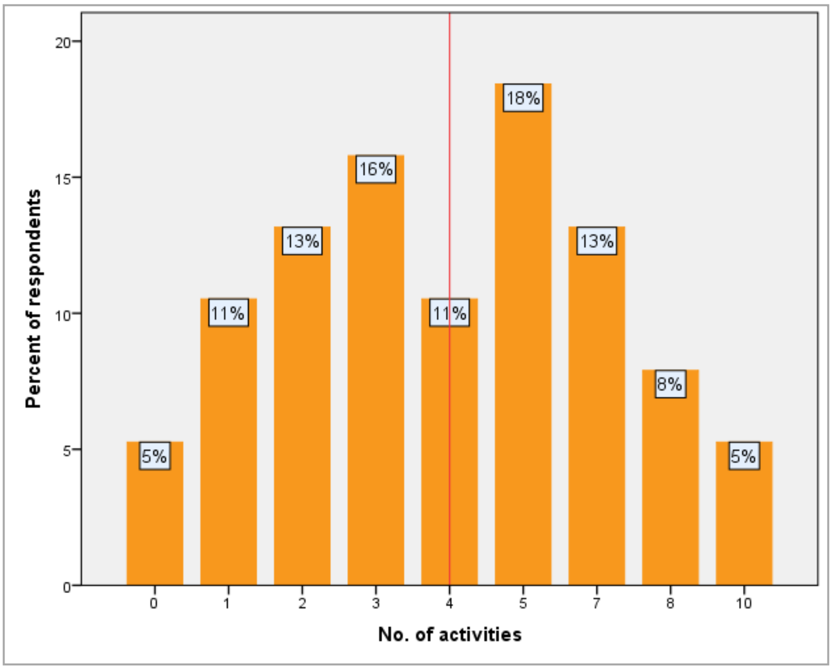


Figure A11. Number of activities stakeholders are involved with. *The red vertical line represents the mean result for all respondents.*

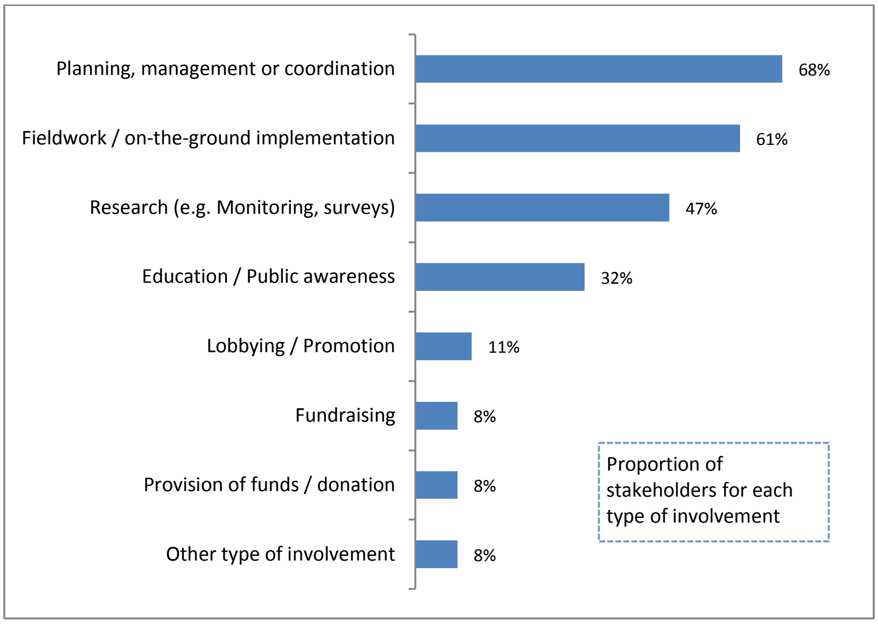


Figure A12. Ways in which stakeholders get involved in conservation related activities

A2. Fitting the Exponential Random Graph Model

To fit the exponential random graph model (ERGMs) the computer package pNet (Wang et al., 2009) was used. A systematic approach was followed to incrementally simplify from a model containing all possible parameters.

The first step was to build a model containing the “collaboration type” configurations, which represented *coordination* and *cooperation* (see Figure 3.3 in section 3.3.4). This simple model is shown in Table A5, and shows significant and positive parameter values for the *coordination* configurations for the all-activities, revegetation and feral animal control networks.

The second step was to fit a model that contained the simple model plus the configurations representing within-scale and cross-scale interactions (see Figure 3.3 in section 3.3.4 and Table A6).

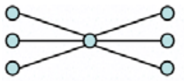
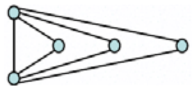
The third step was to re-fit the model adding the configuration representing a scale-bridging role (see Figure 3.3 in section 3.3.4, Table A7).

To help fit the models all of the models include an additional baseline parameter (configuration *D3* in pNet nomenclature). The models for the invasive animal control network also include the *2-star* configuration. These are baseline parameters that helps explain the overall structure of the networks. To avoid confusion I only show the parameters relevant to my analysis.

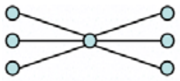
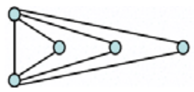
At each step I used goodness-of-fit procedure to generate 1000 random graphs from the model parameters. I compared the configuration counts between the random graphs and the observed data. T-tests show no statistical difference between the observed configuration counts and those from the random graphs, with values between 1 and -1 (Table A5, A6 and A7). This means the models describe the observed distribution of configurations well, even for those configurations not included in the model.

Table A6. Step 1 models for the all-activities (A), revegetation (B) and pest animal control networks (C) estimated as Exponential Random Graph Models using pNet, with fixed Densities of 0.0574, 0.0218 and 0.0176.

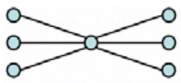
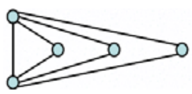
All-activities network

Form of collaboration	Configuration	Parameter estimates (t-stat)	Observed counts (t-stat[†])
Coordination		0.4744 (1.89)^	587 (-0.16)
Cooperation		-0.2092 (-2.19)*	256 (-0.18)

Revegetation network

Form of collaboration	Configuration	Parameter estimates (t-stat)	Observed counts (t-stat[†])
Coordination		1.0221 (3.89)**	193 (0.09)
Cooperation		-0.0342 (-0.17)	78 (0.09)

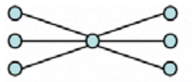
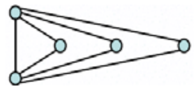
Feral animal control network

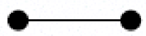
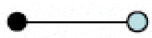
Form of collaboration	Configuration	Parameter estimates (t-stat)	Observed counts (t-stat[†])
Coordination		0.7971 (2.85)**	182 (-0.21)
Cooperation		-0.5558 (-2.18)*	46 (-0.18)

^/*/** shows 90/95/99 % significance for the parameters. [†]T tests compare observed configuration counts against simulation means.

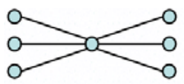
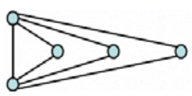
Table A7. Step 2 models for all-activities (A), revegetation (B) and pest animal control networks (C) estimated as Exponential Random Graph Models using pNet, with fixed Densities of 0.0574, 0.0218 and 0.0176 respectively.

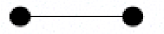
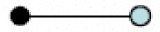
All-activities network

Form of collaboration	**Configuration**	**Parameter estimates (t-stat)**	**Observed counts (t-stat†)**
Coordination		0.002 (0.01)	587 (0.03)
Cooperation		0.0117 (0.16)	256 (-0.01)

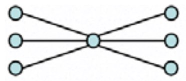
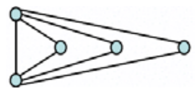
Mode of interaction	**Configuration**	**Parameter estimates (t-stat)**	**Observed counts (t-stat†)**
Within-scale (Property)		-0.4034 (-0.75)	5 (-0.01)
Within-scale (Sub-regional)		-1.8191 (-4.57)**	13 (-0.02)
Within-scale (Supra-regional)		0.9119 (3.37)**	49 (-0.02)
Cross-scale (Property)		0.0149 (0.16)	73 (0.02)
Cross-scale (Sub-regional)		0.5588 (4.29)**	142 (0.002)
Cross-scale (Supra-regional)		-0.288 (-2.61)**	195 (-0.01)

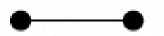
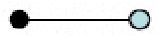
Revegetation network

Form of collaboration	**Configuration**	**Parameter estimates (t-stat)**	**Observed counts (t-stat†)**
Coordination		1.0296 (3.94)**	193 (-0.01)
Cooperation		-0.0327 (-0.18)	78 (0.02)

Mode of interaction	Configuration	Parameter estimates (t-stat)		Observed counts (t-stat†)	
Within-scale (Property) ≠		N/A≠	N/A≠	0	(-1.00)
Within-scale (Sub-regional)		-0.9464	(-1.47)	7	(0.06)
Within-scale (Supra-regional)		0.6114	(1.31)	17	(0.07)
Cross-scale (Property)		-0.0148	(-0.16)	31	(-0.14)
Cross-scale (Sub-regional)		0.2499	(1.27)	54	(0.02)
Cross-scale (Supra-regional)		-0.1291	(-0.72)	71	(0.09)

Animal pest control network

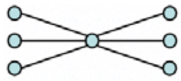
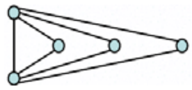
Form of collaboration	Configuration	Parameter estimates (t-stat)		Observed counts (t-stat†)	
Coordination		0.7611	(3.37)**	182	(-0.06)
Cooperation		-0.4233	(-2)*	46	(-0.03)

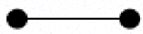
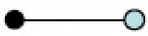
Mode of interaction	Configuration	Parameter estimates (t-stat)		Observed counts (t-stat†)	
Within-scale (Property) ≠		N/A≠	N/A≠	0	(-0.55)
Within-scale (Sub-regional)		-3.2652	(-4.2)**	5	(0.03)
Within-scale (Supra-regional)		2.4439	(3.6)**	11	(-0.16)
Cross-scale (Property)		3.3899	(19.84)**	26	(0.21)
Cross-scale (Sub-regional)		5.2282	(14.06)**	53	(0.12)
Cross-scale (Supra-regional)		2.945	(13.27)**	47	(-0.19)

^/*/** shows 90/95/99 % significance for the parameters. † T tests compare observed configuration counts against simulation means. ≠ One parameter could not converge as there are no instances of this configuration in the observed network.

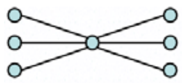
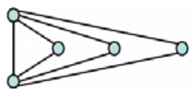
Table A8. Step 3 models for all-activities (A), revegetation (B) and pest animal control networks (C) estimated as Exponential Random Graph Models using pNet, with fixed Densities of 0.0574, 0.0218 and 0.0176 respectively.

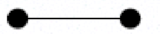
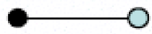
All-activities network

Form of collaboration	**Configuration**	**Parameter estimates (t-stat)**	**Observed counts (t-stat†)**
Coordination		0.3965 (1.84)^	587 (-0.03)
Cooperation		-0.1876 (-1.91)^	256 (-0.02)

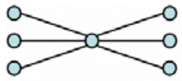
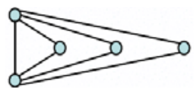
Mode of interaction	**Configuration**	**Parameter estimates (t-stat)**	**Observed counts (t-stat†)**
Within-scale (Property)		-0.275 (-0.52)	5 (0.05)
Within-scale (Sub-regional)		-1.8257 (-4.2)**	13 (0.06)
Within-scale (Supra-regional)		0.9977 (3.59)**	49 (-0.04)
Cross-scale (Property)		0.3267 (3.1)**	73 (0.04)
Cross-scale (Sub-regional)		0.7961 (5.22)**	142 (0.04)
Cross-scale (Supra-regional)		-0.0104 (-0.08)	195 (-0.04)
Scale-bridging (Property)		0.0411 (2.89)**	283 (0.07)
Scale-bridging (Sub-regional)		0.0538 (4.65)**	514 (0.05)
Scale-bridging (Supra-regional)		0.0453 (5.03)**	1845 (-0.04)

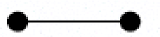
Revegetation network

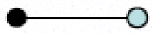
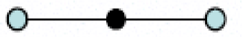
Form of collaboration	**Configuration**	**Parameter estimates (t-stat)**	**Observed counts (t-stat†)**
Coordination		1.0822 (4.01)**	193 (-0.02)
Cooperation		-0.033 (-0.15)	78 (0.04)

Mode of interaction	Configuration	Parameter estimates (t-stat)		Observed counts (t-stat†)	
Within-scale (Property) ≠		N/A≠	N/A≠	0	(-1.07)
Within-scale (Sub-regional)		-0.9841	(-1.59)	7	(-0.04)
Within-scale (Supra-regional)		0.6255	(1.17)	17	(0.04)
Cross-scale (Property)		-4.0711	(-25.23)**	31	(-0.06)
Cross-scale (Sub-regional)		-3.9261	(-17.9)**	54	(-0.02)
Cross-scale (Supra-regional)		-4.1462	(-17.56)**	71	(0.06)
Scale-bridging (Property)		0.088	(1.92)^	80	(-0.05)
Scale-bridging (Sub-regional)		0.1175	(6.89)**	175	(0.1)
Scale-bridging (Supra-regional)		0.084	(2.56)*	271	(0.05)

Animal pest control network

Form of collaboration	Configuration	Parameter estimates (t-stat)		Observed counts (t-stat†)	
Coordination		1.0958	(3.74)**	182	(0.19)
Cooperation		-0.3337	(-1.18)	46	(0.26)

Mode of interaction	Configuration	Parameter estimates (t-stat)		Observed counts (t-stat†)	
Within-scale (Property) ≠		N/A≠	N/A≠	0	(-0.75)
Within-scale (Sub-regional)		-3.4069	(-3.97)**	5	(0.19)
Within-scale (Supra-regional)		2.1195	(2.55)*	11	(-0.07)

Cross-scale (Property)		3.8099	(7.96)**	26	(0.13)
Cross-scale (Sub-regional)		2.548	(5.75)**	53	(0.26)
Cross-scale (Supra-regional)		0.8853	(2.54)*	47	(-0.24)
Scale-bridging (Property)		-1.8949	(-3.83)**	9	(0.13)
Scale-bridging (Sub-regional)		-0.0392	(-0.24)	459	(0.23)
Scale-bridging (Supra-regional)		-0.0454	(-0.27)	180	(-0.39)

^/*/** shows 90/95/99 % significance for the parameters. † T tests compare observed configuration counts against simulation means. ≠ One parameter could not converge as there are no instances of this configuration in the observed network.

Appendix B – Glossary of Terms

Conservation planning: a mechanism used to make decisions about how best to respond to threats affecting biodiversity decline.

Conservation stakeholders: all individuals and organisations that have an interest or concern in conservation endeavours.

Environmental governance systems: The set of formal and informal rules, rule-making systems, mechanisms, and organisations (and actor-networks), through which actors influence environmental actions and outcomes.

Governance: the process of guiding societies towards outcomes that are socially beneficial and away from outcomes that are harmful.

Governance actors: all individuals and organisations who have a role in environmental governance.

Institutions: the norms, rules, rights and decision making processes designed to shape human behaviour. Institutions include rules, laws, constitutions and informal constraints such as norms of behaviour, conventions, and self-imposed codes of conduct.

Problem of fit: when governance systems do not fit or match the characteristics of the biophysical system.

Scale mismatch: A scale mismatch occurs when the scales for planning and implementing conservation actions do not match the scale of biophysical processes.