ECOSYSTEM SERVICES ACROSS CONTRASTING FORESTED LANDSCAPES IN QUEENSLAND'S WET TROPICS BIOREGION: CONTEMPORARY PATTERNS, PROCESSES AND LIKELY FUTURE TRENDS UNDER A CHANGING CLIMATE

Thesis submitted for the Degree of Doctor of Philosophy in the College of Marine and Environmental Sciences, James Cook University, Cairns, Australia

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DEDICATION

I dedicate this thesis to my parents Mohammed Shafiul Alam and Khorshida Begum, who worked hard across their life to provide me a better education.

I also dedicate this thesis to my two children Nusayba Jahan Aqsa and Mohammed Ashaz Alamgir, where I got my happiness after every day hard work throughout my PhD journey.

And finally, to my wife Samia Jahan, who always inspired me throughout my PhD journey.

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ABSTRACT

Ecosystem services are the benefits human communities receive from ecosystems that comprise of both goods and services. Forests provide numerous ecosystem services which are necessary for human well-being. Examples include global climate regulation, air quality regulation, cyclone protection, habitat provision, timber and energy. The supplies of most ecosystem services from forested ecosystems are declining across the globe. Furthermore, the rate of decline is higher in tropical forested areas than other forest biomes. The main reasons for this decline are climate change, forest degradation, land use and land cover change and, most importantly, the reduced focus on the sustainable management of ecosystem services. Ecosystem services from forested landscapes are supplied as a complex interaction between forest vegetation attributes and environmental factors, and interactions among various ecosystem services. Ecosystem services management in natural resource management planning is not well addressed due to the lack of scientific information on the nature and distribution of ecosystem services, poor understanding of the supply of ecosystem services processes and unknown trends of ecosystem services under future climate change scenarios.

In this doctoral thesis, I investigated ecosystem services supply across a complex forested landscape, their interactions, and likely trends of ecosystem services under future climate change scenarios. The study was based in the contrasting forested landscape of the Wet Tropics bioregion in northeast Australia. The study was focused on five objectives: (1) to assess the supply of multiple ecosystem services from rainforests, sclerophyll forests and rehabilitated plantation forests, and how they are spatially distributed across the landscape; (2) to investigate the interactions among multiple ecosystem services and how the supply of one ecosystem service is influenced by others; (3) to evaluate the capacities of different Land Use and Land Covers (LULC) to supply ecosystem services, and to identify the spatial congruence between ecosystem services and biodiversity in the landscape; (4) to compare the carbon storage among rainforests, degraded rainforests and sclerophyll forests and to determine the drivers of carbon storage in this tropical forested landscape; (5) to determine the likely effects of climate change on ecosystem services supplied from rainforests and sclerophyll forests. To achieve these objectives, I collected forest vegetation attribute data from 66 forest

plots (0.05 ha each, 50m×10m transects) over a two-year period (2013-2014), collected current spatial datasets about forest distribution, interviewed different stakeholders, and arranged a forest experts workshop (September 2015) at James Cook University, Cairns. I analysed the collected data using a range of statistical-R, SPSS 20, PC-ORD6, and spatial software-ESRI ArcGIS 10.2. I utilized *OzClim* and *Climate Futures* online software tools for Australia for the climate change modelling.

My study revealed that the supply of global climate regulation, air quality regulation, nutrient regulation, cyclone protection, habitat provision, energy provision and timber provision ecosystem services were found to be significantly higher in rainforests than sclerophyll forests, while erosion regulation ecosystem services were significantly higher in sclerophyll forests than rainforests. The results also showed that rehabilitated plantation forests may provide some ecosystem services that are comparable to rainforests. In the investigation of spatial distributions and patterns, I found that all examined ecosystem services were unevenly distributed and differed considerably both within the same forest type and among the different forest types. The hotspots (supply of significantly higher ecosystem services) for multiple ecosystem services were found in upland rainforests followed by lowland rainforests and upland sclerophyll forests in the study region.

In understanding the interactions among multiple ecosystem services, my research revealed that the global climate regulation ecosystem service had a synergistic impact on the supply of remaining examined ecosystem services while nutrient regulation ecosystem service emerged as having a trade-offs impact. Overall, among eight examined ecosystem services two synergy groups and one trade-off group were identified. I found that elevation, rainfall and temperature gradients, along with forest structure and type, were the main determinants for the quantity of ecosystem services supplied in the landscape.

In the investigation of the spatial congruence between ecosystem services and biodiversity at the landscape scale, my research revealed that a spatial congruence occurred between high-potential biodiversity and high-potential global climate regulation ecosystem service in the intact rainforest areas, while spatial divergence occurred in the sclerophyll and other disturbed and low tree abundance forested areas.

In the evaluation of above ground biomass carbon storage, I found that some degraded rainforests stored similar amounts of above ground biomass carbon like intact rainforests while sclerophyll forests stored lower carbon than intact rainforests as well as heavily degraded rainforests. I also found that large trees and tree abundance are the major drivers of above ground carbon storage in this tropical forested landscape.

In the investigation of the likely effects of climate change on the future supply of ecosystem services, I found that the supplies of most ecosystem services from rainforests in this region are likely to be reduced due to future climate change, while uncertainty exists for the supply of ecosystem services from sclerophyll forests. Additionally, the ecosystem services supplied from the upland rainforests are likely to be more negatively affected than lowland rainforests.

The outcomes of this doctoral thesis indicate that active conservation of rainforests needs to occur, with more emphasis placed on protecting the upland rainforests. Rehabilitated forests are also worth protecting for ecosystem services supply, and increasing structural diversity in the sclerophyll forests is likely to increase ecosystem services supply. My research also concludes that management intervention to maximize the supply of global climate regulation and habitat provision ecosystem services are likely to maximize the supply of other examined ecosystem services in the forested landscape. My research suggests that an integration of high-potential multiple ecosystem services supply and high–potential biodiversity conservation may be possible in the tropical forested landscape provided that the multiple ecosystem services are forest-based (e.g. global climate regulation, air quality regulation, and habitat provision). Appropriate climate change adaptation, specially focusing on the ecosystem services supplied from rainforests, needs to occur without any further delay.

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INTRODUCTION

Ecosystem services are of paramount importance for human well-being (Costanza et al., 1997; MA, 2005; Raymond et al., 2009). The supply of most recognized ecosystem services are declining across the globe (Egoh et al., 2010; Nelson et al., 2009; Shaw et al., 2011), potentially undermining the well-being of communities (Barbier, 2015; Mutoko et al., 2015). Conservation of ecosystem services is necessary to sustain their future supply. Furthermore, without optimal assessment and mapping of ecosystem services, the sustainable benefits of ecosystem services are unlikely to be achievable (Naidoo et al., 2008). Moreover, the science of the assessment of ecosystem services is widely lacking in the literature (Seppelt et al., 2011; Seppelt et al., 2012). Given these gaps in knowledge, this research aims to provide new knowledge and insights to the growing field of ecosystem service sciences.

Figure 1. The spectacular rainforests in the Wet Tropics bioregion (Photo credit: Mohammed Alamgir).

The concept of ecosystem services is relatively a new science (Fisher et al., 2009) but evolving rapidly, especially after the initiative of Millennium Ecosystem Assessment (MA) (MA, 2003, 2005). Costanza et al. (1997) illustrated the concept of ecosystem services as both goods and services that are used by human populations. Examples of goods include: production of timber, fuel, fish, crops and fodder; examples of services include: storm protection, flood control, cloud formation, and greenhouse gas regulation. At the same time, Daily (1997) defined that life sustaining goods and services are essentially ecosystem services. Boyd and Banzhaf (2007) described ecosystem services from an economic point of view, and endeavoured to state them as comprising of intermediate and final services. To do so, they set the condition of "directly consumed" to be as final ecosystem services. More recently, Fisher et al. (2009) considered all services either directly consumed or indirectly consumed as ecosystem services.

Figure. 2. The unique sclerophyll forests in the Wet Tropics bioregion (Photo credit: Mohammed Alamgir)

The MA (MA, 2005) illustrated ecosystem services as the benefits people obtain from ecosystems. This concept is easily understood and conceptually simple but covers most of the services benefitting humans; hence the MA approach is widely used in ecosystem services research (Baral et al., 2014b; Burkhard et al., 2012; Pert et al., 2015). The MA broadly classified ecosystem services into four categories: i) provisioning services- products obtained from ecosystems, e.g. timber, fuelwood; ii) regulating services- benefits obtained from regulation of ecosystem processes, e.g. climate regulation, water regulation; iii) cultural services- non-material benefits obtained from ecosystems, e.g. recreation and ecotourism, and aesthetic; and iv) supporting services- services necessary for the production of all other ecosystem services, e.g. primary production (MA, 2005).

Globally, forests cover ~42 million km^2 of land, which is ~30% of the Earth's land surface (Bonan, 2008; Kirilenko and Sedjo, 2007), supplying a variety of valuable ecosystem services such as climate regulation, water regulation, habitat provision, provision for foods, medicines, and aesthetic values (Costanza et al., 1997; Daily, 1997; MA, 2005; Nkem et al., 2010). However, ecosystem services provided by forests are becoming scarcer in many countries, especially in the tropics (Carrasco and Papworth, 2014; Koellner et al., 2008; Mutoko et al., 2015).

Globally, tropical forests are the single most important terrestrial biome with a capacity to provide multiple ecosystem services (Costanza et al., 1997), although tropical forests only cover ~6% of the global land surface (Saatchi et al., 2011). The rate of decline of ecosystem services supply from the world's tropical forests is also higher than other forest biomes (Liu et al., 2015). Therefore, understanding the dynamics and nature of ecosystem services supplied from a tropical forested landscape is paramount to ensure their future conservation and recognition of their importance to humanity.

Figure 3. The Wet Tropics bioregion a magnificent habitat for biodiversity (Photo credit: Mohammed Alamgir).

The supply of ecosystem services in a forested landscape varies due to differences in vegetation type and cover (Burkhard et al., 2012; de Groot et al., 2010), and environmental factors and forest management systems (Palomo et al., 2013). The interaction among ecosystem services may also be different depending on the ecosystem services under consideration (Harrison et al., 2014). Therefore, a solid understanding of the distribution of ecosystem services across the forested landscape, together with quantifying interactions between ecosystem services and environmental drivers is enormously important. However, very little is known about ecosystem services science in tropical forests (Alamgir et al., 2014a; Seppelt et al., 2011; Seppelt et al., 2012). The heterogeneous tropical forested landscape of the Wet Tropics bioregion in Australia may provide an opportunity to understand the process and interactions of multiple ecosystem services supply, thereby contributing to this knowledge gap.

Figure 4. The damage of forest canopies in the Wet Tropics bioregion due to tropical cyclone (Photo credit: Mohammed Alamgir).

The MA noted that 15 out of 24 globally recognized ecosystem services are in a declining stage and climate change is one of the main factors causing this decline (MA, 2005). A similar trend has also been reported from the temperate forested landscapes (Schröter et al., 2005) and from forests in California (Shaw et al., 2011). We are still unclear about what will be the trend for tropical forest ecosystem services under future climate change scenarios. Over the period 1880 to 2012, the mean global surface temperature has increased by 0.85°C (IPCC, 2014b). It has also been projected that global mean surface temperatures will increase a further 0.3°C to 1.7°C, 1.1°C to 2.6°C, 1.4°C to 3.1°C, and 2.6°C to 4.8°C by the end of this century (2081-2100), relative to

1986–2005 under RCP (Representative Concentration Pathway) 2.6, RCP4.5, RCP6.0 and RCP8.5 respectively with considerable anomalies in the rainfall (Figs. A and B) (IPCC, 2014b). Therefore, it can be hypothesised that climate change will substantially affect the ecosystem services supply in this century.

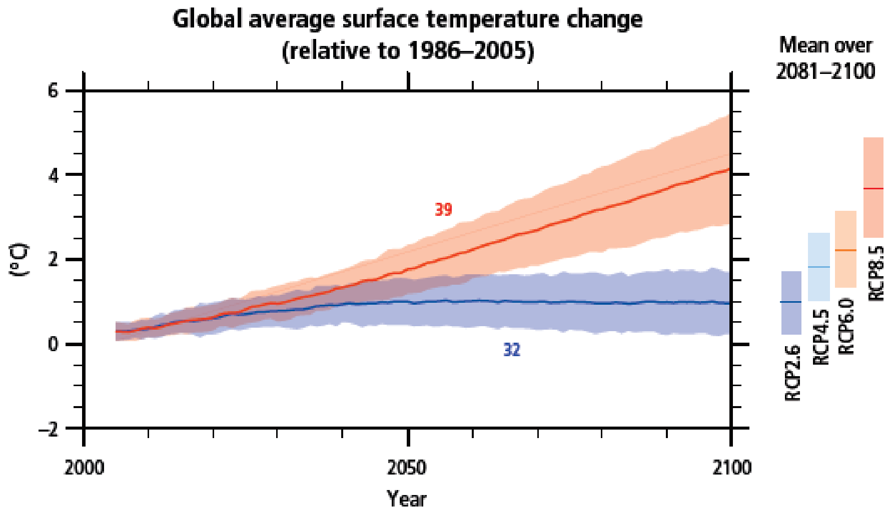

Figure A. Projected global average surface temperature change over 2006 to 2100 relative to 1986-2005 (Adapted from (IPCC, 2014b), page 11)

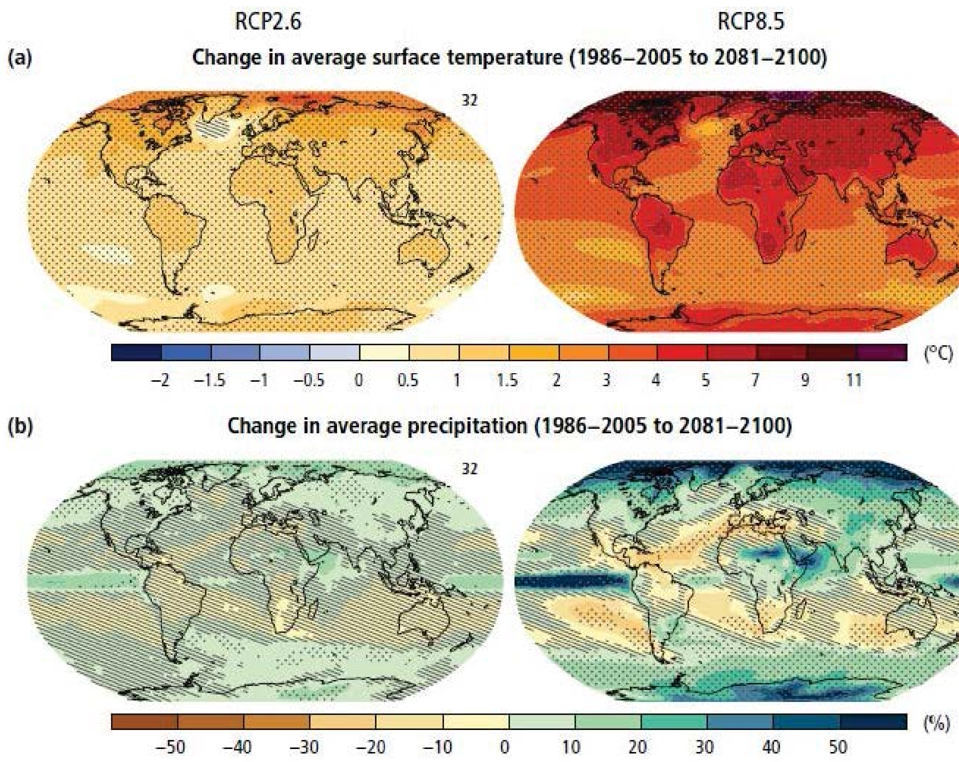

Figure B. Projected global average surface temperature (a) and rainfall change (b) (Adapted from (IPCC, 2014b), page 12)

Australia is one of the 'mega-biodiverse' countries in the world, comprising a large variety of ecosystems (Pittock et al., 2012) and its natural ecosystems are one of the most vulnerable sector to climate change (Smith and Ash, 2011). Under the latest climate change projections (IPCC, 2014b) (Fig. B), the ecosystem services supply from Australian forested landscapes is likely to be under more pressure in this century. However, the trend is yet to be explored for most terrestrial forest ecosystems, particularly for remaining areas of tropical forest of global significance.

This doctoral thesis is based on original research conducted in the Wet Tropics bioregion of northeast Australia, with a strong focus on ecosystem services assessment, interactions among the multiple ecosystem services, spatial distribution of ecosystem services, and the fate of ecosystem services under future climate change scenarios.

Figure 5. Collecting data from a rainforest in the Wet Tropics bioregion, Australia (Photo credit: Mohammed Alamgir)

Aims of the study

The overall aims of the study were:

I. to develop a science based approach to improve our understanding of ecosystem services supply and distribution; and the underlying processes and interactions; in the landscape level and at the level of different forest types (unit level); and

II. to provide science about the likely climate change effects on forest ecosystem services.

Under the stated aims, the present study had following five objectives:

Objective 1: to assess the supply of multiple ecosystem services from rainforests, sclerophyll forests and rehabilitated plantation forests, and how they are spatially distributed across the landscape.

Objective 2: to investigate the interactions among multiple ecosystem services and how the supply of one ecosystem service is influenced by others.

Objective 3: to evaluate the capacities of different Land Use and Land Covers (LULC) to supply ecosystem services, and to identify the spatial congruence between ecosystem services and biodiversity in the landscape.

Objective 4: to compare the carbon storage among rainforests, degraded rainforests and sclerophyll forests and to determine the driver of carbon storage in this tropical forested landscape.

Objective 5: to determine the likely effects of climate change on ecosystem services supplied from rainforests and sclerophyll forests.

Dissertation structure and outline

I have written this thesis as standalone publications. Chapters one and two have been published in peer reviewed journals; chapter three, four, and five have been submitted to journals and are in review (current status as on 26.04.2016); and chapter six will be submitted shortly. Consequently, some necessary overlaps between the chapters have occurred, particularly in the materials and methods section (i.e. study area, data collection and ecosystem services assessment). In addition, some plots are not relevant to the objectives of some chapters. Therefore, the number of plots used differs between chapters. However, the size of each plot is same (50m×10m).

CHAPTER 1: In this chapter I seek to understand the knowledge gaps on ecosystem services science in the Australian context, especially under future climate change. This chapter is based on the meta-analysis of the published papers found in scientific data bases e.g. *Web of Science, Scopus* etc., through a systematic review. I also use climate change modelling to investigate future climate change in the ecosystem services rich areas across Australia.

Based on:
Alamgir, M., Pert, P.L., Turton, S.M. (2014) A review of ecosystem services research in Australia reveals a gap in integrating climate change and impacts on ecosystem services. *International Journal of Biodiversity Science, Ecosystem Services & Management* 10 (2), 112-127.

CHAPTER 2: In this chapter I assess and compare the supply of eight ecosystem services from rainforests, sclerophyll forests and rehabilitated plantation forests. I evaluate the correlation among multiple ecosystem services. I also investigate the variation of supply of those eight ecosystem services across environmental gradients and finally, I identify the hotspots of supply of multiple ecosystem services in the landscape. The analysis of this chapter is based on forest vegetation attributes data collected from the Wet Tropics bioregion and also spatial data collected from the Wet Tropics Management Authority, Australia.

Based on:

Alamgir, M., Turton, S.M., Macgregor, C.J., Pert, P.L. (2016) Assessing regulating and provisioning ecosystem services in a contrasting tropical forest landscape. *Ecological Indicators* 64, 319-334.

CHAPTER 3: This chapter focuses on the capacities of the different Land Use and Land Covers (LULC) to supply multiple ecosystem services. The analysis is based on an expert workshop arranged at the James Cook University, Cairns campus. I show the spatial congruence between high-potential multiple ecosystem services and biodiversity at the landscape level.

Based on:

Alamgir, M., Turton, S.M., Macgregor, C.J., Pert, P.L. (in review). The capacity of a heterogeneous landscape to supply ecosystem services, and spatial congruence between ecosystem services and biodiversity. *Land Use Policy.*

CHAPTER 4: In this chapter I investigate the pattern, synergies and trade-offs among multiple ecosystem services supplied from rainforests, sclerophyll forests, and rehabilitated plantation forests of the Wet Tropics bioregion. I use forest vegetation data collected from the Wet Tropics bioregion.

Based on:

Alamgir, M., Turton, S.M., Macgregor, C.J., Pert, P.L. (in review, second revision submitted). Ecosystem services capacity across heterogeneous forest types: Understanding the interactions and suggesting pathways for sustaining multiple ecosystem services. *Science of the Total Environment*

CHAPTER 5: I estimate above ground biomass carbon storage in the rainforests, degraded rainforests and sclerophyll forests in the Wet Tropics bioregion. I also identify the forest structural features which act as main drivers to store more above ground biomass carbon in the tropical forested landscape.

Based on:

Alamgir, M., Campbell, M.J. Turton, S.M., Pert, P.L., Edwards, W., Laurance, W.F. (in review, first revision submitted). Degraded tropical rainforests possess valuable carbon storage opportunities in a complex, forested landscape. *Scientific Reports*.

<u>CHAPTER 6</u>: This chapter represents the spatial distribution of ecosystem services across rainforests and sclerophyll forests, and the likely effects of climate change on the supply of ecosystem services. I use forest vegetation attributes data to evaluate the spatial distribution of ecosystem services, climate change modelling to show future climate change scenarios in the Wet Tropics bioregion, and determine likely effects on ecosystem services using automatic regression models between ecosystem services and net primary productivity.

Based on:

Alamgir, M., Turton, S.M., Macgregor, C.J., Pert, P.L. (in prep.). Climate change effects on forest ecosystem services. *Ecological Indicators*

OVERVIEW OF CHAPTER 1

This chapter is based on a paper published in *International Journal of Biodiversity Science, Ecosystem Services & Management* [1] with minimal formatting change (reprint of the paper in appendix).

In this chapter I conduct a detailed and comprehensive review of published scientific articles on ecosystem services of Australia. I complete a systematic search (title, keywords and abstract) in the global scientific databases, identify papers focused on the ecosystem services of Australia, and conduct in-depth review of those papers. Then I use climate change models to generate climate change scenarios and spatially show the areas where most of the ecosystem service studies have been conducted and how future climate will be changed in those areas. Key findings from this study are interpreted to identify knowledge gaps on ecosystem services research in Australia.

[1] **Alamgir, M.**, Pert, P.L., Turton, S.M. (2014) A review of ecosystem services research in Australia reveals a gap in integrating climate change and impacts on ecosystem services. *International Journal of Biodiversity Science, Ecosystem Services & Management* 10 (2), 112-127

CHAPTER 1

A REVIEW OF ECOSYSTEM SERVICES RESEARCH IN AUSTRALIA REVEALS A GAP IN INTEGRATING CLIMATE CHANGE AND IMPACTS ON ECOSYSTEM SERVICES

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The rainforest in the Wet Tropics: one of the oldest rainforest on the Earth (Photo credit: Mohammed Alamgir)

ABSTRACT

Ecosystem services (ES) are the benefits people obtain from ecosystems. A substantial part of human wellbeing is dependent on the sustainable flow of ES. Climate change, economic growth and an increasing human population has placed greater pressures on global ES. Australia's ecosystems are amongst the most vulnerable sectors to climate change. Hence, a comprehensive review is necessary to explore ES research that integrates climate change impacts. Our review reveals that ES research in Australia, stimulated in the early 2000's, has continued to increase consistently after the Millennium Ecosystem Assessment. Australian ES research has primarily focused on the impact of land use change and management, policy and governance issues, but less on the impact of climate change on ES. Climate change models show that climate will threaten most of the main ES in Australia by 2050. For the sustainable management of these ES - incorporating climate change - ecosystem and ES specific adaptations are suggested as the best sustainable policy tools for the future. Therefore, further research needs to incorporate climate change and ES for evidence based sustainable management of Australia's ES. We provide the following recommendations for future ES research: (i) evaluating the extent and trend of climate change impacts on ES through consideration of different climate change scenarios; (ii) preparing vulnerability maps of important ES that are likely to be sensitive to climate change; and (iii) developing ecosystem and ES specific adaptations to climate change that involve key stakeholders.

Key words: climate change, ecosystem services, adaptation, stakeholder involvement, land use change, Australia

1.1. Introduction

Ecosystem services (ES) are the benefits people obtain from ecosystems (MA, 2005). They are highly valuable but go largely unrecognised by society (Costanza et al., 1997), and have been described as nature's gift to households, communities and economies (Boyd and Banzhaf, 2007). ES provide many necessities to societies (such as food and clean water) (MA, 2005), and form a distinct relationship between ecosystems and society (Metzger and Schröter, 2006). Substantial parts of human wellbeing depend on the flow of ES (Costanza et al., 1997; Kremen and Ostfeld, 2005). Human wellbeing, through the use of ES, is the core issue in the ES concept (Boyd and Banzhaf, 2007; Costanza et al., 1997; Daily, 1997; Fisher et al., 2009; MA, 2005). As global populations increase so does the ever increasing use of ES (Carpenter et al., 2009). Economic growth, high population growth, increasing global consumption patterns and climate change have placed significant pressures on ES (Seppelt et al., 2011; Shaw et al., 2011; Vitousek, 1997; Williams et al., 2003). Additionally, land use change has resulted in large scale changes in the reliable supply of ES (Schröter et al., 2005). The 'Millennium Ecosystem Assessment' (MA) (MA, 2005) has ascertained that 15 of the 24 recognized ES are in declining stages across the globe. Deterioration of ES will certainly affect human wellbeing, as there is an innate linkage between them (Shaw et al., 2011).

ES research has emerged as an important research issue over the past decade (Fisher et al., 2009), but is still considered to be at an evolving stage (Carpenter et al., 2006; Fisher et al., 2009; Sachs and Reid, 2006). The MA (2003, 2005) was the first international dynamic and integrated document that reported on ES research globally. It established ES as a policy tool for sustainable natural resource management (Seppelt et al., 2011) as well as providing scientific evidence for policy makers about the consequences of changes of ES to human wellbeing (Pert et al., 2010). Scientists and policy makers have continued to conduct further ES research in recent years (Fisher et al., 2009). For example, Seppelt et al.'s (2011) global review on ES studies evaluated the current trend, spatial distribution, weakness and future direction of ES research, whilst Egoh et al. (2007) completed a global review on ES studies, focusing on conservation assessment.

Australia has been described as one of the "mega-biodiverse" countries in the world, containing a very wide range of species and ecosystems (Pittock et al., 2012). Within Australia, the term ES has been widely used since the 2000s (Pittock et al., 2012). Distinguished pioneer publications in ES in the late 1990s (Costanza et al., 1997; Daily, 1997) inspired early ES research in Australia (Pittock et al., 2012). Since then, several studies on ES have been conducted and significant investment has been made in ES research in Australia (Abel et al., 2003; Binning et al., 2001). Currently, incorporation of ES in many different environmental policies is very common in Australia (Pittock et al., 2012; Wallace, 2007). Ensuring a continuous supply of ES requires effective conservation and management of critical ecosystem processes (van Jaarsveld et al., 2005). Managing sustainable ecosystems is a challenge for Australia (Pittock et al., 2012) due to the diverse and complex natural ecosystems found across the continent. Sustainable management of ES is as complex, if not more than ecosystem management (Kremen, 2005). Climate change further adds to the challenges and complexity of the sustainable management of ES. Scientists have revealed that natural ecosystems are one of the most vulnerable sectors to climate change in Australia (Stafford Smith and Ash, 2011). Regardless of the different aspects of ES research undertaken, Pittock et al. (2012) addressed the state of ES knowledge in Australia and their policy implications. More recently, Plant and Ryan (2013) also used a review of ES research in Australia in their research which provided a snapshot of the trend of ES research in Australia.

Our comprehensive and detailed review of ES research in Australia will help policy makers, natural resource managers and scientists identify research gaps and prioritize research aimed at the sustainable management of ES under climate change scenarios across the continent. Our review aims to summarise and identify the current trend, distribution and core facts pertaining to ES studies across Australia. Climate change maps are also generated for 2030 and 2050 across Australia for IPCC (2007) SRES B1 and IPCC SRES A1F1 emission scenarios using OzClim climate change model developed by CSIRO. We use ES study regions as a demonstrative example of the possible effects of climate change on ES. Based on this example, we further identify the important issues needed to be considered in ES research in Australia in the face of human-induced climate change.

1.2. Methods

1.2.1. Literature inventory

Our study is based on publications found in the *ISI Web of Knowledge*, *Scopus*, *ScienceDirect* and *Google Scholar* databases searched in July 2013. We have conducted a title, keywords and abstract search in *Scopus* using the words "ecosystem service(s) AND Australia" and found 185 journal articles (Fig. 1.1a). Subsequently we made a quick review of these journal articles and found 37 articles had a major focus on ecosystem services. We then performed a title search in the *ISI Web of Knowledge*, *ScienceDirect* and *Google Scholar* where search terms included the words "ecosystem service(s) AND Australia", "ecosystem valuation AND Australia", "ecological service (s) AND Australia", "environmental service (s) AND Australia" (further details of search is provided in Table 1.1). We only considered peer reviewed journal articles and conference papers published in the last 20 years (1993-2013), with other publications manually removed. In all databases searched, results with environmental service(s) were manually screened, as there were some publications unrelated to ES. In this way we found an additional 15 publications. We extended our exploration to most of the well-accepted databases for the inclusion of all possible publications in the present study. Furthermore, we have included nine pertinent ES research publications in Australia (Abel et al., 2003; Binning et al., 2001; Bryan et al., 2011a; Bryan et al., 2009; Bryan et al., 2011b; Chong, 2012; Cork et al., 2012; Crossman et al., 2010; Maher and Thackway, 2007) as these publications were missed due to our search criteria and were considered instrumental in ES research beginnings in Australia.

Table 1.1. Features of inventory, searched databases and terms and the elaboration of categories used in analysis.

Features/Categories	Description
Features of inventory	
Searched databases	*ISI Web of Knowledge, Scopus, ScienceDirect* and *Google Scholar*
Searched words	*Title, Keywords and Abstract search in Scopus:* ecosystem service (s), Australia (additional field) Title search in *ISI Web of Knowledge, Scopus, ScienceDirect* and *Google Scholar*: ecosystem service (s), ecosystem valuation, ecological service (s), environmental service (s), Australia (additional field)
Elaboration of categories used in analysis	
Scale of the study	*Local*: study based on a project level, *Catchment*: study based on a single catchment, *Regional*: study based on more than one catchment, region or state, *National*: study based on more than one state, policy, approaches and theoretical analysis without mentioning any particular territory.
Studied ecosystem	Categorized according to dominant land uses, Other- review articles, approaches and theoretical studies
Offsite effects	Local decision can effect ES supply of another place
Trade-off analysis	Change in different ES as well as change in same ES between present and future, *Other*- review articles
Stakeholder involvement	Active involvement of stakeholder in research process, *Other*- review articles
Scenarios	*Land use*- plausible land use/management options in future, *Political*- plausible policy/planning options, *Other*- review articles, approaches and theoretical studies
Indicators used	*Biophysical*- publications assessed the distribution and attributes of ES, *Economic*- publications analysed monetary value of ES, *Social*- publications reflected community perceptions, social preferences to ES and community evaluated interrelationship among ES, *Policy*- publication used different existing policies to frame ES as well as discussed possible future policy issues related with ES, *Combined*- publication used more than one above categorical indicator, *Other*- publication described theories and approaches of ES assessment.
Data source	*Primary*- direct data collection and measurement of ES, *Secondary*- proxy presentation of ES by map, Other- publications discussed theoretical and analytical approach

Altogether we have selected 61 publications that were reviewed thoroughly (see Appendix 1.1) and then categorized (elaboration of each category in Table 1.1) into temporal distribution (Fig. 1.1b); spatial distribution (Fig. 1.2); different factor levels (Fig. 1.3); studied ecosystems and ecosystem services (Figs. 1.4 and 1.5); and focused issues of ES research in Australia (Fig. 1.6). Given that the OzClim climate change model uses surface air temperature and not ocean temperature, our analysis excluded ES research conducted in marine ecosystems, such as the Great Barrier Reef of Queensland e.g. Bohensky et al. (2011).

1.2.2. Generating climate change scenarios

Climate change scenario maps (mean annual temperature change and annual rainfall change) were generated from the base year (1990) across Australia for low emission scenarios (IPCC SRES B1) and high emission scenarios (IPCC SRES A1F1) for the years 2030 and 2050 using the advance module of *OzClim*[2] climate change model. The distribution of most ES studies regions (Fig. 1.7) in the output map was used as an example to represent the threat of climate change to ES.

1.3. Results

1.3.1 Temporal and spatial distribution

Our search terms revealed that the first Australian publication of ES was published in 1998 although we had fixed the search year since 1993. Starting with only one publication in 1998 ES research in Australia remained somewhat static until 2004. It has increased consistently since 2007 with publication numbers peaking in 2010 with 32 publications (Fig. 1.1a). The reviewed articles we primarily focused for our discussion occurred from 2001 to 2013 (Fig. 1.1b).

[2] CSIRO developed the OzClim climate change model to generate climate change scenarios across Australia (http://www.csiro.au/ozclim/home.do).

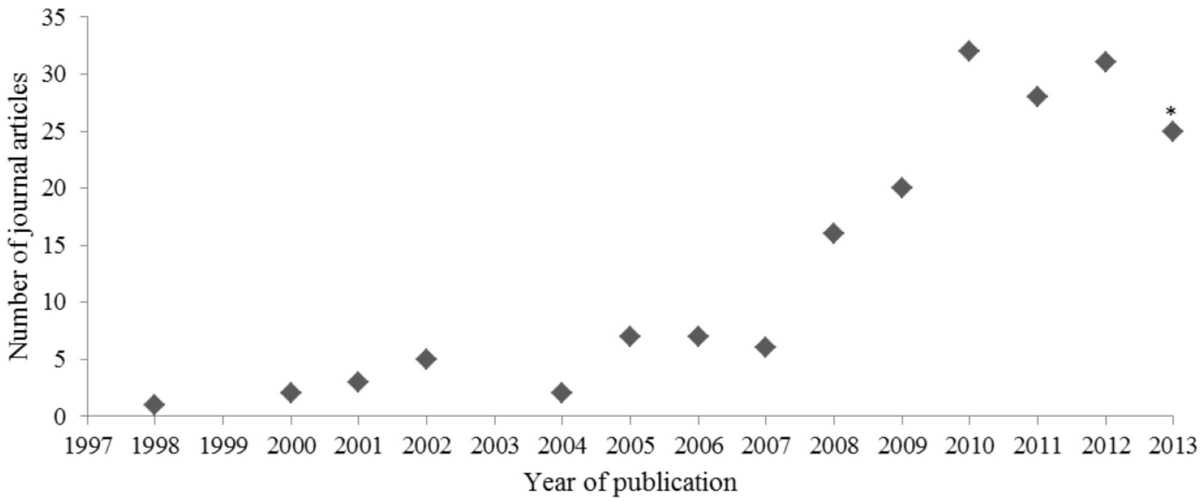

Figure 1.1a. Number of journal articles since 1993 found using the search terms "ecosystem service (s) and Australia" in Title, Abstract and Keywords in Scopus in July 2013 (*until July 2013).

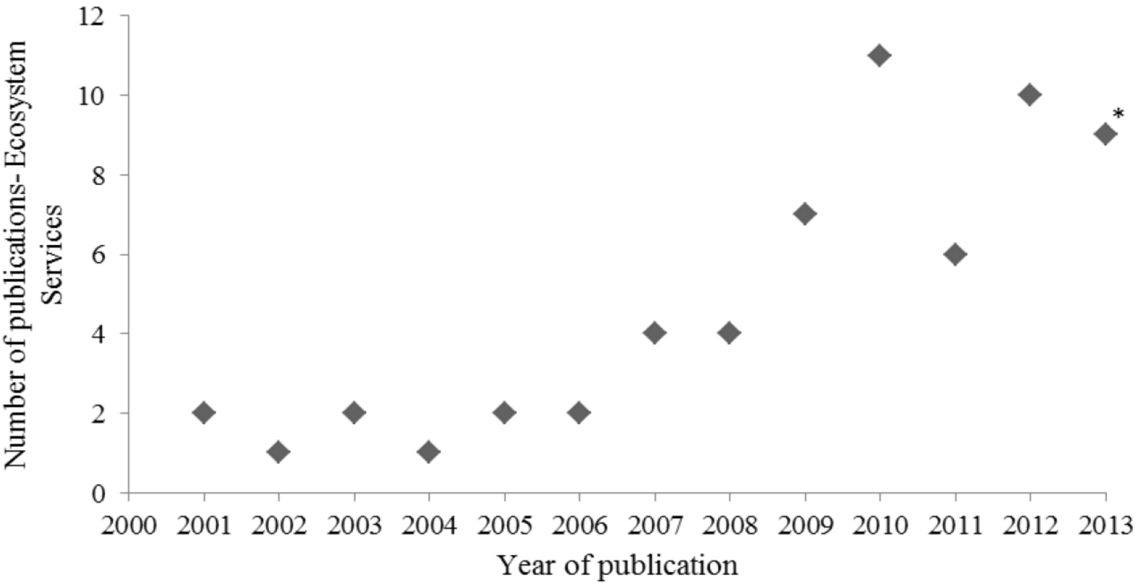

Figure 1.1b. Distribution of publications reviewed in the present study (*until July 2013).

South Australia (n=16) appears to be the centre of ES research in Australia, followed by Victoria (n=13) and Queensland (n=10). A numbers of ES research publications in New South Wales (n=6), Northern Territory (n=6) and Western Australia (n=3) were found whilst very few occurred in the Australian Capital Territory (n=1) and Tasmania (n=1) (Fig. 1.2). Most of the studies in South Australia were conducted in the Murray-Darling Basin region (13 of 16). ES research is also highly concentrated in areas such as: the Goulburn Broken Catchment (n=5); Glenelg Hopkins Catchment (n=2) of Victoria; Gwydir Catchment of New South Wales; Tully Murray Catchment (n=2) Far North Queensland; South East Queensland (SEQ) region (n=4); Wet Tropics of Queensland (n=2); and the Tropical Savanna Catchment of Northern Territory (n=3). Our study revealed that the 'catchment' scale was the most popular physical boundary of ES research representing 44% of total ES research, followed by the regional scale 23% (Fig. 1.3). Although 18% of ES research has been focused at the national scale (Fig. 1.3), most of these were either review work or presented theoretical aspects and conceptual frameworks of ES rather than on-ground ES research.

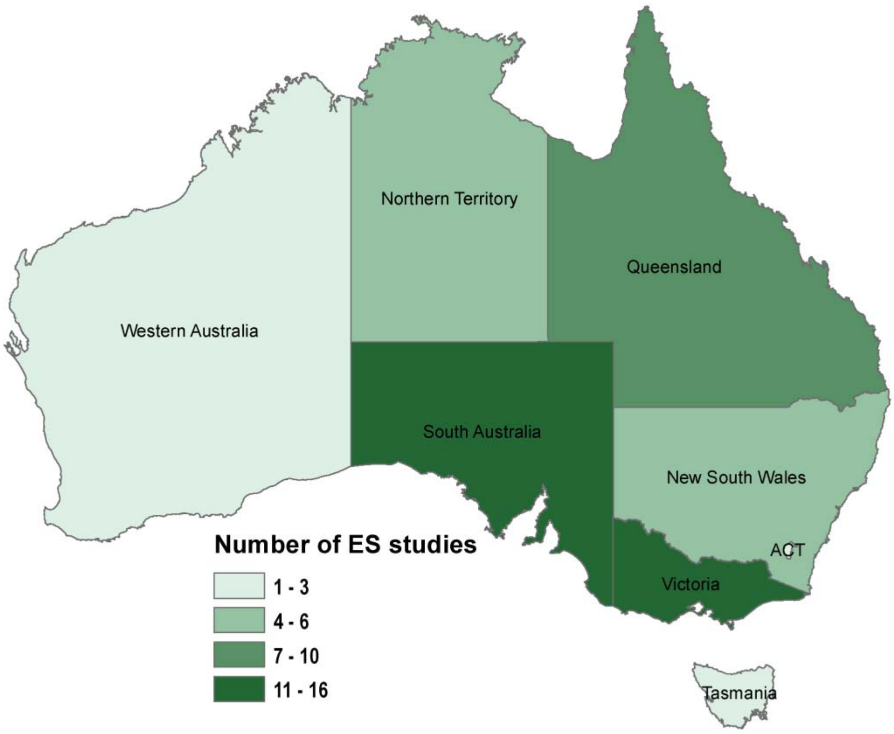

Figure 1.2. Spatial distribution of ecosystem services research across Australia which are reviewed in the present study.

1.3.2. Data sources

Our study revealed that 48% of ES research was derived from primary data sources. These studies are based on direct field observations, measurements, scoring and ranking of ES. Furthermore, 28% of ES research originated from secondary data sources. These studies used proxies for ES, such as land use/land cover maps or other proxy data. Moreover, 25% of ES studies were derived from other data sources (Fig. 1.3) that presented theoretical, conceptual or policy framework approaches for assessment, understanding, planning and management of ES, primarily based on reviews.

1.3.3. Use of indicators

Our study revealed that 25% of ES research has utilized biophysical indicators, whilst 20% used economic indicators (monetary values), 15% social indicators and 25% combined indicators (Fig. 1.3) (see Table 1.1 for further elaboration of indicators). Furthermore, 7% of ES studies have dealt with policy issues exclusively based on reviews. In the past, most of the research used combined indicators. For example, in the Goulburn Broken Catchment of Victoria indicators were combined in the "*Ecosystem Services Project*" (Abel et al., 2003; Binning et al., 2001; Cork, 2003; Cork and Proctor, 2005; Proctor et al., 2002). More recently a model has been developed supporting investment decision making for natural capital and ecosystem services. This has been applied in the Murray-Darling Basin where stakeholders were involved in quantifying the management priorities for capital assets and ecosystem services (Bryan, 2010). Furthermore, cost effective hotspots for natural capital restoration (species and ecosystems, soil and water resources, atmosphere) have been identified in the Lower Murray region of south-eastern Australia (Crossman and Bryan, 2009).

In a biophysical context, researchers have evaluated the spatial distribution of ES (Baral et al., 2013a; Butler et al., 2011; Pert et al., 2010); examined the relative importance of ES and ecosystems (Baral et al., 2013a; Pert et al., 2010); identified ES, ecosystem functions and indicators (Butler et al., 2011); and assessed the impact of land management on ES (Collard and Zammit, 2006; Cork et al., 2012). Furthermore, Ens (2012) has used Cyber Tracker Technology to monitor ecological outcomes of payments for environmental services, and Kragt and Robertson (2012) simulated the possibility of ES production in association with agricultural production. In an economic context, scientists have used spatial approaches for the economic valuation of ES (Baral et al., 2009), environmental flow provision, opportunity cost of ES (Karanja et al., 2008), and for assessing the cost of running auction-based approaches for purchasing environmental services (Lowell et al., 2007). Greiner et al. (2008) presented the conceptual challenges of payment for environmental services, and Zander and Garnett

(2011) evaluated the intention of Australians to pay Indigenous Australians for the conservation of their land using nation-wide interviews.

In a social context, scientists have evaluated Natural Resource Management (NRM) practitioners' understandings about the concept of ES (Plant and Ryan, 2013). Furthermore, Maynard et al. (2011) developed an ES framework for ES planning and NRM engaging various stakeholders. This framework is widely recognized across Australia for ES planning, management and assessment. Scientists have also mapped ES using social values (Raymond et al., 2009) and identified priority areas for ES management and investment decisions (Bryan et al., 2009; Bryan et al., 2010a). Researchers have also assessed the role of ES to the well-being of Indigenous Australian communities (Kaur, 2007). Furthermore,van Riper et al. (2012) has recently conducted interviews with recreationists evaluating 12 different types of social values: aesthetic, biological diversity, cultural, economic, future, historic, intrinsic, learning, life sustaining, recreation, spiritual and therapeutic values.

In a policy context, scientists have conceptualized ES, conferred different existing policies and examined the possibility of using ES for human wellbeing (Pittock et al., 2012). Scientists have also discussed policy consistency regarding ES in Australia (Pittock et al., 2012) as well as the ambiguity about ES rights (Tovey, 2008), and the associated politics in environmental services marketing in Australia (Verran, 2011). Analyses have also compared the historical inclusion of ES in Melbourne's strategic spatial plans with Stockholm's strategic spatial plans (Wilkinson et al., 2013). In the other indicators context, at the very early evolving stage of ES in Australia, scientists created a framework for NRM in Australia, in 2001, which accommodated the ES concept (Cork et al., 2012). The Australian Government has also published a report summarizing available approaches and tools that are being used by State and Federal Government agencies in Australia for ES provided by vegetation and used for assessment of ES with an emphasis on production landscapes (Maher and Thackway, 2007).

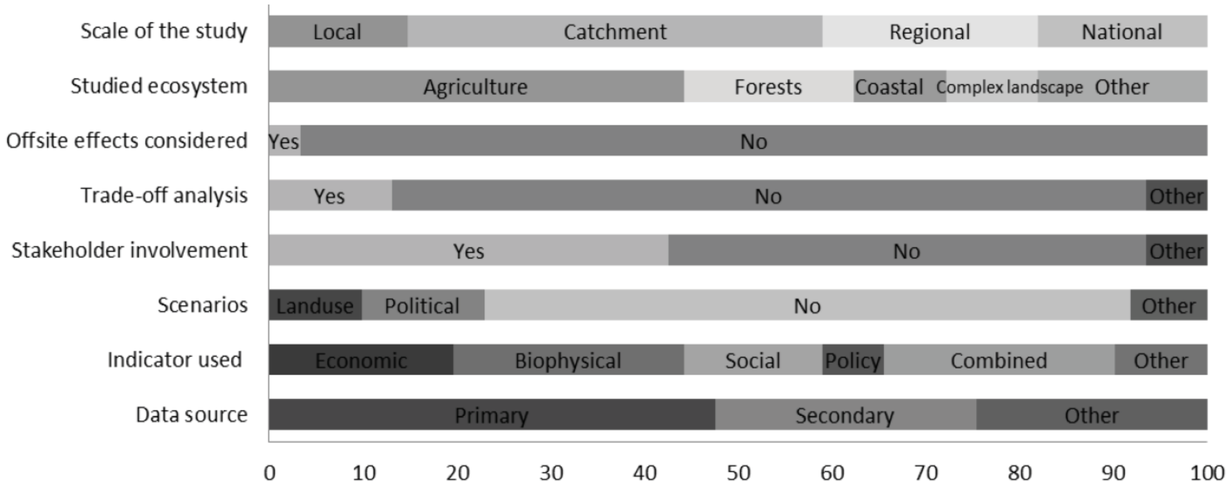

Figure 1.3. The percentage of ecosystem services research according to different factor levels analyzed.

1.3.4. Components of ES research

Global ES research studies commonly utilize scenario analyses, stakeholder involvement, tradeoff analyses and off-site effects. Scenarios are plausible options for the future. There are three types of scenarios usually considered in ES research: i) land use; ii) political/policy; and iii) climate change scenarios. 23% of the ES research we examined used scenarios (13% political scenarios and 10% land use scenarios). Scientists have identified the quantitative and qualitative variation of ES supply considering different land use scenarios (Butler et al., 2011; Kragt and Robertson, 2012), identified alternative management options for investment decisions through considering effective ES mangement (Bryan et al., 2009), and analysed alternative management scenarios for enhanching ES and biodiversity benefits (Bryan and Kandulu, 2009). Moreover researchers have analyzed ecological, economic and social benefits that might be derived from ES through regarding policy options across different catchments in Australia (Abel et al., 2003; Bryan et al., 2009; Cork, 2003; Cork and Proctor, 2005; Proctor et al., 2002). Scientists have also compared two scenarios as ecologically weighted efficient and socially weighted efficient for investment decisions in environmental flow (a regulating ES) considering ecosystem health (Bryan et al., 2013).

Any group or individual who can affect or is affected by ES are known as stakeholders (Heina et al., 2006). Our study revealed that stakeholders have been involved with 43% of ES research. Those commonly involved in ES process are: scientists, (local) experts in their respective fields, community leaders, NRM bodies, (local) community people and tourists. Researchers have utilized expert knowledge to rank the relative capacity of ES (Baral et al., 2013a), for developing approaches for ES planning and assessment (Maynard et al., 2011), and standardizing ecosystem function data layers for ES mapping (Petter et al., 2013). Additionally, Australian ES researchers have utilized community leaders' opinions and interviews for categorizing ES under different sectors and preparing different land use and policy scenarios for ES management (Abel et al., 2003; Binning et al., 2001; Bryan et al., 2011a; Cork, 2003; Cork and Proctor, 2005; Crossman et al., 2010; Proctor et al., 2002). Scientists have also utilized community representatives and NRM bodies to identify, map and value the ES in respective catchment areas (Abel et al., 2003; Bryan et al., 2009; Bryan et al., 2010a; Bryan et al., 2011b; Bryan et al., 2010b; Hatton MacDonald et al., 2013; Raymond et al., 2009) and conducted interviews about stakeholders' understandings of ES and the implications for NRM (Plant and Ryan, 2013). Others have used tourist interviews to identify the spatial distribution of ES and estimated several social values (van Riper et al., 2012) as well as asking the general public to evaluate their willingness to pay for ecosystem services (Zander and Garnett, 2011).

The degree of response of each ES to change varies according to those that are recognized as tradeoffs (Seppelt et al., 2011). Our study revealed that 13% of ES researchers had analyzed and performed tradeoff analysis between multiple ES due to land use change (Baral et al., 2013a; Butler et al., 2011; Kragt and Robertson, 2012), as well as policy change (Abel et al., 2003; Cork, 2003; Cork and Proctor, 2005). Scientists also analyzed tradeoffs between carbon sequestration and other multiple ES production using different planting options (e.g. monoculture plantation) (Perring et al., 2012). Our study revealed that offsite effects have only been considered in 3% of ES research outputs.

1.3.5. Studied ecosystems and ecosystem services

Agricultural ecosystems (44% of total studies) were the predominant ecosystem type where ES research had been conducted, whilst forest ecosystems emerged having 18% of total studies (Fig. 1.3). Furthermore, 10% of ES research has been conducted in coastal ecosystems and complex landscapes (Fig. 1.3). Dryland agriculture and Rangelands were the dominant land uses in agriculture ecosystems and complex landscapes. Additionally, wetlands, sugarcane, bushland, rainforests, and urban ecosystems were common in complex ES landscape-level studies, and production forests, rainforest and Eucalyptus plantations were common in forest ecosystems. Although a number of ecosystems occur in the Murray-Darling Basin of South Australia and Goulburn Broken Catchment of Victoria, we categorized them both primarily as agriculture ecosystems.

The number of individual ES covered in each study varied from one to more than thirty. We found that out of 61 studies, 10 included more than eight ES each, and another 10 studies included only one ES in each study (Fig. 1.4). While 24 studies had not specified any ES, these studies were mostly theoretical approaches, conceptual framework and policy perspectives. Many ES studies in Australia have focused on one or few ES (Pittock et al., 2012). Although several ES studies in Australia focused on few ES, our study revealed that a large number of ES were included in studies which were focused on ES economic and social values and policy analysis. The MA has not recognized 'biodiversity' as an ES - instead it represented biodiversity as a unique entity (MA, 2003). However, many scientists have subsequently assessed biodiversity as an ES. In Australia, 12 studies have assessed biodiversity as an ES (Fig. 1.4) (Abel et al., 2003; Baral et al., 2009; Baral et al., 2013a; Bryan and Kandulu, 2009; Butler et al., 2011; Cork, 2003; Curtis, 2004; George et al., 2012; Karanja et al., 2008; Perring et al., 2012; Wilkinson et al., 2013; Zander and Garnett, 2011). Our study revealed that 34 different ES, distributed over four MA (2003, 2005) categories (provisioning, regulating, cultural and supporting), have been studied (Fig. 1.5). Agricultural production (n = 21 studies, 28% of provisioning services), water regulation and climate regulation (n = 20 studies each, 19% studies of regulating services), and water provision (n = 16 studies, 21% studies of provisioning services) appear to be the most common of ES research in Australia, followed by soil erosion control, pollination (n = 15 studies each), nutrient cycling (n = 14 studies), and aesthetics (n = 13 studies). Although climate regulation represents 19% of studies in the regulating ES category (Fig. 1.5) it primarily includes carbon emission reduction, carbon sequestration, and carbon stock studies. For example Baral et al. (2013a) have assessed the spatial distribution of carbon stocks along with other ecosystem services in a complex production landscape of south-western Victoria. They have also spatially assessed the impacts of land use change on carbon stocks and other ecosystem services (water regulation, biodiversity, forage production, timber production and water provision) over the last 200 years. Porfirio et al. (2010) have estimated carbon storage in biomass and net ecosystem carbon exchange between the land surface and the atmosphere to quantify the potentiality of human modified landscapes to provide ecosystem services in the Australian Capital Territory region.

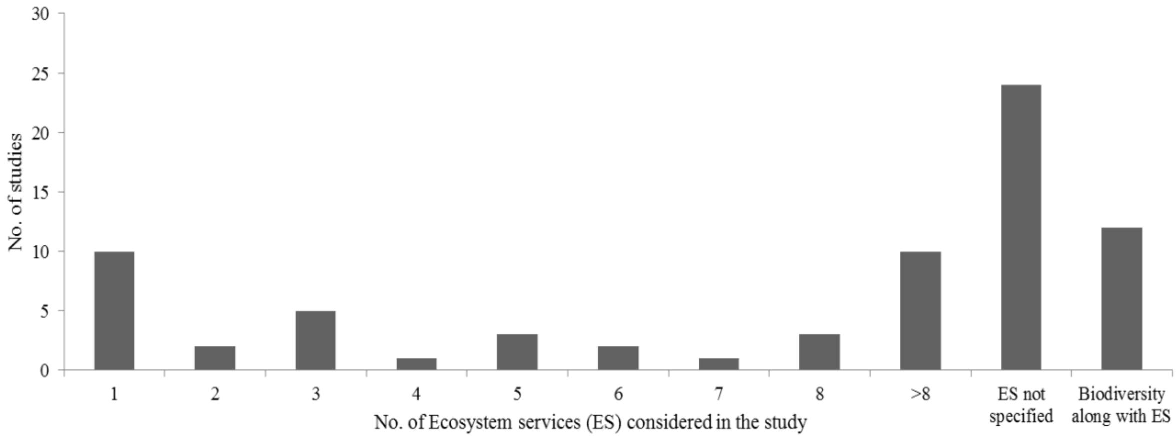

Figure 1.4. The number of ecosystem services considered in each study.

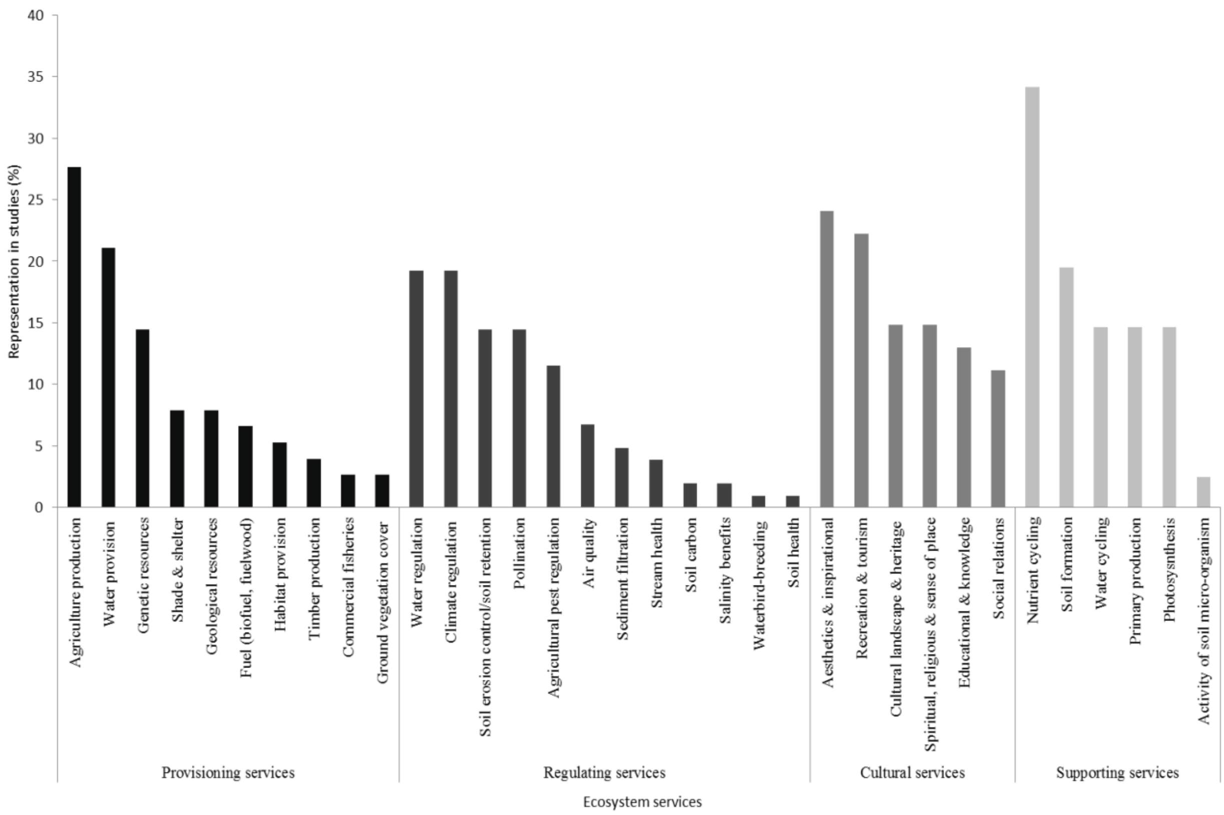

Figure 1.5. Representation of studied ecosystem services. Agriculture production includes crop yields, food & fiber production, forage production, improved grazing; Climate regulation includes carbon emission reduction, carbon sequestration, carbon stock; Recreation includes improved recreation, recreational fisheries; Nutrient cycling includes nitrogen supply, nutrient management.

1.3.6. Focused issues of ecosystem services research

About 48% of ES studies were focused on the ecological and economic impacts on ES due to land use change and change in land management approaches. Few studies (15%) have been conducted that focus on policy and governance issues. Furthermore, social valuation of ES has been covered in very few studies (8%) (Fig. 1.6). Noteworthy, in our research, was the absence of studies that focused on evaluating ES from a climate change perspective. We found no Australian studies that had examined the future trend of ES under different climate change scenarios, vulnerability of different ecosystems and ES, and available adaptation options. However, some scientists have evaluated the impacts of different climate change scenarios on alternative spatial policy options (Bryan et al., 2011a), mapped ecological values of habitat of threatened species due to climate change (Bryan et al., 2011b), analyzed the variation of nutrient retention in tidal mangroves with rainfall variation (Adame et al., 2010), considered species' responses to climate change as one of the indicators for investment decisions (Crossman et al., 2011), conceptualized the adaptive capacity through learning from historical examples (Bussey et al., 2012), and assessed usefulness of agroforestry systems for carbon sequestration and other ES in the face of climate adaptation and mitigation (George et al., 2012).

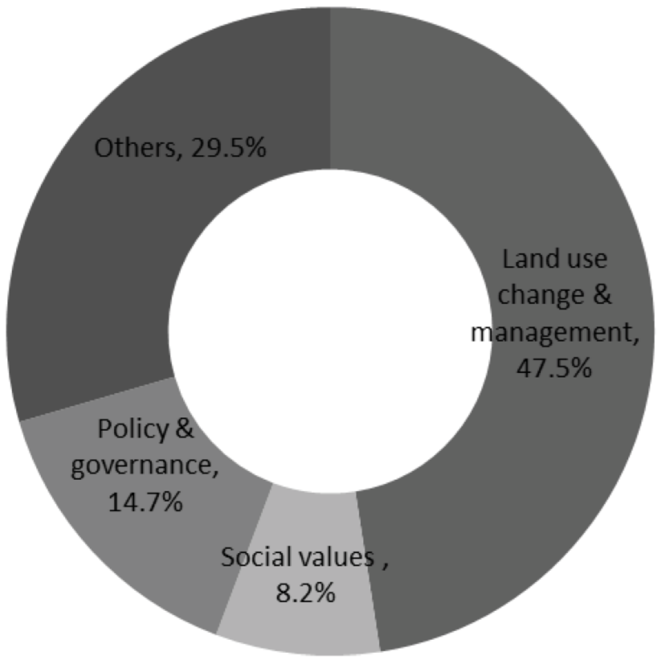

Figure 1.6. Focus of issues of ecosystem services research in publications.

1.4. Discussion

Studies of ES have attracted researchers worldwide after the assimilation of the Millennium Ecosystem Assessment (MA) (2003, 2005). ES research in Australia also gained further momentum after 2006, post the MA. Several collaborative ES studies have been conducted at the catchment scale across Australia (Pittock et al., 2012), and hence the 'catchment scale' has evolved as a popular spatial unit of ES research. The *"Ecosystem Services Project"* (http://www.ecosystemservicesproject.org) was one of the pioneer ES projects in Australia, implemented by CSIRO[3] in the Goulburn Broken Catchment, Victoria in 1999 (Cork, 2003). This project played a significant role

[3] Australia's national scientific research agency, CSIRO – Commonwealth Industrial Scientific Research Organisation.

in the early stages of ES research in Australia. At that time scientists used combined indicators such as biophysical, economic and social indicators for ES assessment and informing Australian policy decisions. More recently ES research has adopted spatial analyses using biophysical indicators.

Stakeholder involvement is one of the most common components of ES research that was also recognized in the MA (2003, 2005). When stakeholders are involved it increases the wider acceptance of ES planning and management (Maynard et al., 2011). A stakeholder engagement review conducted in the USA reported that stakeholder engagement is useful for better incorporation of public knowledge and values, conflict resolution, trust establishment and improved understanding of environmental problems (Beierle and Konisky, 2001). However, stakeholder engagement in planning is sometimes difficult when they have prior expectations from the institution and/or power in the current decision-making process (Spash, 2007). Stakeholders' attitudes and behaviours towards conservation actions depend on their level of knowledge and information (Lichtenberg and Zimmerman, 1999). The value of any ES depends on the stakeholders' views and needs (Vermeulen and Koziell, 2002). Therefore, ES oriented management actions should reflect the desire and aspirations of stakeholders. It is noteworthy that various stakeholders have been involved in a substantial number of ES research studies in Australia.

Tradeoffs can occur in temporal and spatial patterns (Steffan-Dewenter et al., 2007) due to feedback in ecological processes (Rodríguez et al., 2006). Tradeoffs occur between different ES as well as between the present and future supply of the same ES (Carpenter et al., 2006). Understanding tradeoffs, synergies, and interactions among multiple ES is important for making better informed NRM decisions (Bennett et al., 2009); hence tradeoff analysis is a popular approach for effective ES management and planning (Rodríguez et al., 2006). Furthermore, designating the physical boundary of an ES production area is always difficult. Sometimes ES production areas and ES benefit areas are different due to flow effects (Fisher et al., 2009). Local decisions can affect delivery of ES some distance away with significant offsite effects emerging (Seppelt et al., 2011). Therefore offsite effects need to be considered in ES management at the landscape level (Fisher et al., 2009), however offsite effects are not widely evaluated in ES research across the world (Seppelt et al., 2011). Our study also revealed that offsite effects had been incorporated in very few ES research studies.

Ecosystem services research is still in the evolving stage of development across the world. Globally, scientists have been assessing different aspects of ES, such as quantifying and mapping ES (Anderson-Teixeira et al., 2012; Deng et al., 2011; Egoh et al., 2008; Eigenbrod et al., 2010; Kalacska et al., 2008; Li et al., 2011; Naidoo et al., 2008), developing practical frameworks for the assessment of ES (Posthumus et al., 2010), describing the nature of relationships between ES and biodiversity (Egoh et al., 2009; Egoh et al., 2010) and developing models (such as InVEST) and web-based tools (such as ARIES) for ES analysis (Johnson et al., 2012; Nelson et al., 2009; Youn et al., 2011). A number of studies on these aspects have also been conducted in Australia (see Appendix, Figs. 1.3, 1.4, and 1.5 for details).

In the Australian literature, we found a number of ES research studies which assessed various land use change scenarios and policy/political scenarios (Fig. 1.3). Notably, we found no studies that had used ES in climate change scenarios. However, Bryan et al. (2011b) analyzed four different policy options: random, cheapest, the best for NRM and the most cost-effective to achieve NRM targets under future climate change scenarios but few ES are embedded into the NRM targets. Notably, ES research in Australia emphasizing other climate change issues like impacts, vulnerability, resilience and adaptation are also absent in the literature, whereas climate change impact on ES has recently been assessed in California and Europe (Ding and Nunes, 2014; Metzger and Schröter, 2006; Shaw et al., 2011). Shaw et al. (2011) have assessed the climate change impact on California's ES under IPCC (2007) high and low greenhouse gas emission scenarios using Dynamic Global Vegetation Model (DGVM). They have found that the provision and value of ES will decline under most of the future greenhouse gas trajectories. Ding and Nunes (2014) have recently modelled the impact of climate change on ES across European forests. They have found that climate change impacts on ES are regionally specific. They have also found a strong relationship between temperature and the value of ES; however the direction of the relationship may be either positive or negative depending on the type of ES under consideration. A similar study in Australia would contribute significantly to our knowledge of climate change impacts on Australia's ES, which are substantially lacking at present.

If we consider the regions where most of the ES research studies have been undertaken as the 'hotspots' for providing ES in Australia, climate change will significantly affect most of these hotspots, thereby affecting ES. In most locations, mean annual temperatures will rise 1- 2°C by 2030 and 2-3°C by 2050 from the base year (1990) for low emission scenarios (IPCC SRES B1), while 1-2°C by 2030 and 3-4°C by 2050 for high emission scenarios (IPCC SRES A1F1) (Fig. 1.7). Additionally, rainfall will decrease by 50-100 mm by 2030 and 100-150 mm by 2050 for IPCC SRES B1 scenarios. For high emission scenarios (IPCC SRES A1F1) rainfall will decrease by 100-150mm by 2030 and 150-200mm by 2050 from the base year 1990 (Fig.1.7). Similarly, researchers have also predicted a rising trend of mean annual temperatures across most of Australia, although annual rainfall and moisture patterns are likely to vary widely with geographic location (CSIRO and Australian Bureau of Meteorology, 2007; Medlyn et al., 2011; Wood et al., 2011). Historically, it has also been noted, that mean surface temperatures in Australia have increased by more than 1°C over the period 1910 to 2009, whereas the average global temperature has increased around 0.7°C over the past century (Braganza and Church, 2011). A decreasing trend of annual rainfall over most of the populated parts of Australia, as high as 50 mm/decade in some regions, has also been recorded from 1970 to 2011 (Bureau of Meteorology Australian Government, 2012).

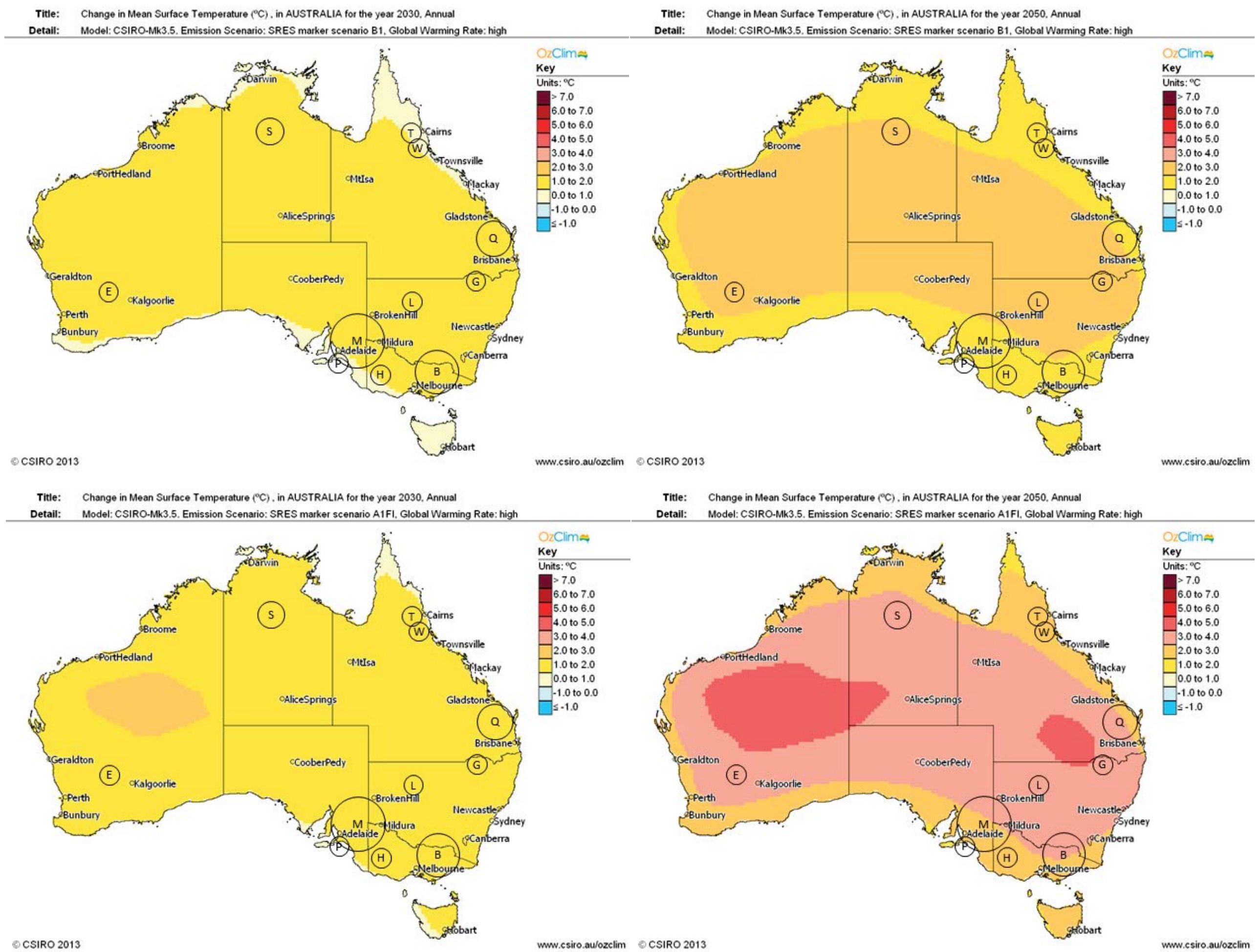

Figure 1.7. Climate change across Australia for IPCC SRES B1 and A1F1 emission scenarios for 2030 and 2050 (output from OzClim climate model) with inserted location of most ES researches, size of circle indicates the relative no. of ES research, B= Goulburn Broken Catchment (n=5), E= Wheat belt (n=2), G= Gwydir catchment (n=2), H=Glenelg Hopkins Catchment (n=2), L= Lachlan catchment (n=2), M= Murray Darling Basin (n=13), P= Myponga River Catchment (n=2), Q= South East Queensland Region (n=4), S=Tropical savanna (n=3), T= Tully Murray catchment (n=2), W=Wet Tropics (n=2).

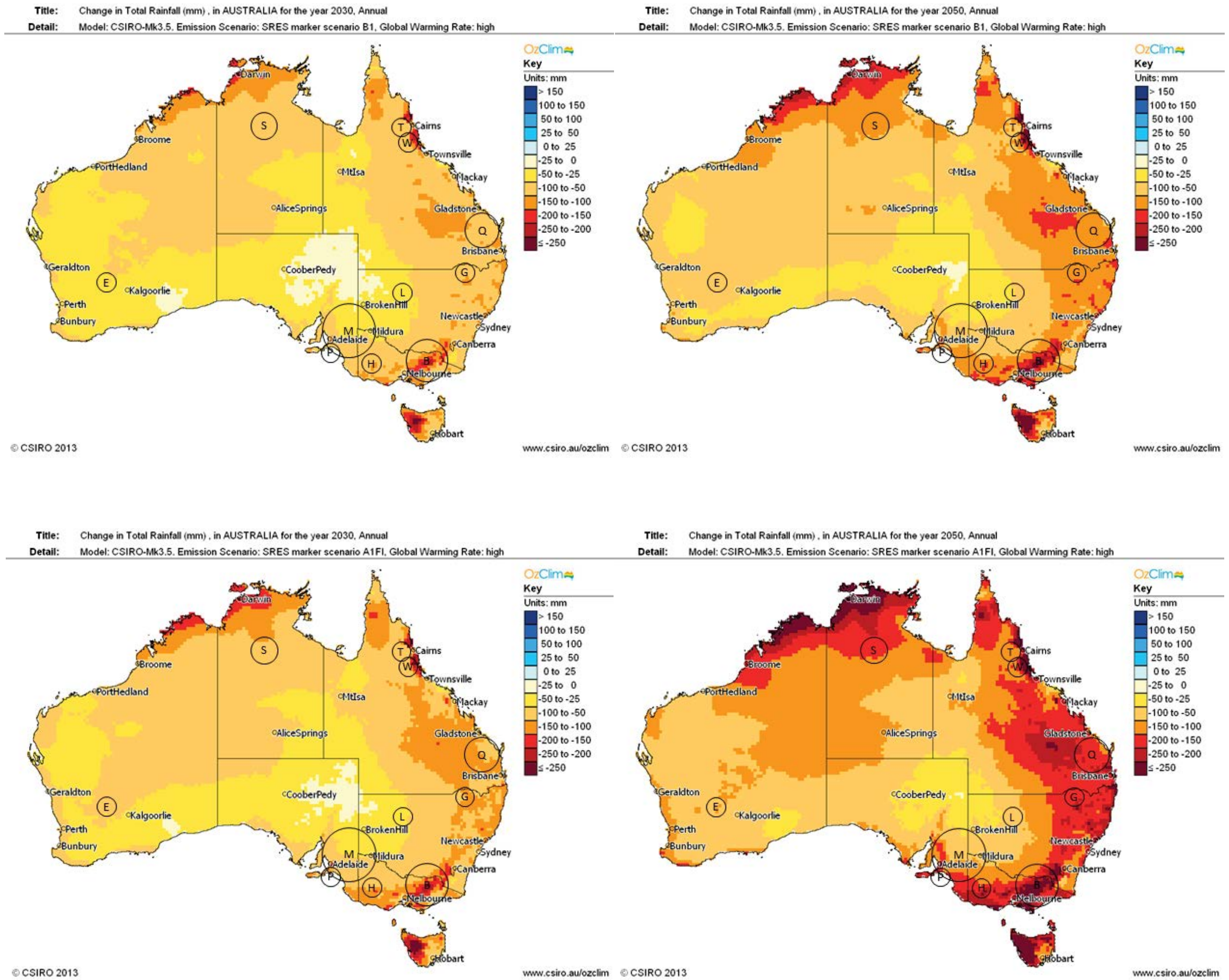

Figure 1.7. Continued.

It is notable that a number of ES research studies have been conducted across Australia over the last 20 years (Figs. 1.1a, 1.1b, Appendix 1.1). However, forest ecosystems and climate regulation ES have only been evaluated in a few studies (Figs. 1.3 & 1.5), regardless of the role that forest ecosystems play in climate regulation. Furthermore, ES research studies in Australia have covered a wide range of factors focusing principally on land-use change and management, whereas studies on integrating climate change and ES are significantly lacking to date. In Australia, impacts of land use change and management on ES would be largely positive when compared with other countries in the world, due to vegetation dominant land use, sustainable conservation, and effective policy implementation capacity and management excellence. Australia is a country of diverse ecosystems which provide significant and mostly unrecognised ES for community wellbeing. It is also apparent that social, ecological and economic values of these ES to the Australian economy are enormous, but not recognised by policy makers. Australia is highly vulnerable to climate change and contains many natural and relatively intact ecosystems that are considered among the most vulnerable ecosystems due to their low coping range and low adaptive capacity (Stafford Smith and Ash, 2011). Our study revealed that the combined effects of temperature rise and a decrease in rainfall threaten Australia's ES (Fig. 1.7). Therefore Australia's ES are probably under more threat from climate change than many other parts of the world and will be affected even more substantially in the future. As the magnitude of climate change is not uniform across all Australian ecosystems (Fig. 1.7), and resilience of all ES to climate change is not the same, the consequences of climate change for Australia's ecosystems and ES will vary, both spatially and temporally. For the sustainable management of Australian ES under climate change, ecosystems and ES specific adaptation would be the best sustainable policy tool providing adaptation options that are derived from evidence-based research integrating climate change and ES.

From our study we conclude that three key research issues need to be addressed to integrate climate change and ES in Australia. (i) evaluating the extent and trend of climate change impacts on ES considering different climate change scenarios; (ii) preparing vulnerability maps of important ES that are likely to be sensitive to climate change; and (iii) developing ecosystem and ES specific adaptations to climate change that involve different stakeholders.

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OVERVIEW OF CHAPTER 2

This chapter is based on a paper published in *Ecological Indicators* journal[4] with minimal formatting change (reprint of the paper in appendix). I address the first objective of the thesis- to assess the supply of multiple ecosystem services from rainforests, sclerophyll forests and rehabilitated plantation forests, and how they are spatially distributed across the landscape.

In this chapter I assess the quantity of five regulating ecosystem services: global climate regulation, air quality regulation, erosion regulation, nutrient regulation, and cyclone protection, and three provisioning ecosystem services - habitat provision, energy provision and timber provision across rainforests, sclerophyll forests and rehabilitated plantation forests. I also evaluate the variation of supply of those regulating and provisioning ecosystem services across environmental gradients, such as rainfall, temperature, and elevation. Furthermore, I investigate the relationships among those ecosystem services, and identify the hotspots of single and multiple ecosystem services supply in the Wet Tropics bioregion, northeast Australia.

[4] **Alamgir, M.**, Turton, S.M., Macgregor, C. J., Pert, P.L. (2016) Assessing regulating and provisioning ecosystem services in a contrasting tropical forest landscape. *Ecological Indicators* 64, 319-334.

CHAPTER 2

ASSESSING REGULATING AND PROVISIONING ECOSYSTEM SERVICES IN A CONTRASTING TROPICAL FOREST LANDSCAPE

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The canopy of an upland rainforest (Danbulla National Park, 761m above mean sea level)- a hotspot of multiple ecosystem services in the study area (Photo credit: Mohammed Alamgir)

ABSTRACT

Ecosystem services are the bridge between nature and society, and are essential elements of community well-being. The Wet Tropics, Australia, is environmentally and biologically diverse, and supplies numerous ecosystem services. It contributes to the community well-being of this region, Australian national economy and global climate change mitigation efforts. However, the ecosystem services in the region have rarely been assessed undermining strategic landscape planning to sustain their future flow. In this study, we attempted to: (i) assess the quantity of five regulating ecosystem services - global climate regulation, air quality regulation, erosion regulation, nutrient regulation, and cyclone protection, and three provisioning ecosystem services - habitat provision, energy provision and timber provision across rainforests, sclerophyll forests and rehabilitated plantation forests; (ii) evaluate the variation of supply of those regulating and provisioning ecosystem services across environmental gradients, such as rainfall, temperature, and elevation; (iii) show the relationships among those ecosystem services; and (iv) identify the hotspots of single and multiple ecosystem services supply across the landscape. The results showed that rainforests possess a very high capacity to supply single and multiple ecosystem services, and the hotspots for most of the regulating and provisioning ecosystem services are found in upland rainforest followed by lowland rainforest, and upland sclerophyll forest. Elevation, rainfall and temperature gradients along with forest structure are the main determinant factors for the quantity of ecosystem services supplied across the three forest types. The correlation among ecosystem services may be positive or negative depending on the ecosystem service category and vegetation type. The rehabilitated plantation forests may provide some ecosystem services comparable to the rainforest. The results demonstrated disturbance regimes (such as tropical cyclones) may have influenced the usual spatial trend of ecosystem service values. This study will assist decision makers in incorporating ecosystem services into their natural resource management planning, and for practitioners to identify the areas with higher values of specific and multiple ecosystem services.

Keywords: sclerophyll forest; rehabilitated plantation forest; environmental gradient; global climate regulation

2.1. Introduction

The goods and services human populations receive from an ecosystem are ecosystem services (Costanza et al., 1997; Daily, 1997; MA, 2005). Forests supply diverse ecosystem services like climate regulation, air quality regulation, and clean water, which are necessary for human well-being (Raymond et al., 2009). Besides enormous ecological values (Harrison et al., 2014; Nelson et al., 2009), the economic values of forest ecosystem services are incredible (Baral et al., 2014a; Costanza et al., 1997; Ninan and Inoue, 2013), although formally unrecognized. Most of the supply of global ecosystem services are declining (MA, 2005; Shaw et al., 2011), potentially undermining community well-being (Barbier, 2015; Mutoko et al., 2015; van Jaarsveld et al., 2005). Therefore, global efforts both in the science and policy arena have intensified to include ecosystem services in landscape management, planning and decision making. This is apparent by the formation of a number of organizations linking ecosystem services science and practice such as the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES http://www.ipbes.net), Ecosystem Services Partnership (ESP, http://www.fsd.nl) and A Community on Ecosystem Services (ACES).

One of the core challenges with the inclusion of ecosystem services in landscape management, planning, and decision making is the multi-scale and multi-dimensional complexity of assessing ecosystem services, (de Groot et al., 2010) including the spatial and temporal interactions among ecosystem services, land use and land cover, and management interruptions (de Groot et al., 2010; Palomo et al., 2013; van Oudenhoven et al., 2012). Furthermore, ecosystem services assessment is even more difficult in tropical forested landscapes, due to the more complex and dynamic nature. As ecosystem services science is a relatively new approach (Fisher et al., 2009), rapid assessment using proxies (like area) and secondary data are widely used (Seppelt et al., 2011), but these methods are unable to explain the variability of ecosystem services supply across the forest types and strong environmental gradients. Additionally, without optimal assessment and mapping of ecosystem services the sustainable benefits of ecosystem services conservation are not achievable (Naidoo et al., 2008).

The nature and quantity of ecosystem services supply from a landscape varies with forest and other land cover types (Baral et al., 2014b; Burkhard et al., 2012; van Oudenhoven et al., 2012). The supplies of ecosystem services are also governed by vegetation and other environmental attributes (de Groot et al., 2010; de Groot et al., 2002; García-Nieto et al., 2013; Müller and Burkhard, 2012; Seppelt et al., 2012). Therefore, assessing ecosystem services for a forested landscape using vegetation attributes of different forest types is likely to be more consistent than using some proxies such as area. Yet, little research is available which uses vegetation attributes for ecosystem services assessment for forested ecosystems (Alamgir et al., 2014a; Seppelt et al., 2012). One of the main reasons for this may be attributed to the diversified data requirements necessary to assess ecosystem services using vegetation attributes.

It has been shown that the diversity and quantity of ecosystem services supply from tropical forests are higher than many other forest biomes such as temperate and boreal forests (Daily, 1997; Galicia and Zarco-Arista, 2014; Liu et al., 2015), and that the supply of ecosystem services is declining at a higher rate from tropical forests (Liu et al., 2015; Mutoko et al., 2015). Yet, it is unclear how ecosystem services supply varies in different forest types within a tropical forested landscape. After a comprehensive review of ecosystem services mapping, Martínez-Harms and Balvanera (2012) concluded that identification of the key areas of ecosystem services supply is necessary for development of appropriate future strategies to ensure a sustainable supply of ecosystem services.

The Wet Tropics of northeast Australia is a unique landscape dominated by wet tropical rainforests, sclerophyll forests, rehabilitated plantation forests, together with other forest types (Stork and Turton, 2008). The exceptional biodiversity values of the Wet Tropics forests are formally recognised both from Australian (Hilbert et al., 2001; Williams et al., 2003) and global studies (Bertzky et al., 2013; Le Saout et al., 2013), while ecosystem service values for these forest complexes are yet to be explored. Only a few efforts have been initiated by Australian Federal and State governments, collaborating with local natural resource management bodies, to include ecosystem services in the natural resource management planning of this region (Alamgir et al., 2014b; Pert et al., 2014). In this study, we attempted to conduct a comprehensive evaluation of five regulating and three provisioning ecosystem services supplied from dominant forest types in the Wet Tropics landscape - rainforests, sclerophyll forests, and rehabilitated plantation forests.

Our specific objectives were to: (i) assess the quantity of five regulating ecosystem services - global climate regulation, air quality regulation, erosion regulation, nutrient regulation, and cyclone protection; and three provisioning ecosystem services- habitat provision, energy provision and timber provision; (ii) evaluate the variation of supply of those regulating and provisioning ecosystem services across environmental gradients, such as rainfall, temperature, and elevation; (iii) show the relationships among those ecosystem services; and (iv) identify the hotspots of single and multiple ecosystem services supply.

2.2. Materials and methods

2.2.1. The study area

Our study was conducted in the Wet Tropics bioregion (hereafter the region) of northeast, Australia (Fig. 2.1). The region is one of 89 bioregions in Australia, with each bioregion having unique climate, geology, landform patterns, ecological features and biological communities (Department of Environment, 2015). The region is more than two million hectares in area (Stork et al., 2011) with a contrasting landscape of various forest types where rainforests and sclerophyll forests are dominant (Fig. 2.1). The region enjoys a seasonally wet tropical climate (Hilbert et al., 2001; Williams et al., 2003) with a diverse range of environmental gradients- elevation from few metres above mean sea level (msl) to more than 1000m; total annual rainfall from less than 1000mm to more than 3000mm (up to 8000mm at the mountain peaks) (Ostendorf et al., 2001) ; and mean annual temperature ranges from 24^{o}C to 26^{o}C (16 to 20^{o}C in the mountains) (Ostendorf et al., 2001).

Due to the high ecological and world heritage values, ~45% of the region (mainly tropical rainforest) has been declared as a World Heritage Area since 1988 (Stork et al., 2011), and is now considered the sixth most irreplaceable natural habitat on the planet (Le Saout et al., 2013).

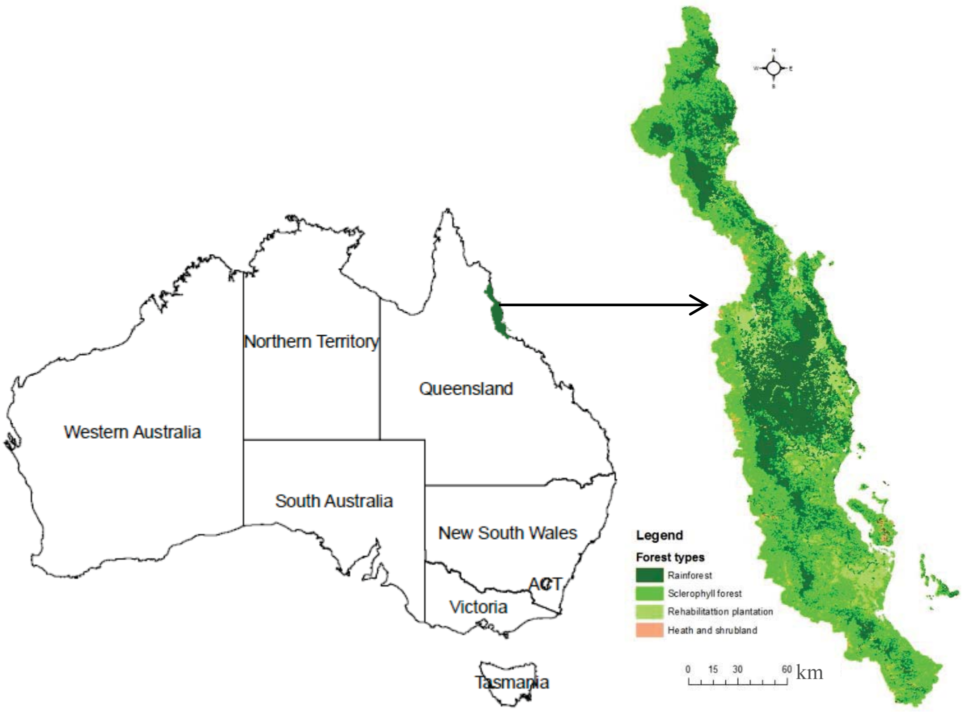


Figure 2.1. The study area – Wet Tropics bioregion, Queensland Australia with studied forest types.

2.2.2. Typology and framework of ecosystem service assessment

The Millennium Ecosystem Assessment (MA) has provided a detailed and comprehensive typology and framework for the assessment of ecosystem services, and has subsequently been widely used (Baral et al., 2013a; Burkhard et al., 2012; Harrison et al., 2014; Schneiders et al., 2012). Therefore, we utilized the typology of MA (MA, 2005) for selected ecosystem services (Table 2.1).

Table 2.1. Ecosystem service components, attributes and indicators used for the assessment.

Ecosystem service component	Attribute	Indicator
Regulating ecosystem services [a]		
Global climate regulation	Ecosystem plays an important role in global climate regulation by sequestering greenhouse gases	Sequestered atmospheric CO_2 by above ground tree biomass (CO_2 equ. Mg ha^{-1})
Air quality regulation	The capacity of ecosystem to remove toxic and other elements from the atmosphere	Tree canopy cover (%)
Erosion regulation	Vegetative cover plays an important role in soil retention	Stratified vegetation cover index
Nutrient regulation	The capacity of ecosystems to carry out (re) cycling of N, P or other nutrients	Nitrogen regulation (N kg ha^{-1})
Cyclone protection [b]	The presence of forest ecosystems can dramatically reduce the damage caused by cyclones	Coefficient of variance (CV) of tree diameter at breast height (dbh)
Provisioning ecosystem services [c]		
Habitat provision	Importance of ecosystem to provide habitat for species (particularly fauna) and natural biodiversity.	Multi-criteria index
Energy provision	Presence of trees or plants with potential use as energy source	Above ground tree biomass (AGB) (Mg ha^{-1})
Timber provision	Presence of species with potential use for timber	Tree basal area (BA) (m^2 ha^{-1})

Based on de Groot et al. (2002, 2010); MA (2005); and Burkhard et al. (2012). [a] Benefits received from the regulation of ecosystem processes (MA, 2005). [b] We use term cyclone protection rather than storm protection (which was used in MA, 2005) considering local preferences. [c] Products obtained from ecosystems (MA, 2005)

2.2.3. Sampling and data collection

In the Wet Tropics region we sampled a total of 66 sites (each plot an area of 0.05ha (50m ×10m). In the rainforests type we sampled a total of 24 sites (i.e. 13 sites found in mesophyll forests, nine in notophyll forests, and two in disturbed rainforests). In the sclerophyll forest type we sampled a total of 34 sites (32 from sclerophyll forests and woodlands, and two from the disturbed sclerophyll forests) and in the rehabilitation plantation forest type (i.e. forests aged between 10-19 years) six sites. In the heath and shrubland forest type we sampled two sites (detailed forest description in supplementary material). The sampled sites were located on a map prior to entering the field to avoid creeks and other water bodies. To avoid edge effects, we maintained at least a 50m distance to our plots from roads, water bodies, and agricultural lands. As the region has a diverse environmental gradient, our plots were distributed from 12m to more than 1000m above msl; from less than 1000mm to more than 3500mm annual rainfall; and less than 20^{o}C to more than 25^{o}C mean annual temperature ranges.

We used a modified transect method for collection of tree data within the 0.05ha (50m × 10m) plots (Fig. 2.2) (Burrows et al., 2002; Preece et al., 2012). This modified transect method has been shown to be suitable for estimating high tree densities in rainforests (Preece et al., 2012) and also for sclerophyll forests (Burrows et al., 2002). The diameter at breast height (dbh) of trees was measured to the nearest mm using a dbh tape; height of the representative trees (2-3 trees from each subplot) was measured to the nearest cm using Forestry Pro Laser Range Finder Hypsometer; and canopy cover was measured using convex spherical crown densitometer.

Large trees (≥ 20cm dbh) were measured in the 0.05 ha plots, medium trees (≥ 10 cm dbh) and small trees (≥ 2.5cm) were measured in the 0.03 and 0.015 ha subplots respectively (Fig. 2.2). In this way, we measured a total of 5084 trees and sampled an area of 33,000 m^2. Along the centre line of transects we counted the number of fine and coarse woody debris pieces that intersected the centre line less than 1m in height. We also measured ground cover (%) from 1m high at three 1m^2 subplots placed at 5m, 25m and 45m. In these subplots, we also measured canopy cover. Each plot was spatially referenced and physical attributes such as slope, soil, and elevation were also recorded. Spatial vegetation datasets were obtained from the Wet Tropics Management Authority, Queensland, Australia, and downscaled climate data sets from WorldClim (www.worldclim.org) using an ensemble of CMIP5 models.

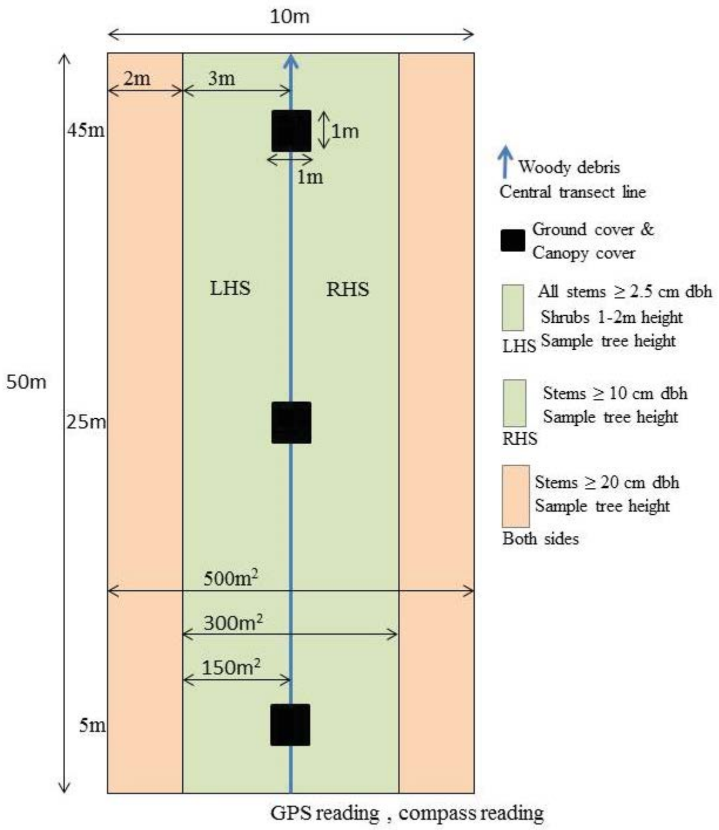


Figure 2.2. Plot layout showing the size of the subplots with the measured vegetation attributes.

2.2.4. Assessment of ecosystem services

2.2.4.1. Global climate regulation

We estimated the sequestered CO_2 equivalent (CO_2 equ Mg ha^{-1}) by above ground tree biomass (AGB) as an indicator of global climate regulation ecosystem service (Table 2.1) using the following equation:

CO_2 equ. = AGB×0.47×3.67

AGB was estimated using Chave's allometric equation (Chave et al., 2005):

$AGB = \rho * \exp(-1.499 + 2.148\ln(dbh) + 0.207(\ln(dbh))^2 - 0.0281(\ln(dbh))^3)$

where AGB is measured in kg, dbh is measured in cm, and ρ is wood density measured in g cm^{-3}. For wood density we used the reported default value for Australian tropical forests 0.5 g cm^{-3} (500 kgm^{-3}) (Department of Climate Change and Energy Efficiency, 2010). To convert AGB into biomass carbon storage we used 0.47, and to convert carbon storage into the CO_2 equ, we used 3.67, both recommended values from the Intergovernmental Panel for Climate Change for tropical forests (IPCC, 2006). Chave's formula (Chave et al., 2005) was derived based on the directly harvested 2410 trees of ≥ 5 cm dbh from 27 sites across the globe in tropical forest area, and provide the reliable and better estimate for AGB in tropical forest of this region (Preece et al., 2012).

2.2.4.2. Air quality regulation

The capacity of a forest to regulate air quality mostly depends on the canopy cover of the forests because tree leaf surfaces are the main structural elements which are used to regulate toxic elements from the atmosphere (Mincey et al., 2013; Nowak et al., 2014), so we estimated canopy cover (%) as an indicator of air quality regulation ecosystem service (Table 2.1). We estimated canopy cover following Lemmon (1956), after modified gap fraction from Bellow and Nair (2003). They used a class of zero with 5-15% openness and divided into six gap fraction classes (0 to 5) with 10 to 15% intervals, whereas we used four classes (1- 4) with equal 25% intervals: < 25% openness (gap fraction class 1); 25 -50% openness (gap fraction class 2); 50 -75% (gap fraction class 3) and >75% openness (gap fraction class 4). Therefore our estimate is very conservative.

2.2.4.3. Erosion regulation

Zhongming et al. (2010) developed a *Stratified vegetation cover index* accommodating the contribution of each layer of forest in soil conservation. We used this *Stratified vegetation cover index* (Zhongming et al., 2010) to estimate erosion regulation ecosystem service.

Stratified vegetation cover index (Cs) (Zhongming et al., 2010) is defined as:

$$C_s = \sum_{i=1}^{i} a_i C_i$$

where a_i is the weighting co-efficient of layer i for its contribution to soil conservation; and i is the number of layers or the strata in the vegetation community; C_i is the measured coverage of the vertical layer i.

2.2.4.4. Nutrient regulation

Nitrogen (N) turnover in the forested ecosystem is used as an indicator for the assessment of nutrient regulation ecosystem service (Burkhard et al., 2012; de Groot et al., 2010) and woody debris is an important source of N found in forested ecosystems (Clark et al., 2002). Therefore, we estimated N turnover in the forest ecosystem through woody debris (Clark et al., 2002) as an indicator of the nutrient regulation ecosystem service. We estimated the volume of woody debris (V) using the line intercept method (Iwashita et al., 2013; van Wagner, 1968), N content in the woody debris as a product of wood volume, density, and N content (%) (Clark et al., 2002; Iwashita et al., 2013) as follows:

$$V = \pi^2/8L \sum_{i=1}^{i} n_i d_i^2$$

$$N = V \times 0.422 \times 0.003$$

where V is the volume ($m^3\ ha^{-1}$), L is the transect length (m), n_i is the number of logs in the i^{th} diameter class, and d_i is the notional diameter (cm) of the i^{th} size class (= lower limit of class+1/3 the class range, to account for the right skewed distribution of log sizes), and N is the Nitrogen content in the fallen woody debris in Mg ha^{-1}. We used reported mean wood density (0.422 g cm^{-3}) of fallen woody debris from tropical forests (Clark et al., 2002).

2.2.4.5. Cyclone protection

The complex forests are more resistant to tropical cyclones and that unevenness of the forest structure is one of the indicators of the forest complexity (Chapman et al., 2008; Foster and Boose, 1992; Lugo, 2008; Turton, 2008, 2012; Xi, 2015). Therefore, we calculated the coefficient of variation (CV) of tree dbh as a proxy to assess the cyclone protection ecosystem service.

2.2.4.6. Habitat provision

Assessing the capacity of habitat provision ecosystem service using one indicator is very difficult due to the variation in habitat requirements for different wildlife (McElhinny et al., 2005; Pasher and King, 2011). A habitat complexity scoring method to evaluate the capacity of forested ecosystems to provide habitat for birds

(Watson et al., 2001) and another scoring system for small mammals (Barnett et al., 1978) has been developed based on few attributes of forests such as canopy cover, shrub cover, ground cover, and logs and litter cover on the forest floor. For the assessment of habitat provision, it is also worth taking into account other attributes of forests such as the CV of dbh, recruitments, and dead standing trees (Pasher and King, 2011). More recently Ochoa-Gaona et al. (2010) developed a multi criteria index to evaluate tropical forest condition, and Pasher and King (2011) developed another forest structural complexity index. Both indices accounted for a wide range of forest attributes. As a holistic approach to evaluate the capacity of the forest of the region to supply habitat provision ecosystem services we used a modified version of a multi criteria index (Ochoa-Gaona et al., 2010) based on relevant indices (Barnett et al., 1978; Ochoa-Gaona et al., 2010; Parkes et al., 2003; Pasher and King, 2011; Watson et al., 2001) so as to take into account the local Wet Tropics habitat conditions (an example of a fact sheet is presented Table 2.2.).

Table 2.2. Calculation of the multi-criteria index to assess habitat provision ecosystem service of tropical forest.

Habitat gradient	Ecological attribute	Score					Sub-index score	Total score
		Values of 5 levels						
		0	0.25	0.5	0.75	1		
Canopy (C)	Tree height (m)	< 5	5-10	10-15	15-20	≥ 20		
	Canopy cover (%)	< 20	20-40	40-60	60-80	≥ 80	∑C	
		Values of 4 levels						
		0	0.33	0.66	1			
Intermediate (I)	No. of large trees [a]	0-4	5-8	9-12	≥ 13			
	Density (trees per 0.05ha)	≤ 49	50-81	82-113	≥ 114			
	Mean dbh (cm)	6-11	11-15	15-20	≥ 20			
	CV [b] of dbh	0.39-0.68	0.69-0.98	0.99-1.28	≥ 1.29			
	CV of snag dbh	< 0.30	0.31-0.60	0.61-0.90	≥ 0.91			
	No. of snags (per 0.05 ha)	0-4	5-8	9-12	13-16			**∑CIF**
	No. of seedlings /recruitments [c]	≤ 99	100-199	200-299	≥ 300		∑I	
		Values of 3 levels						
		0	0.5	1				
Forest floor (F)	Seedlings cover (%) (1m high)	<12	12-25	≥ 25				
	Litter cover (%)	< 30	30-60	≥ 60				
	Coarse woody debris cover (%)	< 9	9-18	≥ 18				
	Fine woody debris (no.) [d]	0-44	45-85	≥ 86				
	Coarse woody debris (no.) [d]	0-4	5-8	≥ 9			∑F	

[a] Trees with > 40cm dbh (per 0.05 ha), [b] CV- Co-efficient of variance, [c] no. per 0.015ha (1-2m high), [d] Fine woody debris (< 10cm diameter) and coarse woody debris (≥ 10cm diameter) were counted along the 50m transect line.

2.2.4.7. Energy provision and timber provision

We estimated AGB following Chave et al. (2005) as an indicator of energy provision ecosystem service (section 2.2.4.1. for calculation details) while tree basal area (BA) as an indicator for timber provision. BA was estimated as $BA = \pi (dbh)^2/4$, where dbh stands for diameter at breast height.

2.2.5. Data analyses

Data was analyzed using IBM SPSS 20 statistical software and ESRI ArcGIS 10.2. After estimating all ecosystem services at the plot level, we exported all the data into SPSS 20 for statistical analysis, and into ArcGIS 10.2 for spatial analysis. Using ArcGIS 10.2, we constructed a look-up table, and then extracted rainfall and temperature data for each plot from the respective WorldClim spatial layers (www.worldclim.org). A Kruskal-Wallis test ($\alpha = 0.05$) was performed to examine the significance difference of each ecosystem service value for the different forest types. We used pairwise correlation (Spearman rank correlation) to evaluate the relationships among different ecosystem service values separately for rainforests and sclerophyll forests. In ArcGIS 10.2, we used spatial statistics - hotspots analysis (mapping clusters) - to identify the statistically significant hotspots at three different confidence intervals (90%, 95% and 99%) for single ecosystem service, and a grouping analysis (mapping clusters) to classify eight ecosystem service layers into three groups.

2.3. Results

2.3.1. Regulating ecosystem services

The supply of global climate regulation ($\chi^2 = 27.92$, df = 2, $P_{2\text{-tailed}} < 0.001$), air quality regulation ($\chi^2 = 29.49$, df = 2, $P_{2\text{-tailed}} < 0.001$), nutrient regulation ($\chi^2 = 15.403$, df = 2, $P_{2\text{-tailed}} < 0.001$) and cyclone protection ($\chi^2 = 13.413$, df = 2, $P_{2\text{-tailed}} = 0.001$) ecosystem services were found to be significantly different across the forests types; while the erosion regulation service ($\chi^2 = 4.72$, df = 2, $P_{2\text{-tailed}} = 0.094$) was not significantly different. In the pairwise comparison, it was found that the supply of global climate regulation, air quality regulation, nutrient regulation and cyclone protection ecosystem services were significantly higher in rainforests than sclerophyll forests (Kruskal-Wallis test, $P_{2\text{-tailed}} < 0.001$, $P_{2\text{-tailed}} < 0.001$, $P_{2\text{-tailed}} = 0.003$, $P_{2\text{-tailed}} < 0.001$, respectively), while differences between rainforests and rehabilitated plantation forests were not significant. Furthermore, nutrient regulation ecosystem service emerged as being significantly higher (Kruskal-Wallis test $P_{2\text{-tailed}} = 0.005$) in rainforests than in rehabilitated plantation forests. In the pairwise comparison, none of the five evaluated regulating ecosystem services were emerged significantly different between sclerophyll forests and rehabilitated plantation forests (Fig. 2.3).

The distribution of regulating ecosystem service values along the environmental gradients- rainfall, temperature and elevation, and their trend is illustrated in Fig. 2.4. The global climate regulation value in rainforests was shown to decrease with higher rainfall and temperature, but increased for the higher elevation gradient; whereas the opposite trend was observed in the sclerophyll forests. Our study results depicted no influence of annual rainfall, temperature and elevation on the air quality regulation value in rainforests of this region, while values in sclerophyll forest were higher for more rainfall and temperature; whilst values were lower along the higher elevation gradients. A similar increasing trend of erosion regulation and nutrient regulation values was observed for more rainfall, and temperature gradients; and a decreasing trend for higher elevations was evident in rainforests, sclerophyll forests and rehabilitated plantation forests. Cyclone protection values increased in rainforests along the higher rainfall and lower temperature gradients; increased in sclerophyll forests with more rainfall and higher temperatures; and decreased for higher elevations.

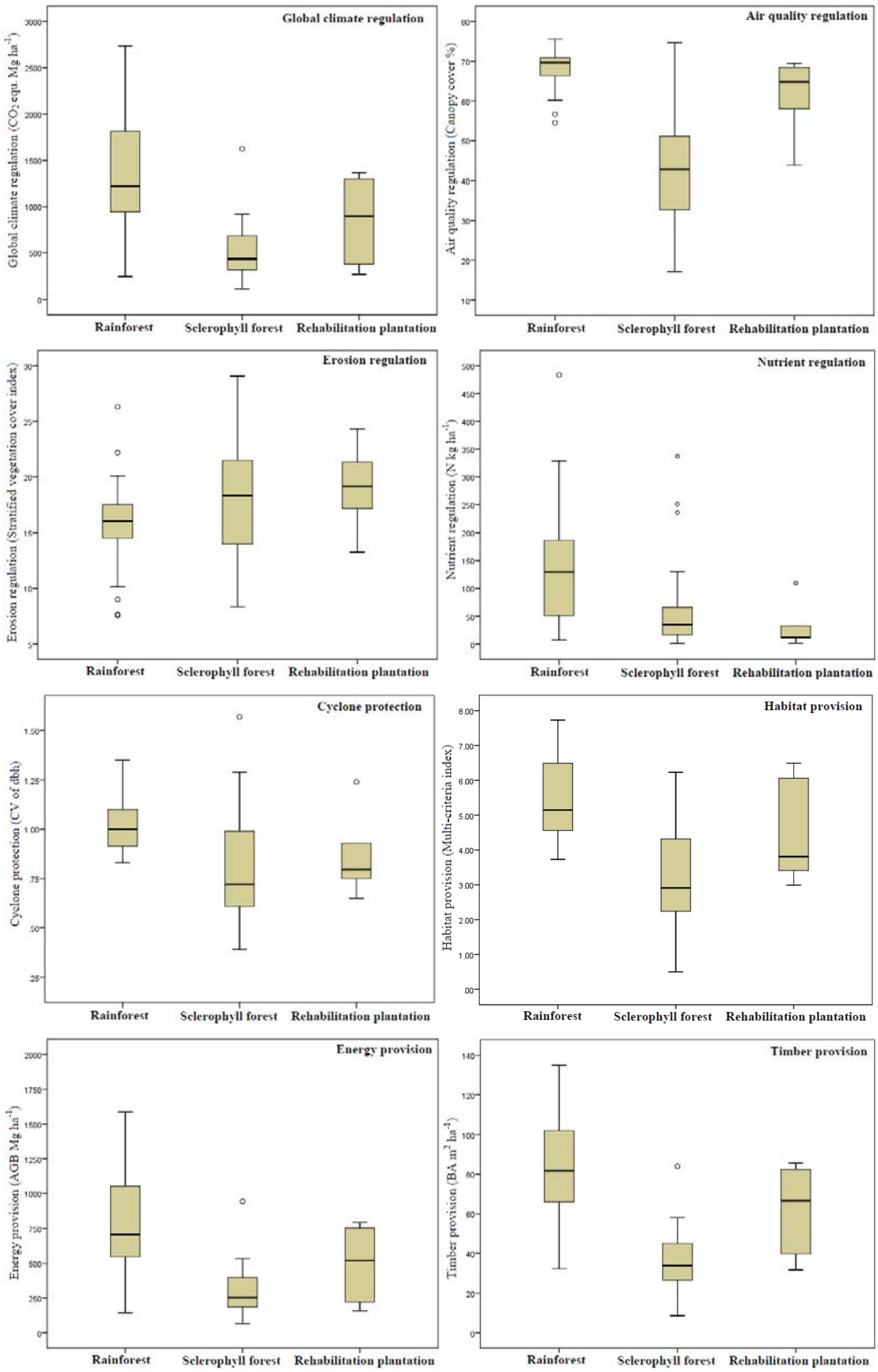
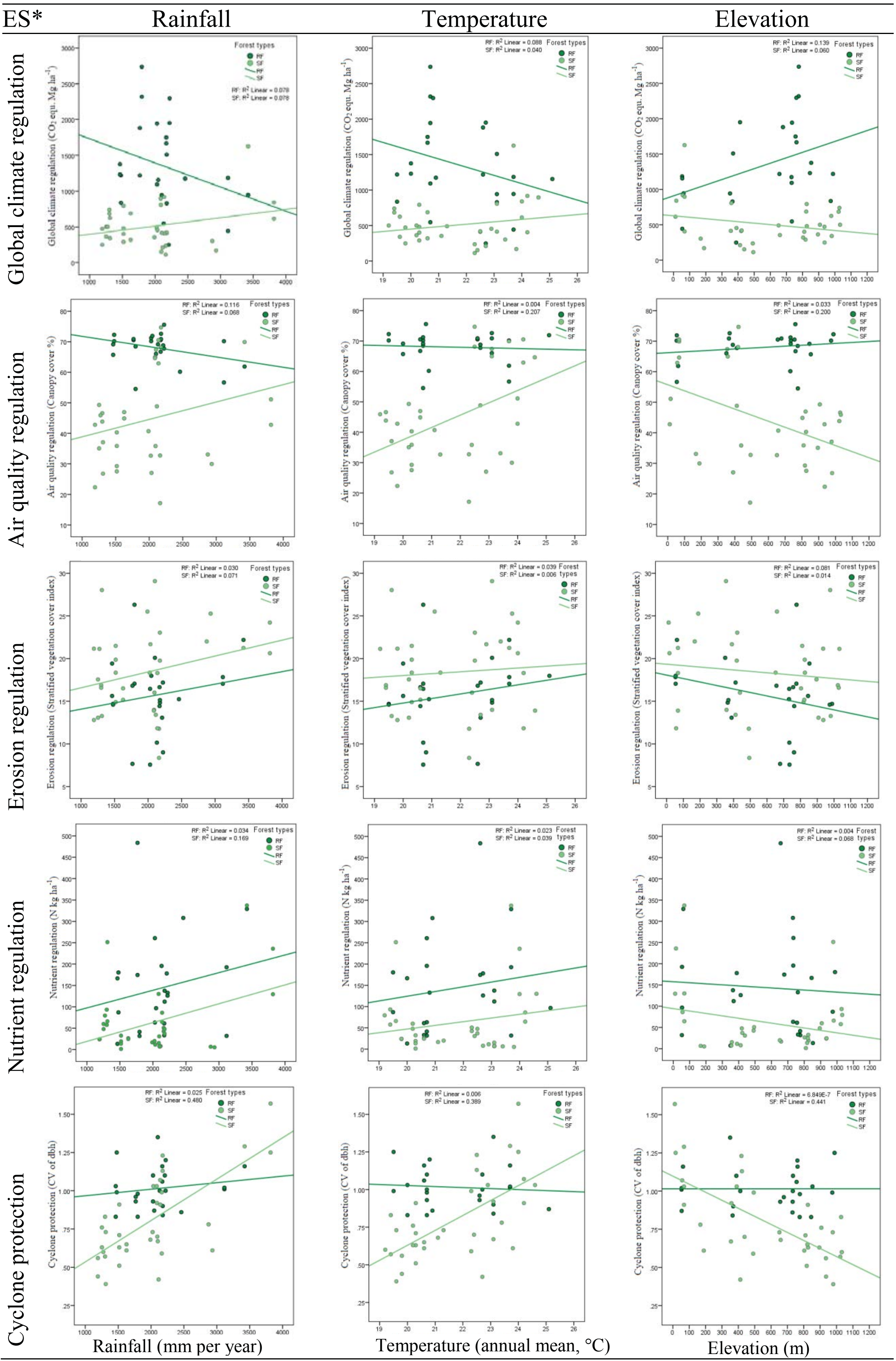


Figure 2.3. Box and whisker diagram showing the relationship between forest types, and regulating and provisioning ecosystem service values. Middle line = median; upper edge = 75th percentile; lower edge = 25th percentile; whisker caps = variability outside the quartiles; circles = outliers.



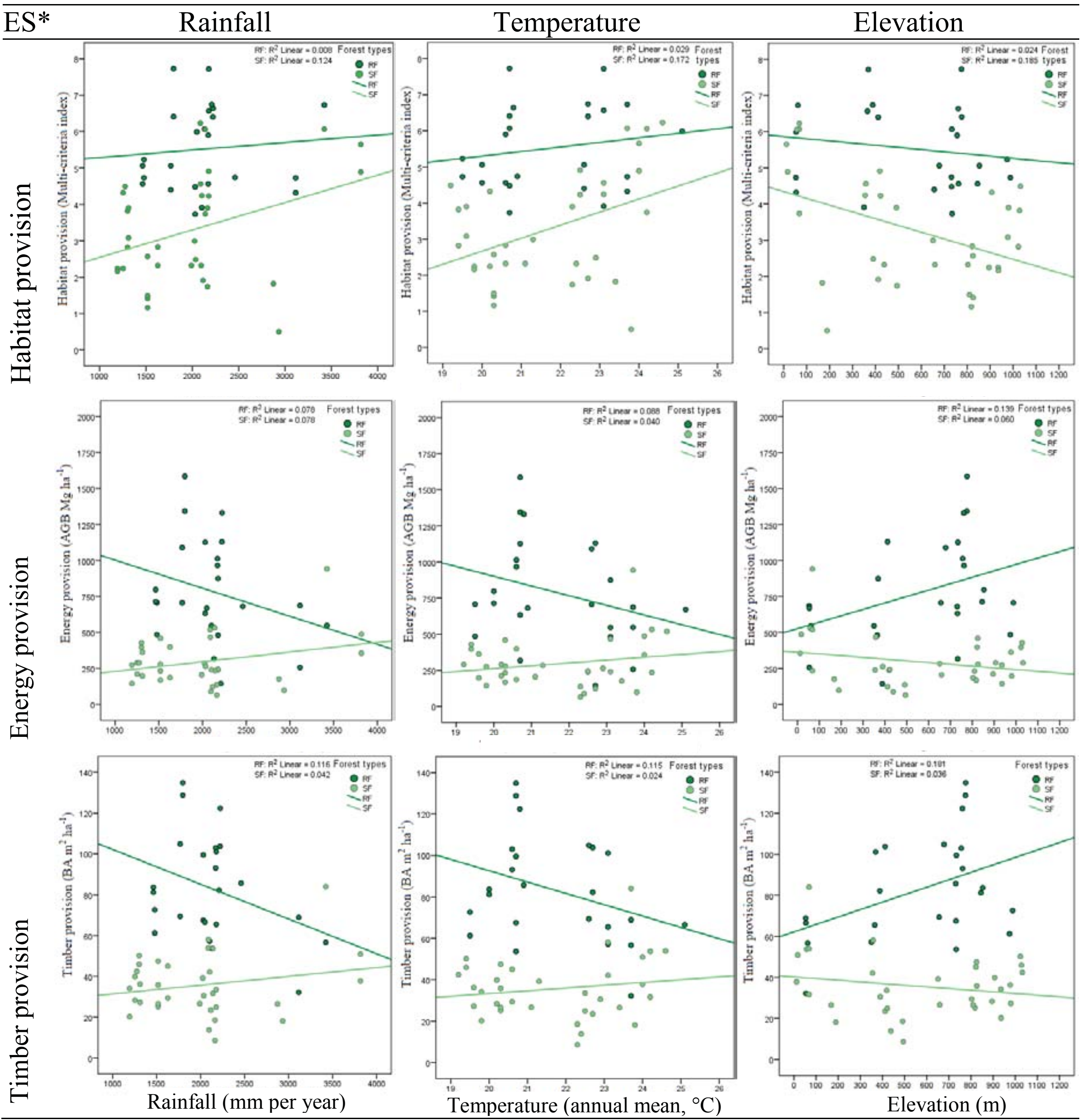
*ES- Ecosystem service; in figure RF- Rainforest, SF-Sclerophyll forest

Figure 2.4. The relation between regulating ecosystem service values with rainfall, temperature and elevation, the solid line shows the trend in respective forest type.

2.3.2. Provisioning ecosystem services

Our results showed (Fig. 2.3) that the supply of habitat provision ($\chi^2 = 24.92$, df = 2, $P_{2\text{-}tailed} < 0.001$), energy provision ($\chi^2 = 27.91$, df = 2, $P_{2\text{-}tailed} < 0.001$) and timber provision ($\chi^2 = 36.54$, df = 2, $P_{2\text{-}tailed} < 0.001$) ecosystem services were significantly different across the forest types. In pairwise comparison, habitat provision, energy provision and timber provision services were found to be significantly higher (Kruskal-Wallis test $P_{2\text{-}tailed} < 0.001$ in each case) in rainforests than sclerophyll forests; however no significant difference was observed between rainforests and rehabilitated plantation forests. The timber provision ecosystem service emerged as being significantly higher (Kruskal-Wallis test $P_{2\text{-}tailed} = 0.049$) in rehabilitated plantation forests than sclerophyll forests; while no significant difference was observed in case of habitat provision and energy provision ecosystem services.

The distribution and trend of provisioning ecosystem services with rainfall, temperature and elevation are shown in Fig. 2.5. The habitat provision values (multi criteria index) of both rainforests and sclerophyll forests were found to be lower in low rainfall and temperature zones but higher in low elevation zones. The energy provision values (AGB Mg ha^{-1}) in rainforests were higher in low rainfall and temperature zones and higher in high elevation zones. The opposite trend was found for sclerophyll forests. Timber provision values (BA m^2 ha^{-1}) in sclerophyll forests increased with increasing rainfall and temperature values and decreased with elevation. The trend in timber provision values with rainfall, temperature and elevation gradients in rainforests was found to be opposite to that of sclerophyll forests.



*ES- Ecosystem service; in figure RF- Rainforest, SF-Sclerophyll forest

Figure 2.5. The relation between provisioning ecosystem service values with rainfall, temperature and elevation, the solid line shows the trend in each respective forest type.

2.3.3. Correlation between ecosystem services

In the rainforest, the global climate regulation values were significantly positively correlated with the energy provision and timber provision values (Spearman rank correlation, $p < 0.01$) while air quality regulation was significantly negatively correlated ($p < 0.05$) with erosion regulation values. The erosion regulation ecosystems service values were also significantly negatively correlated ($p < 0.05$) with nutrient regulation values (Table 2.3). In the sclerophyll forest, global climate regulation values were significantly positively correlated ($p < 0.01$) with air quality regulation, nutrient regulation, cyclone protection, habitat provision, energy provision and timber provision values (Table 2.4).

Table 2.3. Pairwise correlation co-efficient (Spearman rank correlation) between ecosystem service values in the rainforest.

	Global climate regulation	Air quality regulation	Erosion regulation	Nutrient regulation	Cyclone protection	Habitat provision	Energy provision	Timber provision
Global climate regulation	1	0.085	0.014	-0.285	-0.042	0.145	**1.000****	**.897****
Air quality regulation		1	**-.487***	0.095	-0.105	0.108	0.085	0.084
Erosion regulation			1	**-.411***	-0.08	0.035	0.014	-0.143
Nutrient regulation				1	-0.058	-0.058	-0.285	-0.245
Cyclone protection					1	-0.199	-0.042	-0.102
Habitat provision						1	0.145	0.323
Energy provision							1	**.897****
Timber provision								1

Significant values are in bold; ** $p < 0.01$, * $p < 0.05$

Table 2.4. Pairwise correlation co-efficient (Spearman rank correlation) between ecosystem service values in the sclerophyll forest.

	Global climate regulation	Air quality regulation	Erosion regulation	Nutrient regulation	Cyclone protection	Habitat provision	Energy provision	Timber provision
Global climate regulation	1	**.533****	0.124	**.530****	**.522****	**.680****	**1.000****	**.954****
Air quality regulation		1	0.131	**.344***	**.511****	**.642****	**.533****	**.559****
Erosion regulation			1	-0.179	0.244	-0.024	0.124	0.071
Nutrient regulation				1	**.401***	**.681****	**.530****	**.474****
Cyclone protection					1	**.570****	**.522****	**.413***
Habitat provision						1	**.680****	**.679****
Energy provision							1	**.954****
Timber provision								1

Significant values are in bold; ** $p < 0.01$, * $p < 0.05$

2.3.4. Hotspots for ecosystem services

We were able to spatially identify the distribution of hotspots (supply of significantly higher ecosystem services) and coldspots (supply of significantly lower ecosystem services) of each ecosystem service for three different confidence intervals- 99%, 95% and 90% (Fig. 2.6 and Table 2.5). Hotspots for most of the ecosystem services supply were found mainly within the rainforest plots; while coldspots for most of the ecosystem services supply were found within the sclerophyll forest plots. Furthermore, in the rainforests plots most of the hotspots were located in the upland rainforest areas (Table 2.5).

In the grouping analysis (Fig. 2.7), we were able to identify the sites which provide the highest values of the multiple ecosystem services (group 3, green colour), the sites which provide medium values of the multiple ecosystem services (group 1, blue colour), and the sites which provide the lowest values of the multiple ecosystem services (group 2, red colour). In the green group, the mean value of global climate regulation was 1825 (CO_2 equ Mg ha^{-1}), air quality was 68 (canopy cover %), erosion regulation (stratified vegetation cover index) was 16, nutrient regulation was 105 (N kg ha^{-1}), cyclone protection was 1 (CV of dbh), habitat provision was 6 (multi-criteria index), energy provision was 1058 (AGB Mg ha^{-1}) and timber provision was 102 (BA m^2 ha^{-1}). Whereas in the blue group the mean values were 843, 65, 17, 144, 1, 5, 489 and 58; whilst in the red group the values were 414, 40, 19, 39, 0.69, 3, 240 and 32 respectively.

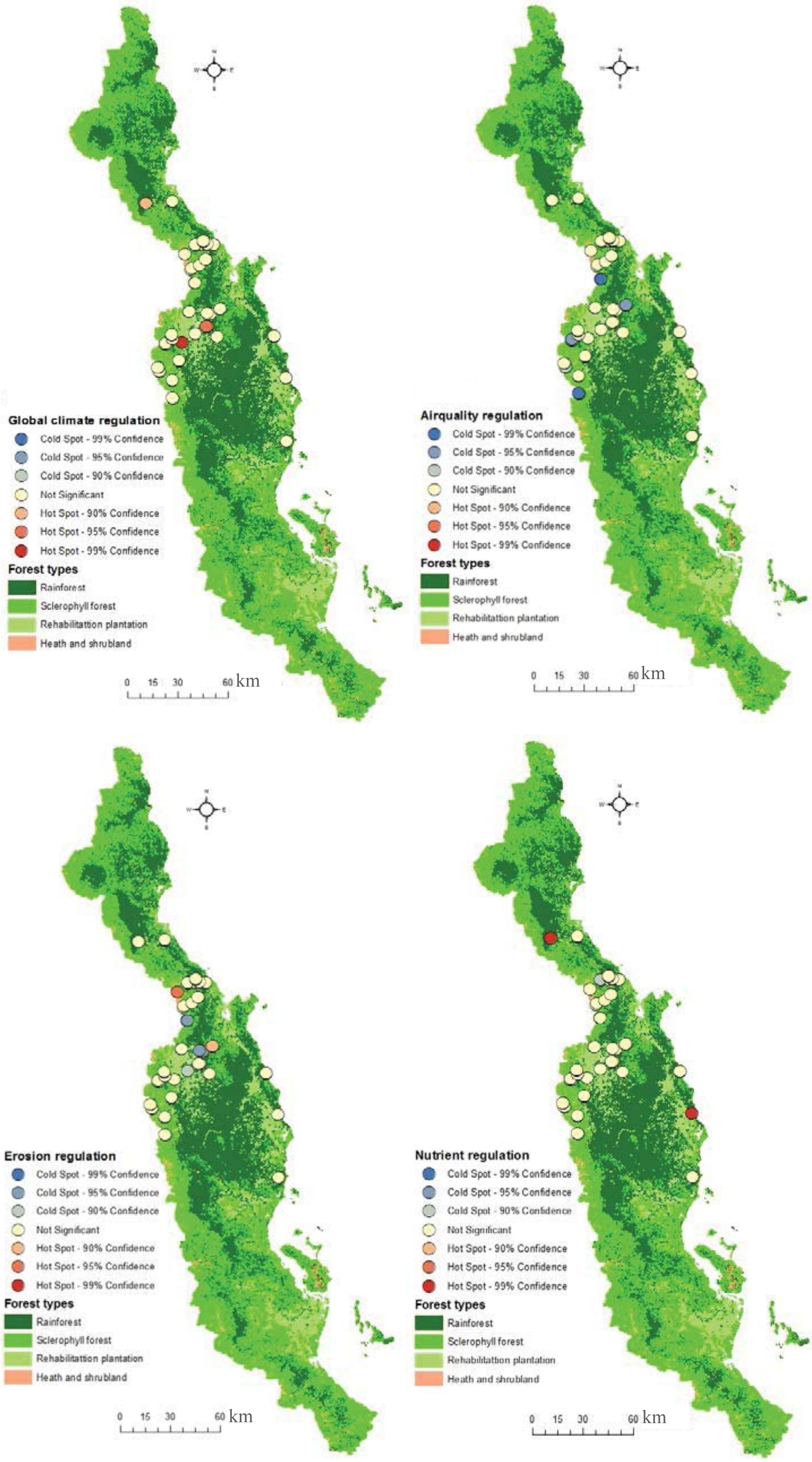


Figure 2.6. The hotspots and coldspots of regulating and provisioning ecosystem service values in the Wet Tropics, Australia. Clockwise from top: global climate regulation, air quality regulation, erosion regulation, and nutrient regulation;

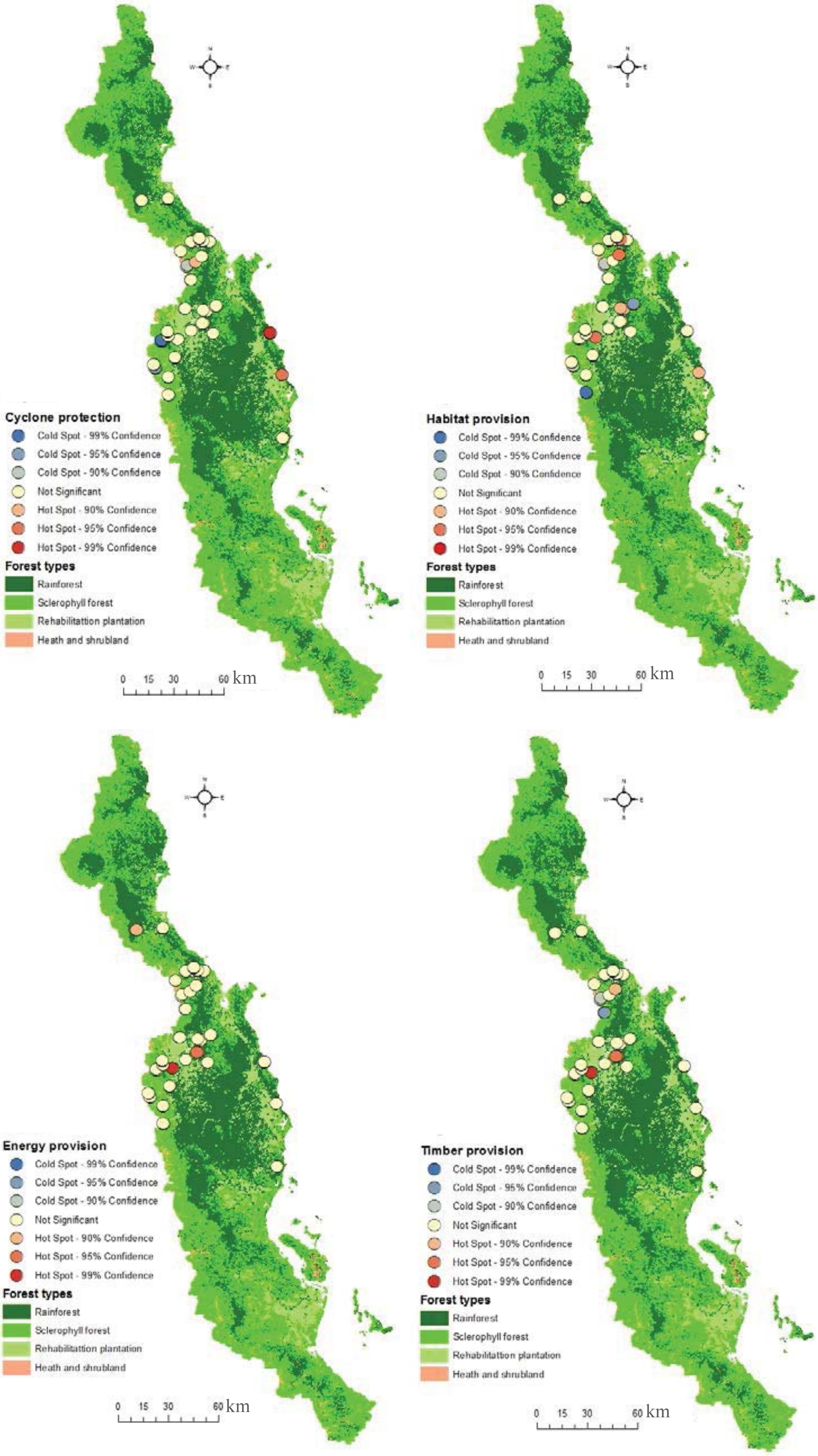


Figure 2.6. Continued- clockwise from top: cyclone protection, habitat provision, energy provision, and timber provision.

Table 2.5. Hotspots and coldspots of regulating and provisioning ecosystem service values in the Wet Tropics bioregion, g, global climate regulation; a, air quality regulation; e, erosion regulation; n, nutrient regulation; c, cyclone protection; h, habitat provision; en, energy provision; and t, timber provision.

Plot site	Forest types	Elevation (m)	Hotspot (Confidence Interval- CI)			Cold spot (CI)		
			99%	95%	90%	99%	95%	90%
Wongabel SF [a]	Rainforest	776	g, en, t	h				
Crater Lakes NP	Rainforest	762		g, en, t				
Danbulla NP	Rainforest	761			h		e	
Curtain Fig NP	Rainforest	734						e
Mount Lewis NP	Rainforest	678	n		g, en			
Barron Gorge NP	Rainforest	412		h	t			
Kuranda NP	Rainforest	365		h	a			
Moresby Range NP	Rainforest	62	n	c	h			
Rehabilitated forest	Rehabilitated forest	364						n
Bilwon SF	Heath and shrubland	388		e				
Herberton Range SF	Sclerophyll forest	1032				c		
Herberton Range NP	Sclerophyll forest	980				c		
The Bluff FR	Sclerophyll forest	935					c	h
Millstream Falls NP	Sclerophyll forest	825				a, h		
Ravenshoe FR	Sclerophyll forest	818				a, h		
Davies Creek NP	Sclerophyll forest	491					a, e, t	
Dinden SF	Sclerophyll forest	438						h, c, t
Dinden NP	Sclerophyll forest	419			c			
Little Mulgrave NP	Sclerophyll forest	168			e		h	
Russell River NP	Sclerophyll forest	17	c					

[a] SF, State forest; NP, National park; FR, Forest reserve.

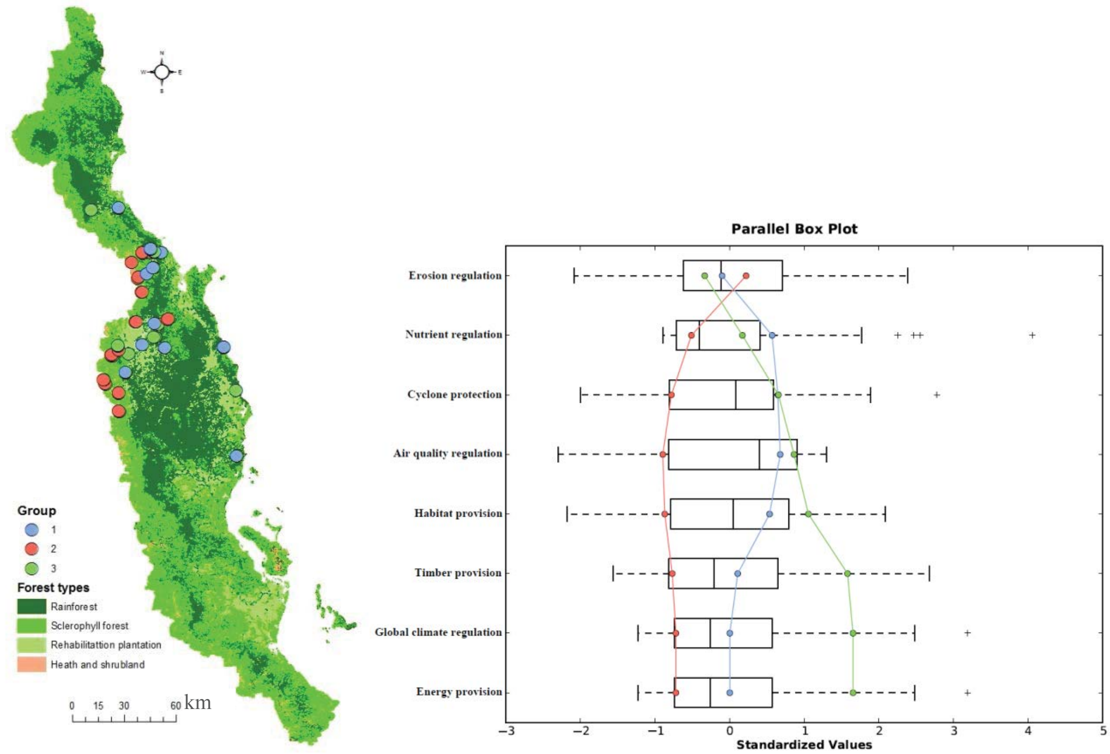


Figure 2.7. The three groups of hotspots of multiple regulating and provisioning ecosystem service values in the forested landscape of the Wet Tropics, Australia. Left panel: spatial distribution of hotspots of multiple ecosystem services; right panel: parallel box plots showing the values of each ecosystem services in each three groups.

2.4. Discussion

2.4.1. Forest types and ecosystem services supply

All regulating and provisioning ecosystem service values assessed (except for erosion regulation) were significantly higher (Kruskal-Wallis test, $\alpha = 0.05$) in the rainforests when compared to sclerophyll forests (Fig. 2.3). Our findings are consistent with Amazon rainforests (Tian et al., 2000), rainforests of Panama (Chave et al., 2003) and French Guiana (Chave et al., 2001), and several previous findings across the globe (Lewis et al., 2009; Liu et al., 2015; Malhi and Grace, 2000) where reported higher values of global climate regulation ecosystem service occur in the rainforest. Most of the rainforests in this region are highly dense with a nearly closed canopy (Liddell et al., 2007) which can be accountable for providing significantly higher values of global climate regulation, air quality regulation, energy provision and timber provision ecosystem services. Our erosion regulation ecosystem service estimation (section

2.2.4.3) estimates the contribution from each stratum of the forest with more weight applied to the near ground layer. The rainforests in this region are almost closed i.e. >70% canopy cover (section 2.3.1.) and have a lack of near ground layer (e.g. grasses); hence this reduces the erosion regulation ecosystem service values of the rainforests and increases the values found in sclerophyll forests. As we collected most of our sclerophyll forest data before the controlled burning season, there will be a strong influence of the near ground layer. Our study reveals significantly higher values (Kruskal-Wallis test, $\alpha = 0.05$) of habitat provision ecosystem service found in the rainforests when compared to the sclerophyll forest (Fig. 2.3). The findings are comparable with previous research (Hilbert et al., 2001; Laurance and Laurance, 1999; Laurance, 1994; Williams and Pearson, 1997) who reported higher biodiversity values of the rainforest in this region. Le Saout et al. (2013) reported that the rainforest of this area is the sixth most irreplaceable natural habitat on the planet considering birds, mammals, and amphibian species which certainly supports our findings, the very high capacity of the rainforest of this region to supply habitat provision ecosystem service.

The hotspots of one ecosystem service value may be the 'coldspots' of another ecosystem service value due to the lack of structural and functional elements in the forest to provide particular ecosystem service. In this study, hotspots for most of the key ecosystem services were identified in the rainforests; but most of them (hotspots of a particular ecosystem service) later on were not identified as coldspots for other ecosystem services (Fig. 2.6). This demonstrates a tremendous capacity of the rainforests to provide multiple ecosystem services. Contrastingly, the coldspots of the most of the ecosystem services were found to occur in the sclerophyll forests (section 2.3.4), which were subsequently not identified as hotspots for other ecosystem services. This indicates the lower capacity of the sclerophyll forests to provide key as well as multiple ecosystem services. This may be due to the lower stem density, open canopy, and more abundance of near ground vegetation (like grasses) in the sclerophyll forests.

The regulating and provisioning ecosystem service values in the rehabilitated plantation forests are noteworthy (Fig. 2.3), and indicate the potential capacity of rehabilitated plantation forests to provide regulating and provisioning ecosystem services. The studied rehabilitated plantation forests were established with native local rainforest species and were 10-19 years old. Furthermore, some of the rehabilitated plantation forests (~19 years old), which are similar to the nearby rainforest, were observed to have high ecosystem service values also. After studying the tropical rainforests of Sabah, Borneo Malaysia, Edwards et al. (2009) reported that rehabilitated forests can provide global climate regulation as well as habitat provision services similarly to those close to the adjacent rainforests. Our study results also reveal this same trend. Kanowski and Catterall (2010) reported that rehabilitated plantation forests in this region are more densely stocked and contain a significant number of large trees. This might be the reason for higher values of regulating and provisioning ecosystem services found in rehabilitated plantation forests studied. Edwards et al. (2014) found that degraded rainforests can retain their capacity for ecosystem functioning and provide ecosystem services. Our study results are consistent with their findings.

2.4.2. Ecosystem services supply along the environmental gradient

The regulating and provisioning ecosystem service values vary along the rainfall, temperature and elevation gradients within the same forest type and also between forest types (Fig. 2.4 and 2.5). The higher elevation and lower temperature rainforests (popularly called cloud forest) provide higher global climate regulation ecosystem service (Fig. 2.4). Reasons include: these forests are very close (e.g. Mount Hypipamee National Park, Atherton) consequently provide higher air quality regulation ecosystem service, and are less frequently disturbed than lowland rainforests from tropical cyclones (such as those forests in the Innisfail and Mission Beach area) (Turton, 2008, 2012) consequently provide low nutrient regulation ecosystem service and higher global climate regulation ecosystem service. The trend of global climate regulation ecosystem service values and air quality service values with elevation in the sclerophyll forest was found to be negative because sclerophyll forests in higher elevations are relatively more open (like the sclerophyll forests of Millstream Falls National Park, Ravenshoe). The rainforests in the lower elevation (such as in the Innisfail and Mission Beach region) have higher values for erosion regulation, and nutrient regulation due to the disturbances caused by tropical cyclones. Dense near ground vegetation and relatively high amounts of fallen woody debris in the forest floor were also observed. The findings are consistent with the findings from Puerto Rico forests (Beard et al., 2005) and Mississippi forests (Chapman et al., 2008). Our study reports lower values of habitat provision ecosystem services occurring in higher elevation rainforests and sclerophyll forests which differ from the general understanding that upland rainforests provide habitat for endemic species (Costion et al., 2015). Our habitat provision assessment using multiple attributes of forest focused on overall wildlife habitat requirements for different strata of the forest rather than the distribution pattern of wildlife, and did not particularly focus on the habitat of upland endemic species. Due to the frequent disturbances (such as tropical cyclones) and resulting regrowth in the low elevation rainforests, and sclerophyll forests, higher values of habitat provision ecosystem services can be observed there than forests found in higher elevations. The relatively frequent disturbance from tropical cyclones in the lowland rainforest in this region compared with upland rainforest, and resulting ecological restructuring reported by Turton (2008, 2012) and Laurance and Curran (2008) further strengthen our findings.

2.4.3. The relationships between ecosystem services

Different ecosystem service values may be positively or negatively correlated both in the rainforests and/or sclerophyll forests (Table 2.3 and 2.4). The negative correlations found for global climate regulation, nutrient regulation and cyclone protection services in the rainforests (Table 2.3) might be due to three ecological impacts caused by recent tropical cyclones: i) reduction of above ground biomass due to tree falls; ii) increase of woody debris on the forest floor; and iii) regeneration and recruitment after creation of canopy gaps. Iwashita et al. (2013) also reported a negative correlation with coarse woody debris and standing living biomass from tropical montane wet forests of Hawaii. A negative correlation between air quality regulation and erosion regulation services in the rainforest (Table 2.3) may contrast with our general preconceived understanding. This is due to more weight found in the near ground layer than found in the tree canopy layer in the stratified vegetation cover index (details in section 2.2.4.3 and discussed in section 2.4.1). Most of the ecosystem service values in the sclerophyll forests were significantly strongly correlated, while a wide variation existed in the rainforest (Table 2.3 and 2.4). This might be due to the higher structural and functional diversity found in rainforests than in sclerophyll forests. The ecosystem service values in the rainforests depend on multiple structural elements (such as diversified dbh class, and multiple vertical stratification), therefore any changes in one ecosystem service may have very few impacts on other ecosystem services. On the other hand, ecosystem service values in the sclerophyll forest depend on a few structural elements (such as a single vertical stratum, and limited dbh distribution), so that any changes in one ecosystem service value may have significant impact on many other ecosystem service values.

2.4.4. Upland rainforests are the hotspots of multiple ecosystem service supply

Most of the hotspots of multiple ecosystem service values were located in the upland rainforest (Fig. 2.7); while the hotspots for key regulating and provisioning ecosystem services varied across the landscape (Fig. 2.6, Table 2.5). Wallace and McJannet (2013) described a higher capacity of upland rainforests in their study area provided hydrological ecosystem services due to the complex canopy and forest structure. We argue that less frequent cyclone disturbances, and more ecological resilience against tropical cyclones, along with higher structural diversity and complex canopy might influence the upland rainforest being identified as hotspots of multiple ecosystem services supply. An example of this can be found in the Wongabel State Forest, which contained hotspots of global climate regulation, habitat provision, energy provision and timber provision. This State Forest is an upland rainforest (776m elevation) located in the Atherton area, which is very well stocked, and is less frequently disturbed by tropical cyclones (Turton, 2008, 2012); hence ecosystem service values are very high. Mount Lewis National Park (678m elevation) also an upland rainforest, was observed as a hotspot of global climate regulation, nutrient regulation and energy provision services and is conversely situated in the cyclone disturbance zone (Innisfail – Port Douglas) (Turton, 2008, 2012), and hence provides more nutrient regulation ecosystem services. The higher values of climate regulation and energy provision in this forest definitely indicate its ecological resilience against tropical cyclones. The higher values of multiple ecosystem services from montane cloud forests of Mexico (Martínez et al., 2009), the alpine forests of Italy (Häyhä et al., 2015) have been assessed in monetary values. Our study reveals the ecological values of multiple ecosystem services from upland rainforests are higher. The relatively lowland rainforests are hotspots for nutrient, and erosion regulation ecosystem services, due to the occurrence of tropical cyclones and more fallen wood hence more nutrient regulation, more recruitment, and near ground layers provides subsequently more cyclone protection and erosion regulation ecosystem services. The upland sclerophyll forests were shown to be the hotspots of erosion regulation ecosystem services; however other ecosystem service values were significantly lower like cyclone protection due to the evenness of structure, nutrient regulation due to the lack of disturbances, air quality regulation due to more openness of the canopy, global climate regulation due to the low density of trees, and habitat provision due to the lack of stratified structure of the forest.

2.5. Conclusions

Through large scale field data collection of vegetation attributes of rainforests, sclerophyll forests and rehabilitated plantation forests we estimated five regulating ecosystem services- global climate regulation, air quality regulation, erosion regulation, nutrient regulation, cyclone protection; and three provisioning ecosystem services- habitat provision, energy provision and timber provision. We found that, in general, rainforest provides higher ecosystem services, followed by rehabilitated plantation forests, and sclerophyll forests. The rainforests studied possess a very high capacity to supply multiple ecosystem services, whilst upland rainforests were observed to be the main hotspots of multiple ecosystem services supply. The rehabilitated forests may provide some ecosystem services nearly equivalent to those found in nearby rainforests. The ecosystem services supply from the sclerophyll forests was found to be lower, most likely due to their lower structural diversity. Therefore, we argue that active conservation of rainforests needs to occur to maintain ecosystem services supply from a contrasting forested landscape like the Wet Tropics bioregion with more emphasis placed on protecting upland rainforests. Rehabilitated forests are also worth protecting for ecosystem services supply and increasing structural diversity in the sclerophyll forests will also enhance ecosystem services supply. In case of limited resources and options for protection of rainforests, the more important ecosystem services for the community should be identified, and subsequently the hotspots of that ecosystem service supply should be identified and protected, as ecosystem services supply widely vary across forest types and environmental gradients.

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OVERVIEW OF CHAPTER 3

This chapter is based on a paper under review in *Land Use Policy* journal[5] with minimal formatting changes. The focus of this chapter is objective number three of the thesis - to evaluate the capacities of different Land Use and Land Covers (LULC) to supply ecosystem services, and to identify the spatial congruence between ecosystem services and biodiversity in the landscape.

I assess the capacities of different LULC in the Wet Tropics bioregion landscape, based on expert opinion. I arranged an expert workshop at James Cook University, Cairns campus, Australia in September 2015. Experts completed an ecosystem service and ecological integrity assessment matrix using relative scoring methods. I spatially analysed that matrix using ESRI ArcGIS10.2 to evaluate ecosystem services potential and biodiversity potential of this multifunctional landscape. The workshop was arranged following James Cook University, human research ethics guidelines. For this project James Cook University human research ethics approval number is H4877.

[5] **Alamgir, M.**, Turton, S.M., Macgregor, C.J., Pert, P.L. (in review). The capacity of a heterogeneous landscape to supply ecosystem services, and spatial congruence between ecosystem services and biodiversity. *Land Use Policy.*

CHAPTER 3

THE CAPACITY OF A HETEROGENEOUS LANDSCAPE TO SUPPLY ECOSYSTEM SERVICES, AND SPATIAL CONGRUENCE BETWEEN ECOSYSTEM SERVICES AND BIODIVERSITY

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The Wet Tropics -a unique heterogeneous forested landscape on the planet (Photo credit: Mohammed Alamgir)

ABSTRACT

Ecosystem services and biodiversity are the two most important, distinct but interlinked components in ecological processes and functioning of ecosystems. Unsustainable use of ecosystem services and biodiversity can threaten human well-being, hence efforts to conserve them is of global concern. Integration of both ecosystem services and biodiversity in conservation planning and assessment remains a challenge, as the interlinkages between the two are still unclear. We utilized expert opinion to assess ecosystem services and biodiversity (as an ecological integrity component) in a tropical heterogeneous forest dominated landscape. We found that key and multiple ecosystem services supply varies across the landscape and that forest disturbances reduce the capacity to supply key and high-potential multiple ecosystem services. Our study revealed a spatial congruence between high-potential biodiversity and high-potential global climate regulation ecosystem service in the intact rainforest areas, but a divergence in the sclerophyll and other disturbed and low tree abundance forested areas. Overall, our study showed a spatial congruence between biodiversity and multiple high-potential ecosystem services – mainly forest based ecosystem services. In addition to interior (relatively intact) conserved forests, we argue that management intervention priorities should focus on increasing tree abundance both in non-tree vegetated land cover areas and within disturbed forested areas to increase the high-potential multiple ecosystem services supply at the landscape level. The integration of high-potential multiple ecosystem services supply and biodiversity conservation is possible in the tropical forested landscape provided that the multiple ecosystem services are forest-based (e.g. global climate regulation, air quality regulation, and habitat provision). Therefore, we argue a careful selection of multiple ecosystem services is required to integrate high-potential multiple ecosystem services and high-potential biodiversity in conservation planning and assessment measures.

Keywords: ecological integrity; land cover; expert opinion; rainforest; sclerophyll forest

3.1. Introduction

In recent years, a widespread decline of several ecosystem services supply has been reported across the world (Häyhä et al., 2015; Martínez et al., 2009; Shaw et al., 2011) while an enormous contribution of ecosystem services to human well-being are well recognized and well documented (Costanza et al., 1997; Ninan and Inoue, 2013; Scolozzi et al., 2012). The scientific and effective management of such areas are critical that maintain ecosystem functioning and integrity and ecosystem components to supply key and multiple ecosystem services at the landscape (de Groot et al., 2010; Egoh et al., 2007). Yet, conservation of ecosystem services and management are rarely prioritized in the landscape management (Harrison et al., 2014), while priorities are geared more towards biodiversity conservation (Heller and Zavaleta, 2009).

Ecosystem services are the goods and services human communities receive from an ecosystem (MA, 2005). Ecosystem services and biodiversity are identified as the two most important, distinct but interlinked components of the ecosystems (de Groot et al., 2010; MA, 2005). Yet, the link is not well understood. Chan et al. (2006) reported a positive correlation between biodiversity and forest based ecosystem services (e.g. carbon storage and outdoor recreation). Harrison et al. (2014) ascertained that improvement of certain ecosystem service provision (e.g. landscape aesthetics) in most cases usually delivers biodiversity values. Egoh et al. (2009) determined that biodiversity conservation actions increase certain ecosystem services in the landscape but were weakly correlated. Therefore, spatially explicit mapping is required for the effective integration of ecosystem services and biodiversity into conservation planning and assessment (Chan et al., 2006). Spatially explicit mapping at the landscape level brings together spatial and temporal information indicating where management intervention should be focused for integrating ecosystem services and biodiversity (Baral et al., 2014b; García-Nieto et al., 2013; Schneiders et al., 2012; Schulp et al., 2012).

The problem is that the approach of making ecosystem service assessment spatially explicit and the integration of necessary ecological information into the ecosystem service assessment are both still unclear (Eigenbrod et al., 2010; Martínez-

Harms and Balvanera, 2012). One of the main reasons is that ecosystem services are the products of multiple interactions between the landscape components and ecological processes (van Oudenhoven et al., 2012), and the supply is largely determined by Land Use and Land Cover (LULC) (de Groot et al., 2010; Foley et al., 2005; Müller and Burkhard, 2012). Therefore, LULC maps with expert assessment have commonly been used to assess the relative capacity of different vegetation across a landscape to supply multiple ecosystem services and biodiversity components (Burkhard et al., 2009; Burkhard et al., 2012; Maes et al., 2011; Schneiders et al., 2012; Sohel et al., 2014).

Overall, ecosystem services assessment science in Australia is in its early stages of development (Alamgir et al., 2014a). Ecosystem service studies focusing on production landscapes in Australia revealed that urgent conservation priorities are required for current and future ecosystem services supply (Baral et al., 2013a; Baral et al., 2014b). Yet, the spatial distribution of ecosystem services, and the congruence of ecosystem services and biodiversity in forest-dominated landscape are still to be explored.

The Wet Tropics bioregion in northeast Australia is a unique tropical landscape with several contrasting vegetated and non-vegetated land covers, where rainforests and sclerophyll forests dominate (Stork and Turton, 2008). The biodiversity values of this landscape, both in the rainforests and sclerophyll forests, are widely explored (Costion et al., 2015; Hilbert et al., 2001; Williams et al., 2003), and the rainforested areas are globally recognized for their exceptionally high biodiversity value (Le Saout et al., 2013). Most of the management efforts and planning in this landscape have been directed towards biodiversity conservation, largely ignoring its capacity to supply multiple ecosystem services (WTMA, 2011). However more recently, limited efforts have been initiated by the Australian Federal Government in collaboration with the Queensland State Government and local stakeholders to include ecosystem services in the natural resource management planning of the Wet Tropics landscape (Alamgir et al., 2014b; Pert et al., 2014). Furthermore, very few studies on ecosystem services of this landscape are available in the literature, with those studies either focusing on only one ecosystem service from a particular vegetation type (e.g. hydrological service in (Pert et al., 2010)), or in a broad category of ecosystem services (e.g. cultural services in (Pert et al., 2015)). The aim of our study is to adopt a landscape approach to assess the relative capacity of different types of land covers to provide multiple ecosystem services and biodiversity conservation (as an ecological integrity component), and hence to further inform the science about the possibility of integrating ecosystem services and biodiversity in natural resource management planning using expert assessment.

In this study, we answer three research questions: i) how are the key and multiple ecosystem services supply varied across the landscape? ii) what is the spatial congruence between biodiversity and the global climate regulation ecosystem service? iii) what is the spatial congruence between biodiversity and multiple ecosystem services?

3.2. Materials and methods

3.2.1. The study area

The study was conducted in the Wet Tropics bioregion of northeast Australia – covering nearly two million hectares (Fig. 3.1). The landscape of the region is multifunctional and biologically diverse, consisting of rainforests, sclerophyll forests, mangrove forests and shrub lands with many other LULC types. The region supports the Wet Tropics World Heritage Area, which is the second most irreplaceable natural habitat (Bertzky et al., 2013), sixth most irreplaceable natural habitat on the planet considering all species, and the eighth most irreplaceable natural habitat considering threatened species (Le Saout et al., 2013). The region supports the highest biological diversity in Australia, and supplies huge ecosystem services (WTMA, 2009a). The Wet Tropics World Heritage Area has also been listed as a national heritage area of Australia considering its significant indigenous cultural heritage values (Government of Australia, 2007).

Among the different land covers located in the Wet Tropics bioregion, mesophyll, notophyll and microphyll are classified as rainforest types. Additionally, sclerophyll forest is also found in the Wet Tropics and is dominated by eucalyptus and acacias. Mesophyll forests occur in very wet to moist lowland areas with dominant canopy leaf length of more than 12.5 cm, while microphyll forests generally occur in wet highland areas with a dominant canopy leaf length of less than 7.5 cm (Tracey, 1982; Webb, 1968). Notophyll forests are the most extensive rainforests in the Wet Tropics bioregion ranging from the foothills to the uplands with dominant canopy leaf length of 7.5 to 12.5cm (Tracey, 1982; Webb, 1968) (details in supplementary material 1).

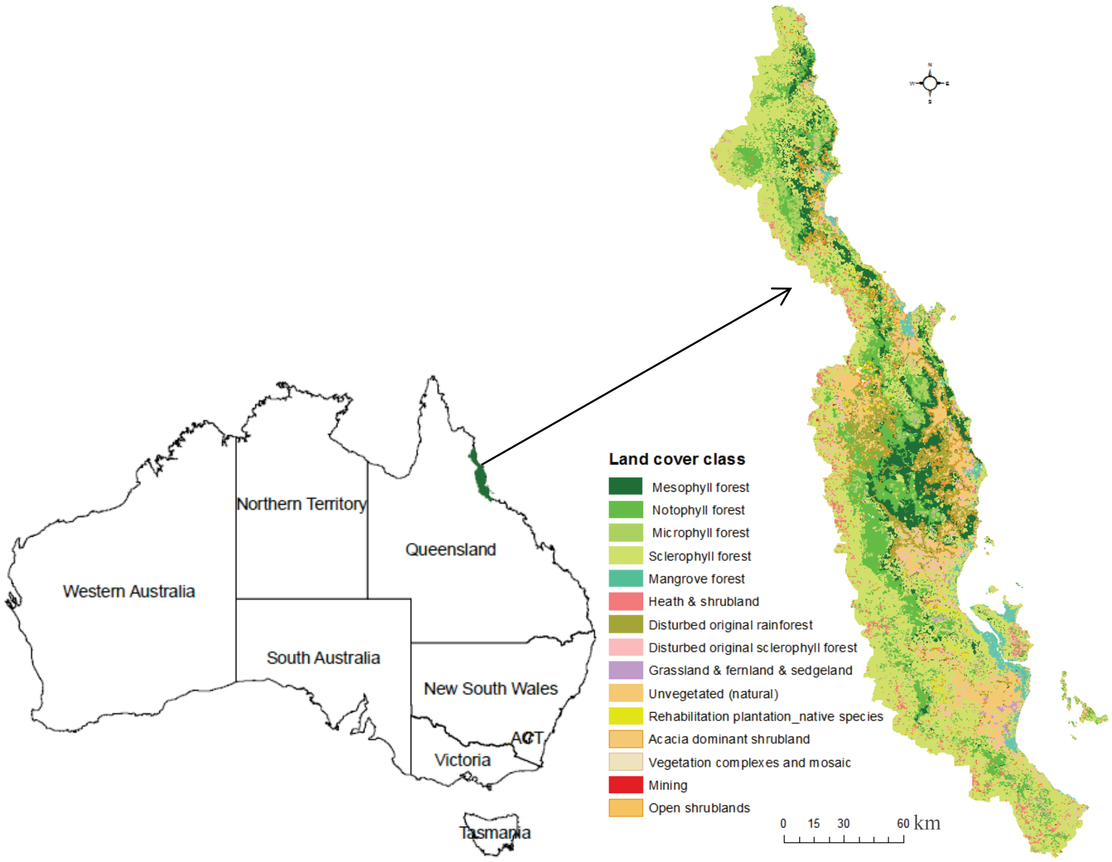

Figure 3.1. The study area – Wet Tropics bioregion, northeast Australia with major Land Use and Land Cover type.

3.2.2. Typology and framework for the ecosystem service assessment

Various typologies for ecosystem service assessment are available in the literature (Boyd and Banzhaf, 2007; Daily, 1997; de Groot et al., 2002; Fisher et al., 2009; MA, 2003, 2005; Mouchet et al., 2014; Posthumus et al., 2010; Raudsepp-Hearne et al., 2010; van Oudenhoven et al., 2012; Wallace, 2007). The MA (MA, 2005) provided the first comprehensive and detailed typology which has subsequently been widely used for ecosystem service assessment (Alamgir et al., 2014a; Banerjee et al., 2013; Baral et al., 2013a; Burkhard et al., 2012; García-Nieto et al., 2013; Sohel et al., 2014). Therefore, in this study, we also followed the typology used in the MA.

Burkhard et al. (2009, 2012) developed a framework for the spatially explicit assessment of ecosystem services supply of a landscape using LULC maps. We collected the spatial datasets of LULC of the region from the Wet Tropics Management Authority (WTMA), and a non-monetary evaluation scheme, based on indicators which were categorized and mapped in relation to relative supply, was applied (after Burkhard et al., 2009, 2012; Sohel et al., 2014).

3.2.3. Indicators for ecological integrity and ecosystem service

For ecosystem service assessment, appropriate indicators are required to comprehensively describe the interaction among ecological process, function and integrity, their components and relative capacities of LULC to supply ecosystem services (Burkhard et al., 2012; de Groot et al., 2010; van Oudenhoven et al., 2012), and as such appropriate ecosystem service indicators play a crucial role in ecosystem service evaluation. van Oudenhoven et al. (2012) have shown that ecosystem service indicators need to be quantifiable, scalable, land cover sensitive and spatially explicit. Burkhard et al. (2009, 2012) and de Groot et al. (2010) have developed a set of ecological integrity and ecosystem service indicators linking LULC utilising the ecosystem service typology of MA (2003, 2005). They have also successfully applied that set of indicators for the assessment of ecological integrity and the ecosystem services linking capacities of LULC. In our study, we have adopted the ecological integrity and ecosystem service indicators approach from Burkhard et al. (2009, 2012) (details in supplementary material 2).

3.2.4. Mapping landscape's capacities to supply ecosystem service

We classified the LULC of the Wet Tropics into 15 broad classes (Fig. 3.1, detailed description of classes is in supplementary material 1) using the current spatial datasets of LULC from WTMA. We developed a matrix linking seven ecological integrity indicators and 23 ecosystem services (on the x-axis) to 15 land cover types (on the y-axis) (Table 3.1). The capacities of different land cover classes to support ecological integrity and particular ecosystem services were assessed on 0-5 scale; where 0 = no relevent capcity of the particular land cover to support the selected ecological integrity component or to supply selected ecosystem service, 1 = low relevant capacity, 2 = relevant capacity, 3 = medium relevant capacity, 4 = high relevant capacity, and 5 =

very high relevant capacity (after Burkhard et al., 2012). The relative scoring of each ecosystem service was conducted in an expert worksop.

3.2.5. Expert assessment of ecosystem services

For the relative scoring of ecological integrity components and ecosystem services a day long expert workshop was arranged at James Cook University, Cairns Campus, Australia in September 2015. There were seven participants from practioners and academics. Prior to the selection of practitioners, semi-structured interviews were conducted from a list of potential participants to elicit their understanding about ecosystem services and ecological dynamics in this region. The selected practitioners had more than 10 years of experience each working directly in the different aspects of forestry in the Wet Tropics, and also possessed an ecological academic background. The academics were selected based on their engagement with ecosystem services research (published) focusing on this region. The relative scoring for each ecological integrity component and ecosystem service was finalised by a complete consensus of all participants. The discussion was continued for each ecological integrity and ecosystem service untill all participants agreed to a relative score for respective ecological integrity component and ecosystem service.

3.2.6. Spatial and statistical analysis

Using ESRI's ArcGIS 10.2 the assessment matrix (Table 3.1) was joined to the polygon attribute tables of the land cover layer and spatially represented and analyzed. Furthermore, in multiple ecosytem services comparison and biodiversity (ecological integrity component) analysis we considered 3-5 relative scores as high-potential, and a 0-2 relative scores as low-potential. We used radar plots to show the relative capacity of each land use and land cover to supply multiple ecosystem services.

3.3. Results

3.3.1. Quantification of ecosystem services

It is clear that undisturbed forest biomes (rainforests (mesophyll-microphyll), sclerophyll forests, and mangrove forests) contain a higher capacity to supply almost all ecosystem services while disturbed forest biomes (disturbed rainforests, disturbed sclerophyll forests) were identified as having a lower capacity to supply almost all ecosystem services (Table 3.1). The heath and shrublands were identified as having a higher capacity to supply few ecosystem services such as erosion regulation, aesthetic and recreational values; whereas grassland, fernland and sedgeland were identified as possessing a higher capacity to supply only two ecosystem services (out of 23) – livestock and intrinsic values of biodiversity (Table 3.1). Rehabilitated plantation forests with native species contained a higher capacity to supply most of the regulating and provisioning ecosystem services, but a lower capacity for cultural services, while *Acacia* dominant shrublands hold a higher capacity to supply most of the cultural services, but a lower capacity for most of the regulating and provisioning services. Almost no regulating and provisioning ecosystem services were found in unvegetated (natural) and mineral extraction sites (Table 3.1).

Table 3.1. Assessment matrix illustrating the capacities of different land cover classes to support ecological integrity and ecosystem services in the Wet Tropics bioregion, Australia. The values/colors indicate the capacities as follows: 0/purple = no relevant capacity; 1/orange = low relevant capacity; 2/olive green = relevant capacity; 3/light green = medium relevant capacity; 4/green = high relevant capacity; and 5/dark green = very high relevant capacity.

Land cover type	**Ecological integrity**	Abiotic heterogeneity	Biodiversity	Biotic waterflows	Metabolic efficiency	Energy capture (Radiation)	Reduction of nutrient loss	Storage capacity (SOM)	**Regulating services**	Local climate regulation	Global climate regulation	Flood protection	Groundwater recharge	Air quality regulation	Erosion regulation	Nutrient regulation	Water purification	Pollination	Cyclone protection	**Provisioning services**	Livestock	Fodder	Capture fisheries	Wild foods	Timber	Wood fuel	Energy	Biochemical and medicines	Freshwater	Habitat	**Cultural services**	Aesthetic values	Recreational values	Intrinsic value of biodiversity
Mesophyll forest		4	5	5	4	4	5	5		5	5	5	5	5	5	5	5	5	5		0	0	0	2	4	1	1	5	5	5		5	5	5
Notophyll forest		5	5	4	5	5	5	5		5	5	5	5	5	5	5	5	5	5		0	0	0	2	4	1	1	5	5	5		5	5	5
Microphyll forest		4	4	3	2	2	5	4		5	5	5	5	5	5	5	5	5	5		0	0	0	1	2	1	1	5	5	5		5	5	5
Sclerophyll forest		4	4	3	2	3	3	3		3	2	3	3	3	3	3	3	5	3		4	3	0	2	4	4	4	5	3	4		5	5	4
Mangrove forest		3	3	4	4	4	4	4		4	5	5	5	5	5	5	5	4	5		0	0	5	2	1	2	2	4	0	4		5	5	4
Heath & shrubland		4	3	2	3	2	3	2		3	2	3	3	3	4	3	3	4	2		2	0	0	1	0	0	0	4	2	3		5	4	4
Disturbed original rainforest		2	3	2	2	2	2	3		2	1	2	2	2	2	2	2	2	1		3	1	0	1	1	1	1	1	4	2		2	2	1
Disturbed original sclerophyll forest		2	2	1	1	1	1	2		1	1	1	1	1	1	1	1	2	1		4	3	0	1	1	2	2	1	2	2		2	2	1
Grassland,fernland & sedgeland		3	3	2	3	2	3	2		2	2	3	3	2	3	3	3	2	1		5	4	0	1	0	0	0	2	3	3		3	2	4
Unvegetated (natural)		1	1	0	0	0	0	1		0	0	0	0	0	0	0	0	0	0		0	0	0	1	0	0	0	0	3	1		3	3	3
Rehabilitation plantation_native species		2	2	3	4	4	3	2		3	5	3	3	4	3	3	3	2	4		2	0	0	1	4	3	3	1	4	2		2	2	2
Acacia dominant shrubland		2	3	2	2	2	3	2		3	2	2	2	2	2	3	2	4	3		3	3	0	1	1	3	3	3	2	3		4	4	3
Vegetation complexes and mosaic		3	4	2	3	3	3	3		3	3	3	3	3	3	3	3	3	4		1	0	0	1	2	2	2	4	4	4		4	4	4
Mineral extraction sites		0	1	0	0	0	0	1		0	0	0	0	0	0	0	0	0	0		0	0	0	0	0	0	0	0	3	1		1	1	1
Open shrublands-exotics dominant		2	2	2	3	2	3	2		2	2	3	3	3	3	2	3	4	3		2	1	0	1	1	2	2	2	3	3		3	2	2

The spatial distribution of key regulating, provisioning, cultural ecosystem services, and ecological integrity component (Fig. 3.2) showed that the aesthetic value and biodiversity value across the landscape were very high, however the capacity to provide global climate regulation and timber provision ecosystem services was lower in most parts of the landscape.

3.3.2. Landscape pattern of multiple ecosystem services

Fig. 3.3 shows the spatial pattern of multiple ecosystem services for each category – regulating, provisioning, and cultural services – based on above average values (high-potential). All 10 assessed regulating ecosystem services of high-potential were found in the interior area (mainly undisturbed rainforests) while no high-potential regulating service was found in the adjacent areas. Eight high-potential provisioning ecosystem services were identified towards the outside area of the landscape (containing mainly sclerophyll forests and disturbed rainforests) while four high-potential provisioning ecosystem services were found towards the inside area of the landscape (mainly rainforests). However, no area was identified that did not provide at least one high-potential provisioning ecosystem service. All three assessed cultural ecosystem services of high-potential were found in the majority areas of the landscape while some lowland areas and upland areas adjacent to the rainforests were identified with no high-potential cultural services. Overall, more than 15 high-potential ecosystem services (total 23 assessed) were found across the majority of the areas of the landscape while a very few areas were identified providing 1-4 high-potential ecosystem services (Fig. 3.3). Each land cover showed a heterogeneous capacity to supply multiple ecosystem services (Fig 3.4).

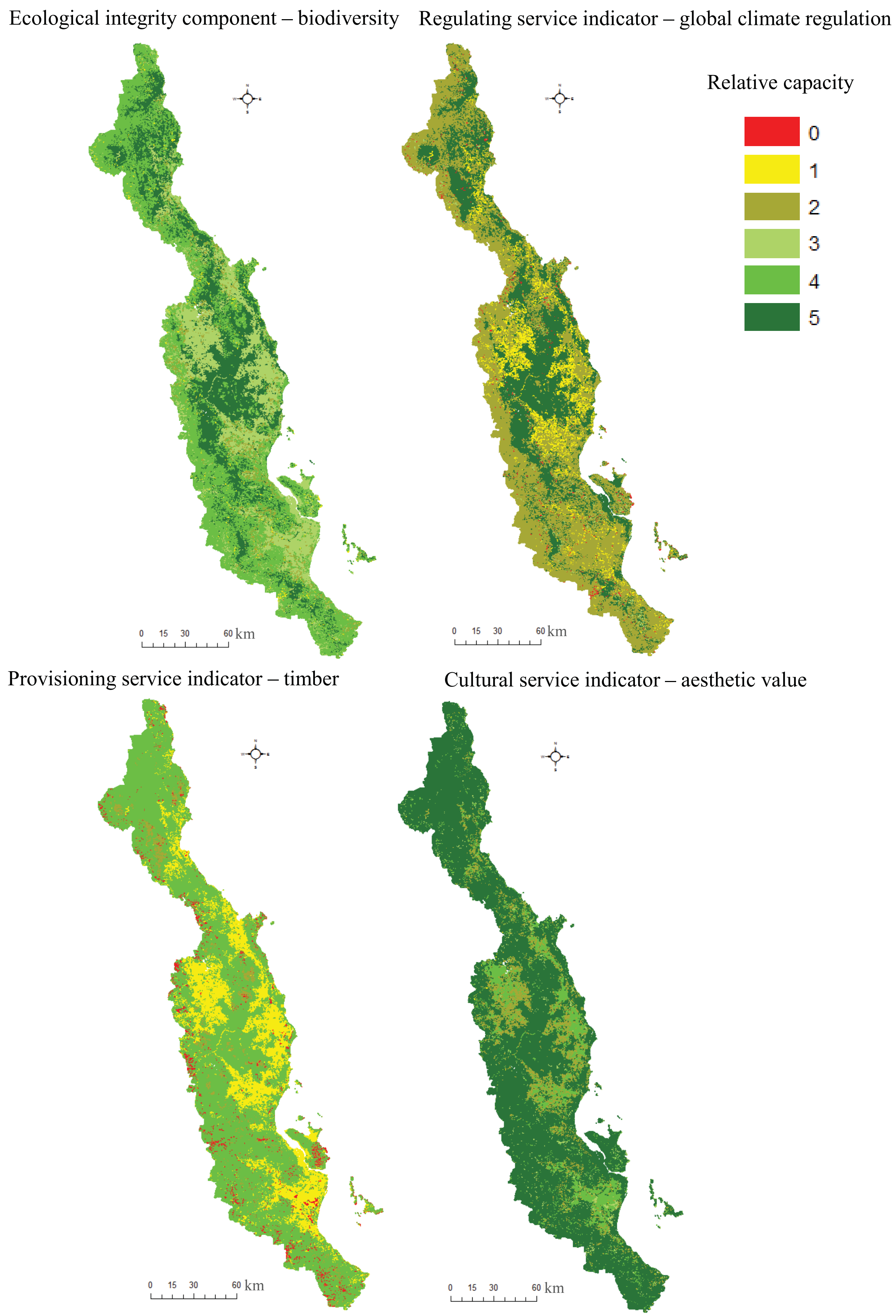

Figure 3.2. The spatial distribution of supply of selected ecological integrity and ecosystem services in the Wet Tropics bioregion, northeast Australia

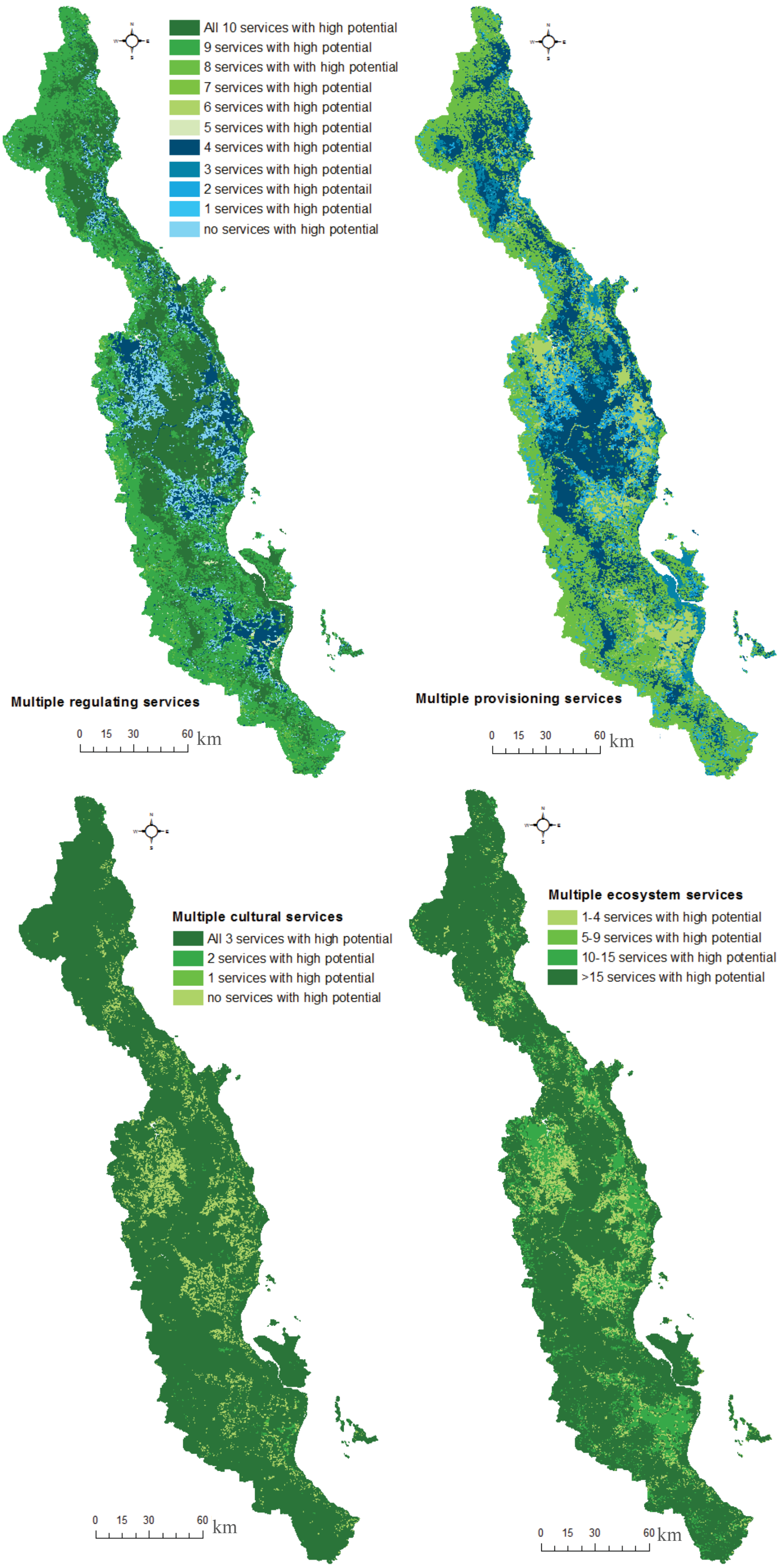

Figure 3.3. The spatial distribution of multiple ecosystem services with high-potential. The bottom right figure considered all ecosystem services irrespective of categories.

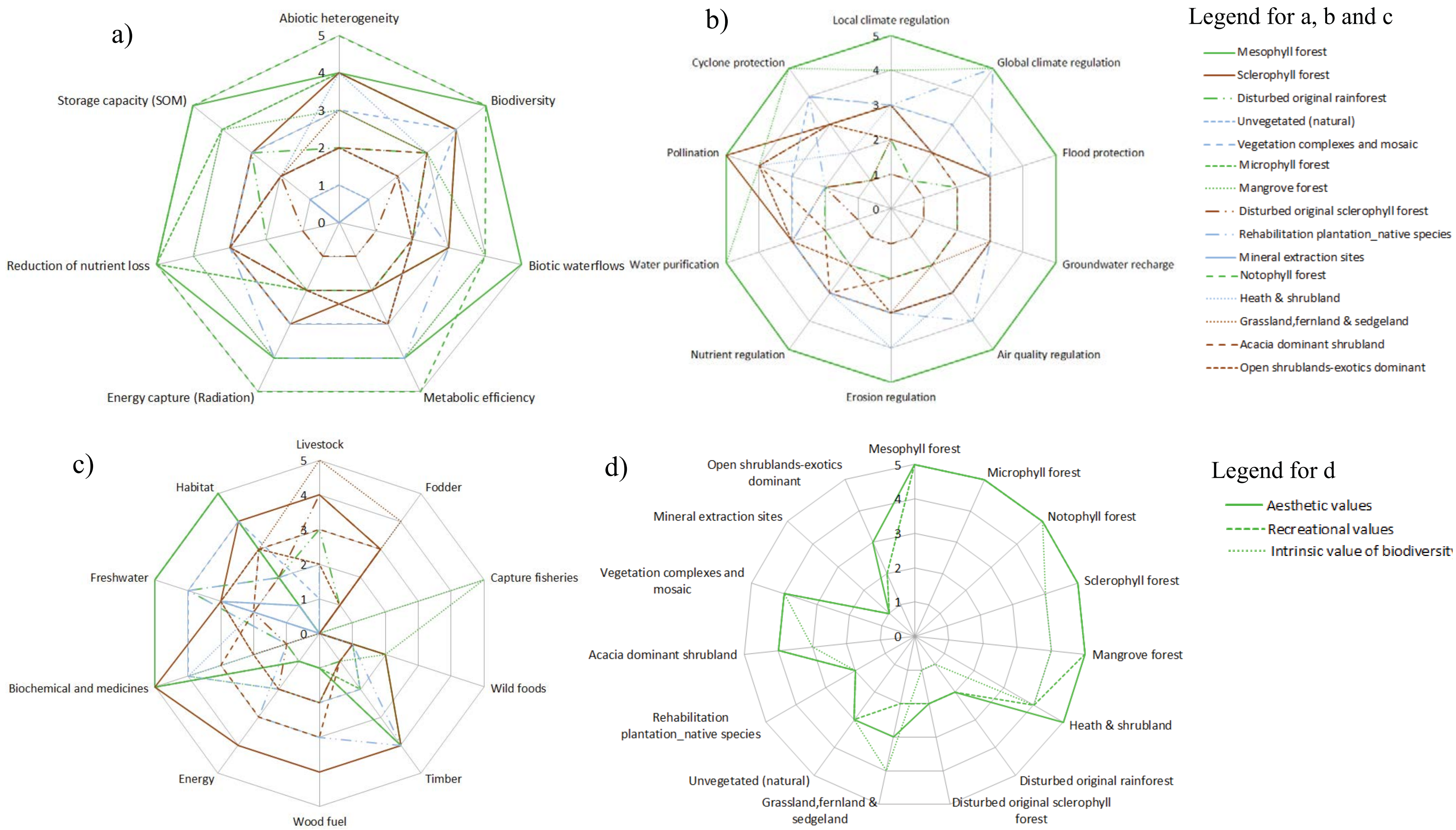

Figure 3.4. Ecological integrity and ecosystem services: a) ecological integrity, b) regulating services, c) provisioning services d) cultural services.

3.3.3. Relation between biodiversity and ecosystem services

Initially, we spatially compared the biodiversity (ecological integrity component) and global climate regulation ecosystem service (Fig. 3.5 left panel). The interior areas (different sub-categories of the rainforests and upland rainforest) exhibited high-potential for biodiversity and global climate regulation ecosystem service while a substantial part of the area exhibited high-potential for biodiversity, but low-potential for global climate regulation ecosystem service. In comparing multiple ecosystem services and biodiversity (Fig. 3.5, right panel), majority of the areas in the landscape showed > 15 high-potential ecosystem services as well as high potential biodiversity, while few areas were identified as providing 1-9 high-potential ecosystem services but low-potential biodiversity.

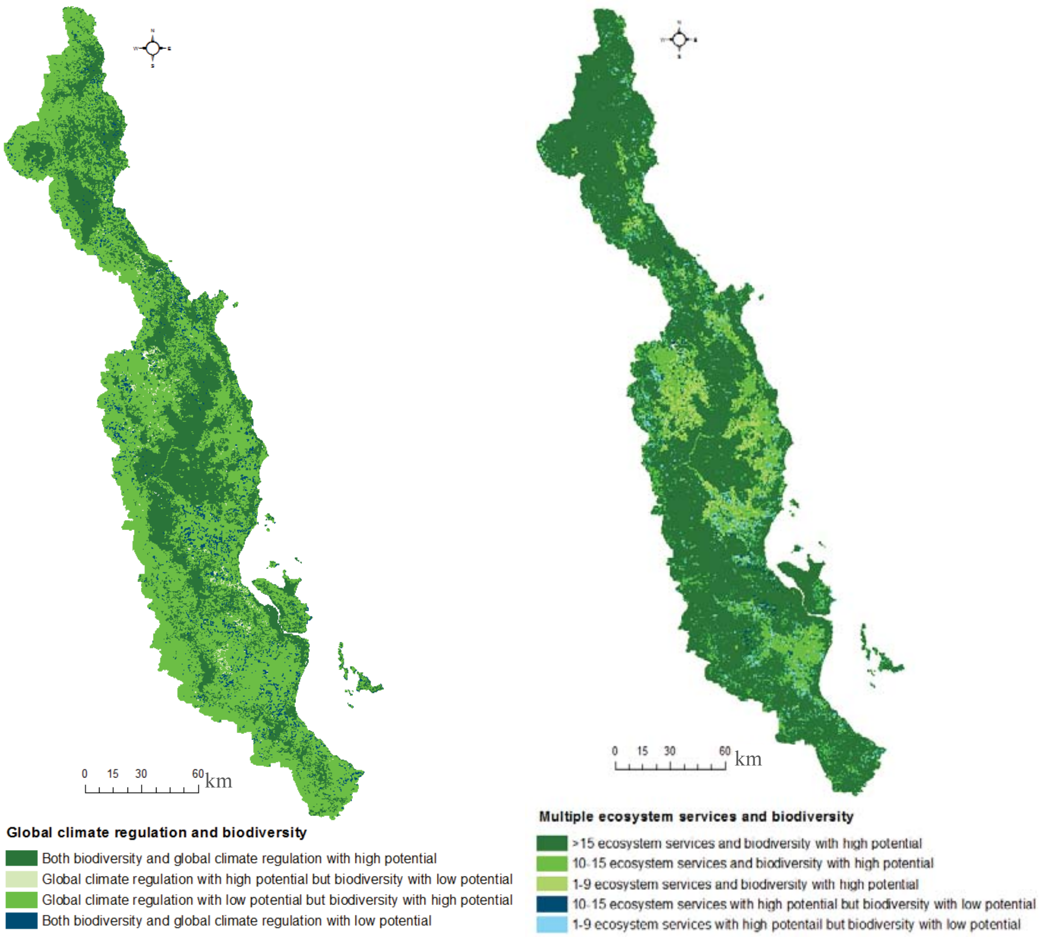

Figure 3.5. The spatial congruence and divergence between biodiversity (ecological integrity component) and ecosystem services. Left panel: the areas with global climate regulation ecosystem service and biodiversity values; right panel: multiple ecosystem services and biodiversity values.

3.4. Discussion

The rainforests components (mesophyll-microphyll forests) provide higher amounts of key ecosystem services (Table 3.1 and Fig.3. 2) and a high-potential of multiple regulating, cultural and overall multiple ecosystem services in the landscape (Figs. 3.3 and 3.4). The higher capacity of the rainforests to provide key and multiple ecosystem services are also reported from other regions around the world (Grimaldi et al., 2014). Multiple ecosystem services are produced from several interactions in a complex system (Harrison et al., 2014). Therefore, the diversified structural and functional ecological components that exist in the Wet Tropics rainforests may be the reason for them containing higher capacity to provide high potential multiple ecosystem services. Degraded forests usually produce more multiple provisioning ecosystem services while intact rainforests produce more multiple regulating services (Baral et al., 2013a), and our study revealed a similar trend (Fig. 3.3). Our study also revealed that even sparsely vegetated tree land cover (e.g. open shrublands) provide higher key ecosystem services (Table 3.1), and high-potential multiple ecosystem services (Fig. 3.3) when compared with non-tree vegetated land cover (e.g. grasslands, fernland, and sedgeland; and mineral extraction sites). Likewise, Dhanya et al. (2014) reported that sparse vegetation plays a fundamental role in enhancing the landscape capacity to supply multiple ecosystem services. Isolated trees positively influence ecosystem dynamics (Williams-Linera et al., 1998), which can enhance ecosystem services supply in the landscape.

Most of the high potential regulating, cultural and multiple ecosystem services supply decline can be attributed to disturbances, which are more common along the edges of the rainforests and sclerophyll forests in this landscape (Fig. 3.3). The microclimate in the forest edges are different and more susceptible to strong wind, hence vegetation structure from the interior forests are different (Magnago et al., 2015). Forest fragmentations are mainly responsible for creating more edges (Briant et al., 2010). In the Wet Tropics landscape, forest fragmentation (Stork and Turton, 2008; Williams and Pearson, 1997) is frequent due to forest disturbances caused by tropical cyclones (Turton, 2012) which predominantly produce more edge effects (disturbed forests both rainforests and sclerophyll forests), which substantially reduces the capacity of the landscape to supply multiple ecosystem services. Similar negative effects of forest disturbances on the supply of key ecosystem services have been reported from Amazonian rainforests (Nascimento and Laurance, 2004).

Our findings of the congruence of large areas in the landscape with high-potential biodiversity and high-potential global climate regulation ecosystem service (Fig. 3.5) supports findings by Harrison et al. (2014) who found a strong positive link between biodiversity and global climate regulation ecosystem service. Chan et al. (2006) also reported a positive association between biodiversity and global climate regulation ecosystem service. The high biodiversity values of these rainforest areas are globally recognized (Le Saout et al., 2013). The spatially congruent areas between biodiversity and global climate regulation ecosystem service are only found in the interior undisturbed rainforest areas of the Wet Tropics bioregion (comparing Figs. 3.1 and 3.5). The postulation of spatial divergence between biodiversity and global climate regulation ecosystem service may be because of the lower tree abundance or lower number of large trees that exist. Most of the identified divergence areas included sclerophyll forests, disturbed rainforests and disturbed sclerophyll forests (Fig. 3.5). Our findings are aligned with the findings of Slik et al. (2013) who reported that large trees are critical in providing climate regulation ecosystem service in tropical forested landscapes, and Oliveira et al. (2008) also found that there were reduced numbers of large trees in the disturbed and fragmented tropical forest landscape. Whatever the postulation is, this area in the Wet Tropics may be a priority to enhance both biodiversity and climate regulation ecosystem service in this landscape.

Figure 3.6. Contrasting landscape in the Wet Tropics bioregion, Australia (clockwise from the top left) – (a) mesophyll forest (b) notophyll forest (c) sclerophyll forest (d) mangrove forest (e) heath and shrub land (f) rehabilitation plantation (Photo credits: Mohammed Alamgir).

We found a spatial congruence between high-potential multiple ecosystem services and high- potential biodiversity over a large area of this landscape (Fig. 3.5). This may be due to the different vegetation types that provide different-high potential ecosystem services due to the diversified vegetation structure of this landscape (Fig. 3.6). For example, undisturbed rainforests contain a higher capacity to supply global climate regulation ecosystem service, but lower capacity to supply fodder. Contrastingly, the disturbed sclerophyll forests contain a higher capacity to supply fodder but a lower capacity to supply global climate regulation ecosystem services. Spatial congruence between biodiversity and certain forest based ecosystem services in the landscape has been revealed previously (Chan et al., 2006; Egoh et al., 2009; Harrison et al., 2014). We urge that enhancement and conservation of forested vegetation are required to fulfill both objectives- maintaining ecosystem services supply (forest-based) and biodiversity conservation in the tropical forested landscape. However, careful selection of multiple ecosystem services is required, especially in the case of non-forest based ecosystem services.

3.5. Conclusions

Conservation of interior undisturbed rainforest is critical to maintain the supply of key and multiple ecosystem services in the landscape. As spatial congruence exists in these areas with regards to biodiversity and the global climate regulation service, so biodiversity conservation objectives may also be achieved with a contribution to global climate change mitigation efforts. As forest disturbances reduce the capacity to provide key and multiple high-potential ecosystem services, so does minimal or no disturbance need to be ensured. Furthermore, management intervention priorities need to increase tree abundance both in non-tree vegetated land covers and disturbed forested lands. This may then assist in increasing the high-potential multiple ecosystem services supply from this landscape. The integration of high-potential multiple ecosystem services supply and high-potential biodiversity conservation is possible in the tropical forested landscape provided that multiple ecosystem services are forest-based (e.g. global climate regulation, air quality regulation, and habitat provision). Therefore, we argue a careful selection of multiple ecosystem services to integrate high-potential multiple ecosystem services and high-potential biodiversity in the conservation planning and assessment is required.

Acknowledgements

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OVERVIEW OF CHAPTER 4

This chapter is based on a paper under review (second revision submitted) to *Science of the Total Environment* journal[6], with minimal formatting changes. I address the second objective of the thesis: to investigate the interactions among multiple ecosystem services and how the supply of one ecosystem service is influenced by others.

In this chapter I investigate the pattern, synergies and trade-offs among multiple ecosystem services supplied by rainforests, sclerophyll forests and rehabilitated plantation forests of the Wet Tropics bioregion, northeast Australia. I interpret the findings to recommend the options that need to be considered for the maximization of the supply of multiple ecosystem services from the forested landscape of the Wet Tropics bioregion.

[6] **Alamgir, M.**, Turton, S.M., Macgregor, C.J., Pert, P.L. (in review, second revision submitted). Ecosystem services capacity across heterogeneous forest types: Understanding the interactions and suggesting pathways for sustaining multiple ecosystem services. *Science of the Total Environment*

CHAPTER 4

ECOSYSTEM SERVICES CAPACITY ACROSS HETEROGENEOUS FOREST TYPES: UNDERSTANDING THE INTERACTIONS AND SUGGESTING PATHWAYS FOR SUSTAINING MULTIPLE ECOSYSTEM SERVICES

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A sclerophyll forest area in the Wet Tropics: provides multiple ecosystem services (Photo credit: Mohammed Alamgir)

ABSTRACT

As ecosystem services supply from tropical forests is declining due to deforestation and forest degradation, much effort is essential to sustain ecosystem services supply from tropical forested landscapes, because tropical forests provide the largest flow of multiple ecosystem services among the terrestrial ecosystems. In order to sustain multiple ecosystem services, understanding ecosystem services capacity across heterogeneous forest types and identifying certain ecosystem services that could be managed to leverage positive effects across the wider bundle of ecosystem services are required. We sampled a total of 64 plots from three forest types, tropical rainforests, sclerophyll forests, and rehabilitated plantation forests, over an area of 32,000 m^2 from the Wet Tropics bioregion, Australia, aiming to compare supply and evaluate interactions and patterns of eight ecosystem services (global climate regulation, air quality regulation, erosion regulation, nutrient regulation, cyclone protection, habitat provision, energy provision, and timber provision). On average, multiple ecosystem services were highest in the rainforests, lowest in sclerophyll forests, and intermediate in rehabilitated plantation forests. However, a wide variation was apparent among the plots across the three forest types. The global climate regulation service had a synergistic impact on the supply of multiple ecosystem services, while the nutrient regulation service was found to have a trade-off impact. Considering multiple ecosystem services, most of the rehabilitated plantation forest plots shared the same ordination space with rainforest plots in an ordination analysis, indicating that rehabilitated plantation forests may supply certain ecosystem services nearly equivalent to rainforests. Two synergy groups and one trade-off group were identified. Apart from conserving rainforests and sclerophyll forests, our findings suggest two additional integrated pathways to sustain the supply of multiple ecosystem services from a heterogeneous tropical forest landscape: (i) rehabilitation of degraded forests aiming to provide global climate regulation and habitat provision ecosystem services and (ii) management intervention to sustain global climate regulation and habitat provision ecosystem services.

Key words: pattern; synergies; global climate regulation; habitat provision

4.1. Introduction

The global flow of ecosystem services from tropical forests is the highest among terrestrial ecosystems (Costanza et al., 1997). Tropical forests are also rich in multiple ecosystem services (Alamgir et al., 2016; Lewis, 2006; Saatchi et al., 2011). Tropical forest areas are shrinking worldwide due to deforestation, land use change, and many other reasons (Achard et al., 2002; Hansen et al., 2013). Moreover, a substantial part of the remaining tropical forests are experiencing various forms of degradation (Ghazoul et al., 2015). Globally, more than 500 million hectares of tropical forests are estimated to be in various stages of degradation (Ghazoul et al., 2015), undermining the capacity of tropical forest landscapes to supply valuable multiple ecosystem services (Liu et al., 2015). Therefore, sustaining multiple ecosystem services supply from tropical forests are necessary, but it remains as a challenge both for the scientific and policy arenas (Portman, 2013). Furthermore, identification of certain ecosystem services that could be managed to leverage positive effects across a wider bundle of ecosystem services may be useful in effective decision-making processes (Egoh et al., 2009; Harrison et al., 2014; Villamagna et al., 2013).

One problem of sustaining the supply of multiple ecosystem services from a tropical forested landscape is the optimal quantification and understanding of the pattern and interaction of multiple ecosystem services across different forest types within a landscape, as multiple ecosystem services are often determined by the vegetation attributes of a forest type within the forested landscape (Galicia and Zarco-Arista, 2014; Palomo et al., 2013). Multiple ecosystem services at the landscape level are also affected by the land cover types (Burkhard et al., 2012), environmental variables (Rasche, 2014), and management regimes (Castro et al., 2015; Sohel et al., 2014). However, as a whole, the multiple ecosystem services across different forest types (with different vegetation attributes) within a landscape are still unclear (de Groot et al., 2010). As such, understanding the patterns, interactions, and quantification of multiple ecosystem services using vegetation attributes needs to be further examined if management factors (such as optimal forest rehabilitation) are to be fully considered in sound decision-making processes used to sustain the multiple ecosystem services supply from tropical forested landscapes (Alamgir et al., 2014a; Zuidema et al., 2013).

Another problem of sustaining multiple ecosystem services from a heterogeneous tropical forested landscape is the identification of the groups of ecosystem services, which act synergistically with each other and which may undermine the supply of one another (the so-called trade-off services). On a broader scale (beyond the forest types), the synergies and trade-off groups often vary with land cover (de Groot et al., 2010; Foley et al., 2005). However, there is still an unresolved question, that is, how do synergies and trade-off groups of ecosystem services vary across different forest types within a heterogeneous forested landscape (Galicia and Zarco-Arista, 2014)? If a management option is taken into consideration targeting sustaining an ecosystem service supply, it may have a positive or negative impact on the supply of other ecosystem services, which is principally determined by the synergies or trade-offs between the particular ecosystem service and other ecosystem services (Baral et al., 2013b; Bennett et al., 2009; Galicia and Zarco-Arista, 2014; Raudsepp-Hearne et al., 2010). Therefore, it is crucial to identify the synergies and trade-off groups of multiple ecosystem services across the forest types (Galicia and Zarco-Arista, 2014; Villamagna et al., 2013), if multiple ecosystem services from a landscape are to be enhanced.

Our study compares the following eight ecosystem services: global climate regulation, air quality regulation, erosion regulation, nutrient regulation, cyclone protection, habitat provision, energy provision, and timber provision from rainforests, sclerophyll forests, and rehabilitated plantation forests of the Wet Tropics bioregion of northeast Queensland, Australia. We considered these eight ecosystem services noting their significance at the local–regional scale (air quality regulation, cyclone protection, energy provision, and timber provision) (Turton, 2008; WTMA, 2009a) and global scale (erosion regulation, nutrient regulation, global climate regulation, and habitat provision) (Alamgir et al., 2016; Hilbert et al., 2001; Preece et al., 2012; Williams et al., 2003). Many of these ecosystem services are thought to closely interact with each other (Bennett et al., 2009; Egoh et al., 2009; Harrison et al., 2014; Villamagna et al., 2013) and potentially maintain the forest condition of the Wet Tropics bioregion into the future (Stork and Turton, 2008). In order to achieve the study aims, we collected forest vegetation attribute data at the scale of the forest stand from the Wet Tropics bioregion and conducted mean comparison, multivariate analysis, and two-way cluster analysis. We then assessed the pattern of these eight ecosystem services across three forest types in the landscape. In addition, we evaluated the interaction among these ecosystem services, and the synergies and trade-off groups among the eight ecosystem services identified. Finally, we discuss the opportunities and potential mechanisms required to sustain the supply of multiple ecosystem services across a heterogeneous tropical forest landscape.

4.2. Materials and methods

4.2.1. The study area

Our study was conducted in the Wet Tropics bioregion, northeast Queensland, Australia (Fig. 4.1), which has an area of approximately 2 million ha (Goosem, 2002) and comprises a heterogeneous, complex landscape dominated by intergrading rainforests and sclerophyll forests (Ash, 1988). This is one of the 89 bioregions in Australia that are distinct in climate, geology, landform pattern, ecological features, and biological communities (Department of Environment, 2015). The Wet Tropics bioregion experiences a seasonally wet tropical climate with a mean annual rainfall of 1200–4000 mm, mean annual temperature range of 17–31°C (Goosem, 2002; Trott, 1996), and elevation ranges from a few metres above mean sea level (msl) to approximately 1000 m, although the highest peak within the region is 1622 m.

4.2.2. Overview of the forest types

The rainforests of the Wet Tropics bioregion are characterized by dense canopies, high biological diversity, and broad-leaved trees from a variety of plant families, including the Moraceae, Myrtaceae, Rutaceae, and Sapindaceae (Adam, 1994; Bowman, 2000). The rainforests of this bioregion are distributed from the lowland floodplains to the uplands (>1000 m above msl). The structure and composition of the rainforests of the Wet Tropics are largely controlled by the rainfall, temperature, soil type, and elevation (Webb and Tracey, 1981). The rainforests have broadly been classified into mesophyll, microphyll, and notophyll types, based on dominant canopy leaf sizes. Mesophyll-type dominant canopy leaf blade length of >12.5 cm generally occur in very wet to moist low lands; microphyll-type dominant canopy leaf blade length of <7.5 cm generally occur in wet highland areas; and notophyll-type dominant canopy leaf blade length of 7.5–12.5 cm is the most extensive rainforest in this region ranging from foot hills to uplands (Tracey, 1982; Webb, 1959). Most of the rainforests in the Wet Tropics have been selectively logged since 1880 (Kanowski et al., 2003); however, during the mid-late 20th century, the focus had changed to rainforest conservation and restoration (Kanowski et al., 2003). Subsequently about 45% (894,420 ha, mainly rainforests) of this bioregion were inscribed on the World Heritage List in 1988 (UNESCO, 1988). This region has recently been recognized as the second most irreplaceable natural world heritage area (Bertzky et al., 2013) and the sixth most irreplaceable protected area on the planet (Le Saout et al., 2013). The rainforest restoration in the Wet Tropics bioregion got momentum after 1988, with one of the main forms of rainforest restoration, including small-scale plantings of nearby native rainforest species in the previously cleared or degraded rainforest areas (Catterall and Harrison, 2006). This study examined rehabilitated plantation forests that were 10–19 years old, reestablished with native local rainforest tree species primarily for ecological connectivity, and a significant number of large trees were retained after previous degradation. The sclerophyll forests (including woodland) in the Wet Tropics are distributed across the region with considerable variation in structure and composition, while having relatively open canopies (Stork and Turton, 2008; Tracey, 1982; WTMA, 2014). The dominant species are *Eucalyptus, Corymbia, Melaleuca, Acacia, Allocasuarina, Casuarina, Lophostemon*, and *Syncarpia* (WTMA, 2014).

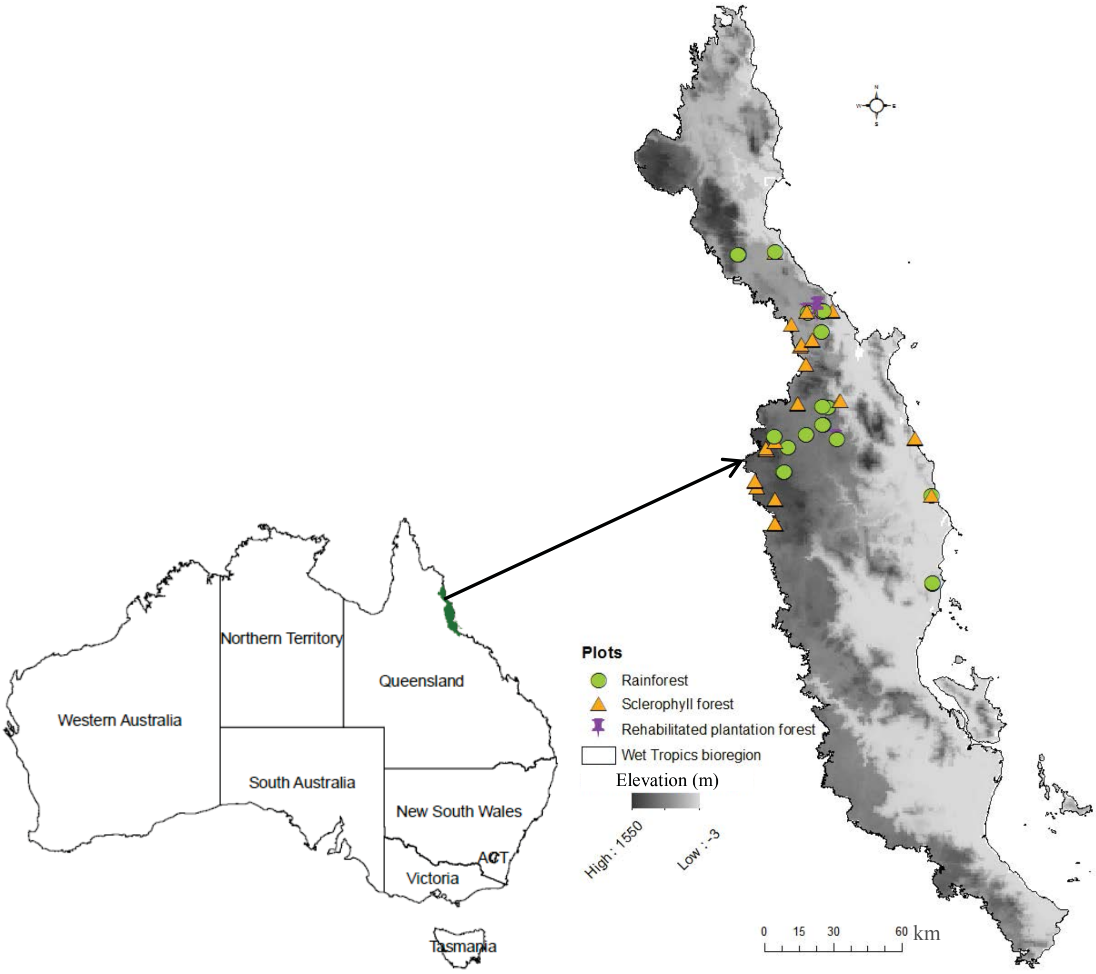

Figure 4.1. Study area of the Wet Tropics bioregion, northeast Queensland, Australia, with location of plots.

4.2.3. Framework for ecosystem services assessment

As ecosystem services assessment is a relatively new and evolving concept (Fisher et al., 2009; Seppelt et al., 2012), a number of frameworks have evolved in the literature for the classification and assessment of ecosystem services (Bagstad et al., 2013; Boyd and Banzhaf, 2007; de Groot et al., 2010; de Groot et al., 2002; Fisher et al., 2009; MA, 2005; van Oudenhoven et al., 2012). The Millennium Ecosystem Assessment (MA) is one of the pioneering frameworks, which has subsequently been widely used in ecosystem services assessment (Baral et al., 2013a; Burkhard et al., 2012; Harrison et al., 2014; Pert et al., 2010; Schneiders et al., 2012; Sohel et al., 2014). Therefore, we used the framework of the MA (MA, 2005) for selected ecosystem services (Table 4.1).

Table 4.1. Ecosystem service components, attributes, and indicators used for the assessment

Ecosystem service component	Attribute	Indicator (measurement unit)
Global climate regulation	Ecosystem plays an important role in global climate regulation by sequestering greenhouse gases	Sequestered atmospheric CO_2 by above ground tree biomass (CO_2 equ. Mg ha^{-1})
Air quality regulation	The capacity of ecosystem to remove toxic and other elements from the atmosphere	Tree canopy cover (%)
Erosion regulation	Vegetative cover plays an important role in soil retention	Stratified vegetation cover index
Nutrient regulation	The capacity of ecosystems to carry out (re)cycling of N, P and other nutrients	Nitrogen regulation (N kg ha^{-1})
Cyclone protection*	The presence of forest ecosystems can dramatically reduce the damage caused by cyclones	Coefficient of variance (CV) of tree diameter at breast height (dbh)
Habitat provision	Importance of ecosystem to provide habitat for species (particularly fauna) and natural biodiversity.	Multicriteria index
Energy provision	The presence of trees or plants with potential use as energy source	Above-ground tree biomass (AGB) (Mg ha^{-1})
Timber provision	The presence of species with potential use for timber	Tree basal area (BA)(m^2 ha^{-1})

According to de Groot et al. (2010); MA (2005); Burkhard et al. (2012); and de Groot et al. (2002); Alamgir et al. (2016), *we used the term cyclone protection rather than storm protection (which was used in MA, 2005) considering local preferences.

4.2.4. Indicators for ecosystem services assessment

Indicators are the variables that represent aggregated information about certain phenomena (Müller and Burkhard, 2012). As the ecosystem services are the products of complex ecological phenomena, direct measurements of many ecosystem services are not possible (Egoh et al., 2012). Therefore, researchers usually rely on proxies to quantify ecosystem services (Egoh et al., 2012; Seppelt et al., 2011; Seppelt et al., 2012). Some examples of widely used proxies are area (Martínez et al., 2009; Raudsepp-Hearne et al., 2010), land use, and land cover (Burkhard et al., 2012; Sohel et al., 2014). These proxies are useful for rapid measurement of ecosystem services (Burkhard et al., 2012), but are likely unable to explain the interaction and pattern of ecosystem services within a vegetation type (Bennett et al., 2009; Kandziora et al., 2013). The indicators used in this study (Table 4.1) are based on vegetation attributes, familiar among forest researchers, easily obtainable, and likely to represent ecosystem services more consistently than other proxies (e.g., area and vegetation cover). The selection of indicators is consistent with several previous studies (Baral et al., 2013a) and propositions (de Groot et al., 2010; Kandziora et al., 2013); however, some indicators are naturally related to each other. All our indicators are state indicators, and hence they represent the state of the supply of each ecosystem service (de Groot et al., 2010).

4.2.5. Sampling and data collection

This study is based on field sampling. A total of 64 plots of 0.05 ha (50 × 10 m) from each of the three forested ecosystems in the Wet Tropics bioregion, that is, 24 from rainforests (mesophyll forest 13, notophyll forest nine, and disturbed rainforest two), 34 from sclerophyll forests (sclerophyll forest and woodland 32 and disturbed sclerophyll forest two), and six from rehabilitated plantation forests, were sampled. The field plots were placed at locations that were away from creeks and had a minimum distance of 50 m from other land uses to avoid edge effects, as determined using ESRI GIS 10.2. Considering environmental variation, our plots were distributed from a few metres to >1000 m above msl and in areas where annual rainfall was <1000 mm to >3500 mm. As different rainforest types and sclerophyll forest types are situated along the same environmental gradient in the Wet Tropics bioregion, some of our sampling plots were very closely located to each other to cover sampling from different rainforest types and sclerophyll forest types on the same environmental gradient (Fig. 4.1). This technique provided an opportunity to assess the interactions and patterns among forest types and also avoided major bias sourced from different forest types within a particular broad forest type (both in rainforests and sclerophyll forests) in the same environmental gradient.

We used a modified transect method for tree data collection within the 0.05-ha (50 × 10 m) plots (Fig. 4.2). This modified transect method was found suitable both for rainforests (Preece et al., 2012) and sclerophyll forests (Burrows et al., 2002). The diameter at breast height (dbh) of trees was measured using a dbh tape, height of representative trees (two to three trees from each subplot) was measured using a

Forestry Pro Laser Range Finder Hypsometer, and canopy cover was measured using a convex spherical crown densitometer. Large trees (≥20-cm dbh) were measured in the 0.05-ha plots and medium (≥ 10 cm dbh) and small trees (≥2.5 cm) were measured in the 0.03- and 0.015-ha subplots, respectively (Fig 2). Along the centerline of transects, we counted the number of fine and coarse woody debris pieces that intersected the centerline ≤1 m high. We also measured canopy cover and ground cover (%) from a height of 1 m at three 1-m^2 subplots placed at 5, 25, and 45 m. Each plot was spatially referenced and attributes such as slope, soil type, and elevation were also recorded.

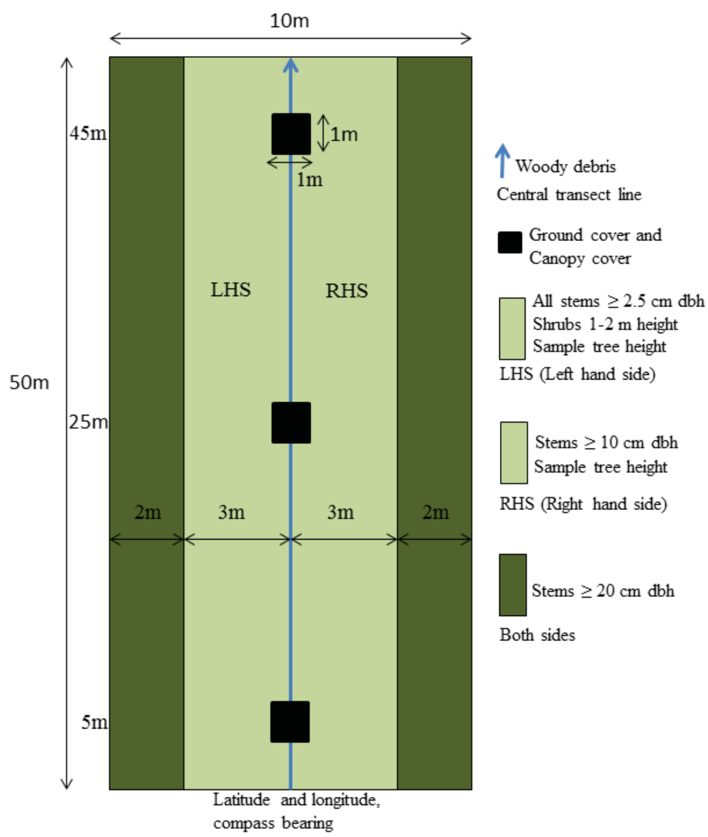

Figure 4.2. Plot layout showing the size of the subplots with the measured vegetation attributes.

4.2.6. Assessing ecosystem services

Most of the ecosystem services we examined in this study support the productivity of the forests over the long term through maintaining natural habitat, cycling N in the ecosystems, regulating local and global climate, producing timber, controlling soil erosion, and enhancing resistance of the forests to tropical cyclones.

4.2.6.1. Global climate regulation

We estimated the sequestered CO_2 equivalent (CO_2 equ Mg ha^{-1}) as an indicator of global climate regulation ecosystem service (Table 4.1) using the following equation:

CO_2 equ. = AGB × 0.47 × 3.67,   (1)

where AGB was estimated using Chave's allometric (Chave et al., 2005), that is

$AGB = \rho \times \exp(-1.499 + 2.148 \ln(dbh) + 0.207 (\ln(dbh))^2 - 0.0281(\ln(dbh))^3)$

(2),

where AGB (above-ground biomass) was measured in kg, dbh is measured in cm, and ρ is wood density measured in g cm^{-3}. For wood density, we used the reported default value for Australian tropical forests 0.5 g cm^{-3} (500 kg m^{-3}) (Department of Climate Change and Energy Efficiency, 2010). In order to convert AGB into biomass carbon storage, we used 0.47, and to convert carbon storage into CO_2 equ, we used 3.67, both values recommended by the Intergovernmental Panel for Climate Change for tropical forests (IPCC, 2006). Chave's formula (Chave et al., 2005) was found reliable and provided a better estimate for AGB for tropical forests in our region (Preece et al., 2012).

4.2.6.2. Air quality regulation

The capacity of a forest to regulate air quality mostly depends on the canopy cover of the forest, because tree leaf surfaces are the main structural elements that are used to regulate toxic elements from the atmosphere (Mincey et al., 2013; Nowak et al., 2014). Therefore we estimated canopy cover (%) as an indicator of air quality regulation ecosystem service (Table 4.1) following Lemmon (1956). We further classified canopy gap fraction into four classes (1 to 4) with equal intervals of 25%: <25% openness (gap fraction class 1); 25–50% openness (gap fraction class 2); 50–75% (gap fraction class 3); and >75% openness (gap fraction class 4) (modified from Bellow and Nair (2003)). Therefore, our estimates may be regarded as conservative.

4.2.6.3. Erosion regulation

In general, vegetation cover reduces soil erosion (Zhongming et al., 2010), but near ground, layers such as grass and litter contribute more than the tree canopy layer to soil conservation (Zhang et al., 2006; Zheng et al., 2008). Therefore, Zhongming et al. (2010) developed a *stratified vegetation cover index* (Cs) (Eq. 3), accommodating the contribution of each layer of forest in soil conservation. We used *stratified vegetation cover index* following Zhongming et al. (2010) to estimate erosion regulation ecosystem service:

$$C_s = \sum_{i=1}^{i} a_i C_i, \qquad (3)$$

where a_i is the weighting coefficient of layer i for its contribution to soil conservation; i is the number of layers or the strata in the vegetation community; and C_i is the measured coverage of the vertical layer i. The a_i value for tree, shrub, grass, and litter layers are 0.060814, 0.54616, 0.281468, and 0.111559, respectively (details in (Zhongming et al., 2010)).

4.2.6.4. Nutrient regulation

In the long term, woody debris is an important source of nutrient return in forested ecosystems (Clark et al., 2002). This is not yet well managed because canopy litter cycling is prioritized for its short-term rapid nutrient return. Nitrogen (N) turnover in forested ecosystems is an indicator of nutrient regulation ecosystem service (Burkhard et al., 2012; de Groot et al., 2010). N turnover in the forested ecosystems, through woody debris breakdown, is a slow but important process for N regulation in the forested ecosystem long term. Therefore, we estimated N turnover in the forested ecosystems through woody debris (Clark et al., 2002) as an indicator of nutrient regulation ecosystem service. We estimated the volume of woody debris (V) by the line intercept method (Eq. 4) (Iwashita et al., 2013; van Wagner, 1968), and then N content in the woody debris (Eq. 5) (Clark et al., 2002; Iwashita et al., 2013):

$$V = \pi^2/8L \sum_{i=1}^{i} n_i d_i^2 \qquad (4)$$

$$N = V*0.422*0.003, \qquad (5)$$

where V is the volume ($m^3\ ha^{-1}$), L is the transect length (m), n_i is the number of logs in the i^{th} diameter class, d_i is the notional diameter (cm) of the i^{th} size class (= lower limit of class + 1/3 the class range, to account for the right skewed distribution of log sizes), and N is the nitrogen content in the fallen woody debris ($Mg\ ha^{-1}$). We used both reported mean wood density ($0.422\ g\ cm^{-3}$) and N content (0.003 or 0.30%) of fallen woody debris from tropical forests as developed by Clark et al. (2002).

4.2.6.5. Cyclone protection

The current literature acknowledges that more complex forests are more resistant to tropical cyclones, and that unevenness of the forest structure is one of the indicators of forest complexity (Chapman et al., 2008; Foster and Boose, 1992; Hook et al., 1991; Lugo, 2008; Turton, 2008, 2012; Xi, 2015). The coefficient of variance (CV) of dbh includes both standard deviation and mean, and therefore provides a useful estimation of unevenness of dbh (a forest structural attribute). Hence, it is considered as an important indicator of forest complexity. Therefore, we calculated CV of tree dbh to assess the cyclone protection ecosystem service.

4.2.6.6. Habitat provision

Assessing the capacity of the habitat provision ecosystem service using one indicator is difficult because of the variation in habitat requirements for different wildlife species (McElhinny et al., 2005; Neumann and Starlinger, 2001; Pasher and King, 2011). Watson et al. (2001) proposed a habitat complexity scoring method to evaluate the capacity of forested ecosystems to provide habitat for birds, while Barnett et al. (1978) used another scoring system to assess the capacity of forests to provide habitat for small mammals. Both of these methods require very few attributes of forests, including canopy cover, shrub cover, ground cover, logs, and litter cover on forest floor. For the assessment of habitat provision, it is worth considering other attributes such as CV of dbh, recruitment of seedlings and saplings, and dead standing trees (Castaño-Villa et al., 2014; Pasher and King, 2011). Recently, Ochoa-Gaona et al. (2010) have developed a multicriteria index to evaluate tropical forest condition, while Pasher and King (2011) have developed another forest structural complexity index. Both indices accounted for a wide range of forest attributes. As a holistic approach to evaluate the capacity of forests of the region to supply habitat provision ecosystem service, we used a modified version (for calculation details, see Table 4.2) of a multicriteria index (Ochoa-Gaona et al., 2010), based on relevant indices (Barnett et al., 1978; Khanaposhtani et al., 2012; Ochoa-Gaona et al., 2010; Parkes et al., 2003; Pasher and King, 2011; Watson et al., 2001) and matching it with our local conditions.

Table 4.2. Calculation of the multi-criteria index to assess habitat provision ecosystem service of tropical forest.

Habitat gradient	Ecological attribute	Score					Sub-index score	Total score
		Values of 5 levels						
		0	0.25	0.5	0.75	1		
Canopy (C)	Tree height (m)	< 5	5-10	10-15	15-20	≥ 20		
	Canopy cover (%)	< 20	20-40	40-60	60-80	≥ 80	∑C	
		Values of 4 levels						
		0	0.33	0.66	1			
Intermediate (I)	No. of large trees*	0-4	5-8	9-12	≥ 13			
	Density (trees per 0.05ha)	≤ 49	50-81	82-113	≥ 114			
	Mean dbh (cm)	6-11	11-15	15-20	≥ 20			
	CV** of dbh	0.39-0.68	0.69-0.98	0.99-1.28	≥ 1.29			
	CV of snag dbh	< 0.30	0.31-0.60	0.61-0.90	≥ 0.91			
	No. of snags (per 0.05 ha)	0-4	5-8	9-12	13-16			**∑CIF**
	No. of seedlings /recruitments***	≤ 99	100-199	200-299	≥ 300		∑I	
		Values of 3 levels						
		0	0.5	1				
Forest floor (F)	Seedlings cover (%) (1m high)	<12	12-25	≥ 25				
	Litter cover (%)	< 30	30-60	≥ 60				
	Coarse woody debris cover (%)	< 9	9-18	≥ 18				
	Fine woody debris (no.)****	0-44	45-85	≥ 86				
	Coarse woody debris (no.)****	0-4	5-8	≥ 9			∑F	

* Trees with > 40cm dbh (per 0.05 ha), ** CV- Co-efficient of variance, *** no. per 0.015ha (1-2m high), **** Fine woody debris (< 10cm diameter) and coarse woody debris (≥ 10cm diameter) were counted along the 50m transect line.

4.2.6.7. Energy provision and timber provision

We estimated AGB following Chave et al. (2005) as an indicator of energy provision ecosystem service (see Section 4.2.6.1 for calculation details). Tree basal area (BA) was estimated as an indicator of timber provision services from the expression $BA = \pi (dbh)^2/4$, where dbh denotes diameter at breast height.

4.2.7. Statistical analyses

In order to uncover differences, interactions, and patterns of multiple ecosystem services in the rainforests, sclerophyll forests, and rehabilitated plantation forests, we used a combination of mean comparisons, multivariate analysis, and cluster analysis suitable for working with multiple ecosystem services. We conducted post hoc

Bonferroni tests (α = 0.05) to compare the mean values of each ecosystem service supply across the three forest types. We used the *Shapiro–Wilk* test to check for normality (Shapiro and Wilk, 1965). Ecosystem service values were log (natural) transformed, which required fulfilling the assumption of the *Bonferroni test*. In order to examine the relative abundance of each ecosystem service and the interaction of multiple ecosystem services, we conducted a nonmetric multidimensional scaling (NMS) ordination analysis. In the NMS ordination analysis, we used *Relative Sorensen* and *Varimax.* In the case of a single ecosystem service, the NMS technique plot provided information on the relative location of each sampling point, considering all ecosystem services, and the size of the sampling points represents the relative abundance of a particular ecosystem service. In the case of multiple ecosystem services, the NMS technique plot found the relative location of each sampling point, considering all ecosystem services, and represented each ecosystem service as a vector considering the correlation with both axes. We correlated both axes and derived r^2 from that correlation. In order to examine the patterns, we used two-way cluster analysis dendrograms using *Sorensen* and *Flexible Beta (-0.25).* This technique allowed us to classify synergy and antagonistic groups along the plots and ecosystem services. All statistical analyses were conducted using IBM SPSS 20 and PC-ORD6 (multivariate analysis software).

4.3. Results

4.3.1. Ecosystem service provision from rainforest, sclerophyll forest, and rehabilitated plantation forest

We found that the supply of most of the examined ecosystem services differed significantly among the three forest types, while erosion regulation service was found to be consistent among forest types ($F = 2.781$, $df = 2$, $p = 0.070$). The significantly different ecosystem services were global climate regulation ($F = 21.97$, $df = 2$, $p < 0.001$), air quality regulation ($F = 23.1$, $df = 2$, $p < 0.001$), nutrient regulation ($F= 9.39$, $df = 2$, $p < 0.001$), cyclone protection ($F = 6.41$, $df = 2$, $p = 0.003$), habitat provision ($F = 18.42$, $df = 2$, $p < 0.001$), energy provision ($F = 21.97$, $df =2$, $p < 0.001$), and timber provision ($F = 39.43$, $df =2$, $p < 0.001$).

In the multiple comparisons, our study revealed that rainforests provided significantly more ecosystem services than sclerophyll forests: global climate regulation (3.07 vs. 2.64, *Bonferroni test*, $p < 0.001$), cyclone protection (1.01 vs. 0.80; *Bonferroni test*, $p = 0.002$), habitat provision (5.52 vs. 3.28; *Bonferroni test*, $p < 0.001$), and energy provision (2.84 vs. 2.4; *Bonferroni test*, $p < 0.001$). However, these ecosystem services in the rehabilitated plantation forests were not significantly different from the other two forest types (Fig. 4.3). The supply of air quality regulation ecosystem service from rainforests and rehabilitated plantation forest was significantly higher than sclerophyll forests (*Bonferroni test*, $p < 0.001$, $p = 0.009$ respectively). The supply of timber provision ecosystem service was found to be significantly different in each of the three forest types (*Bonferroni test*, $p < 0.01$). However, nutrient regulation ecosystem service was found to be significantly higher in rainforests than sclerophyll forests (2 vs. 1.5; *Bonferroni test*, $p = 0.003$) and rehabilitated plantation forests (2 vs. 1.16; *Bonferroni test*, $p = 0.002$) (Fig. 4.3).

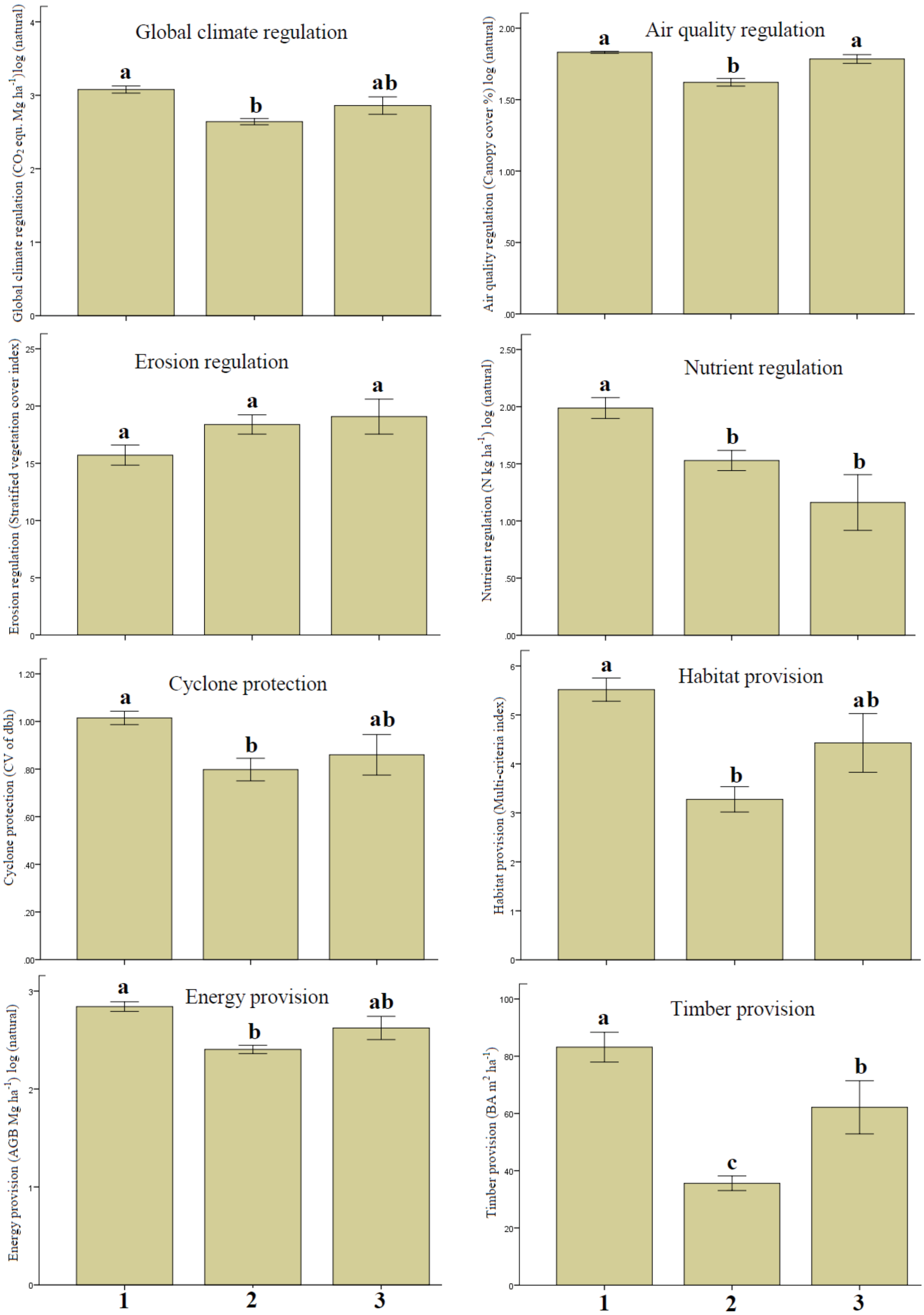

Figure. 4.3. Multiple ecosystem services provision in the rainforest (1), sclerophyll forest (2), and rehabilitated plantation forest (3) (mean ± 1 SE). Means sharing the same letters are not significantly different (α = 0.05; *Bonferroni test*). Data were log (natural) transformed, where necessary to perform *Bonferroni test*.

4.3.2. Pattern of multiple ecosystem services in three forest types

The abundance of all eight ecosystem services varied in the plots of the same forest type and among forest types (Fig. 4.4).

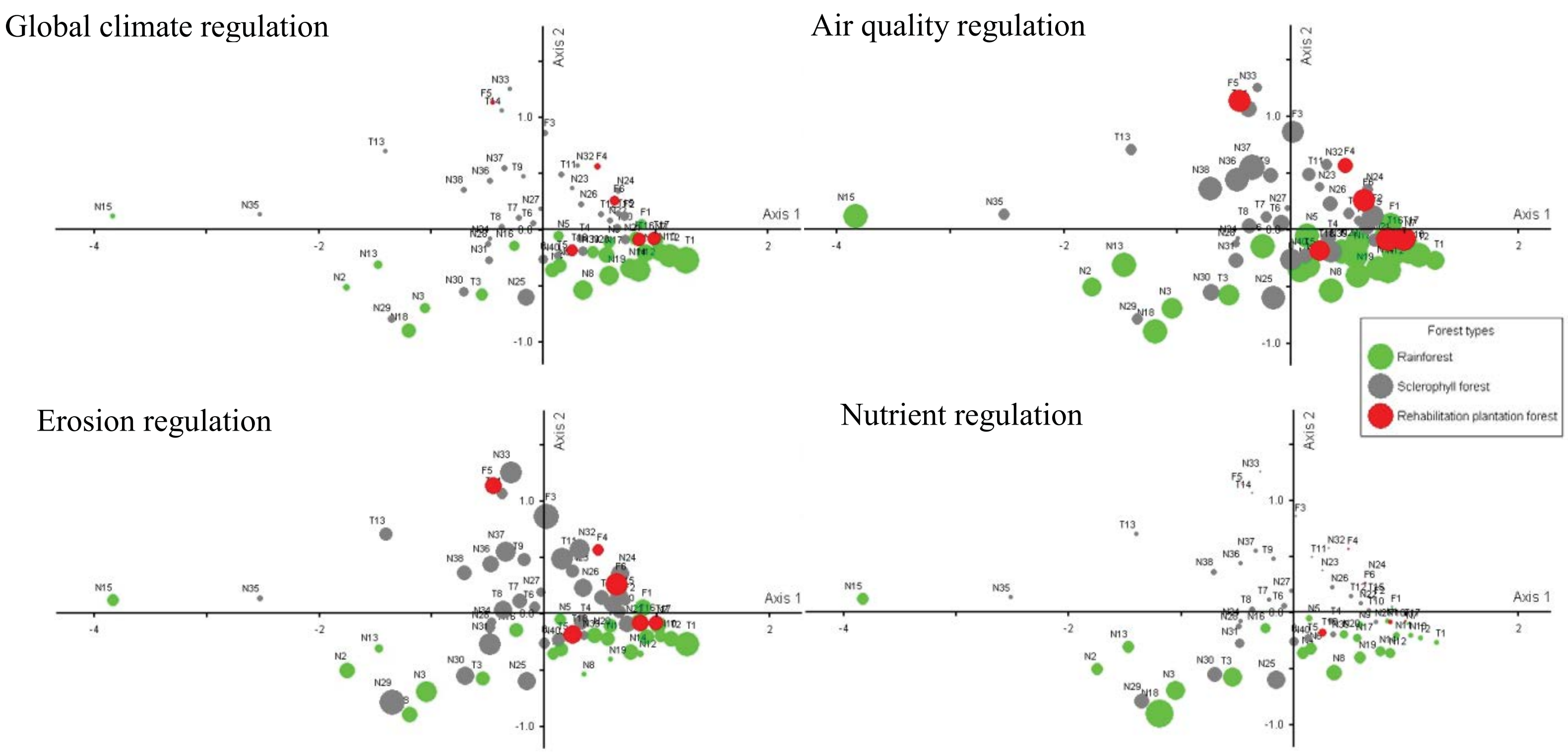

Figure 4.4. Nonmetric multidimensional scaling (NMS) ordination displaying abundance of eight ecosystem services in the plots. Size of the circle indicates the relative abundance of respective ecosystem service and the position of the circle represents relative space of the plots in the ordination. Axes 1 and 2 represent multiple ecosystem services. The combination of letters and numbers indicate only plot coding.

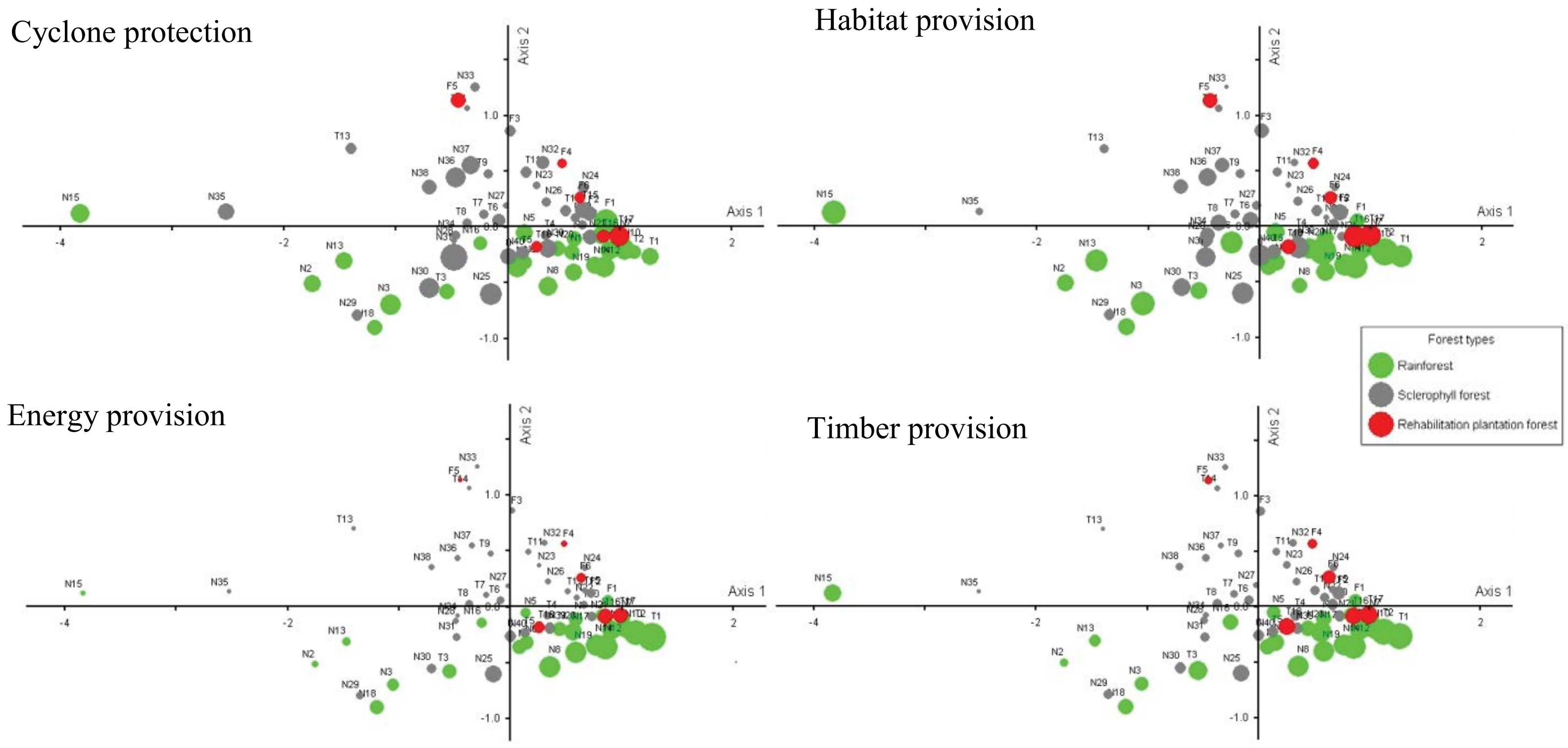

Figure 4.4. Continued.

Most of the rainforest plots were more abundant than sclerophyll forests and rehabilitated plantation forest plots in the supply of global climate regulation, nutrient regulation, energy provision, and timber provision ecosystem services (Fig 4.4). The abundance of air quality regulation, erosion regulation, cyclone protection, and habitat provision ecosystem services from most of the sclerophyll forest plots and rehabilitated plantation forest plots were comparable to the rainforest plots (Fig 4.4)

4.3.3. Interaction of multiple ecosystem services in the three forest types

The NMS explained 99.6% variation of multiple ecosystem services in the three forest types (Fig. 4.5). X-axis (axis 1) explained 89.6% of the variation showing a gradient of higher values (positive scores of NMS) of global climate regulation ($r = 0.511$), habitat provision ($r = 0.06$), erosion regulation ($r = 0.116$), air quality regulation ($r = 0.124$), energy provision ($r = 0.511$), and timber provision ($r = 0.395$) to a gradient of lower values (negative scores of NMS) in one ordination space. By contrast, y-axis explained 10.1% of the variation showing a gradient of higher values (negative score of NMS) of nutrient regulation ($r = -0.418$) and cyclone protection ($r = -0.022$) to a gradient of lower values in another ordination space. Among the eight ecosystem services, global climate regulation service emerged as having the capacity to potentially increase the provision of other examined ecosystem services; however, nutrient regulation service emerged as having capacity to potentially reduce the provision of the other examined ecosystem services in the plots across the three forest types (Fig. 4.5). The NMS showed that plots located within the rainforests and sclerophyll forests were more similar among themselves than plots between forest types; however, a wide variation in the multiple ecosystem service provision can be seen within each three forest-type plots. In the NMS, most of the plots from rainforests occupied one ordination space, whereas most of the plots from sclerophyll forests existed in another extreme ordination space. However, plots from rehabilitated plantation forests shared the same ordination space with rainforests and sclerophyll forests. Seven of the eight ecosystem services shared the same ordination space with rainforests and rehabilitated plantation forests (global climate regulation, air quality regulation, nutrient regulation, cyclone protection, habitat provision, energy provision, and timber provision) however, erosion regulation, shared ordination space with sclerophyll forests (Fig. 4.5).

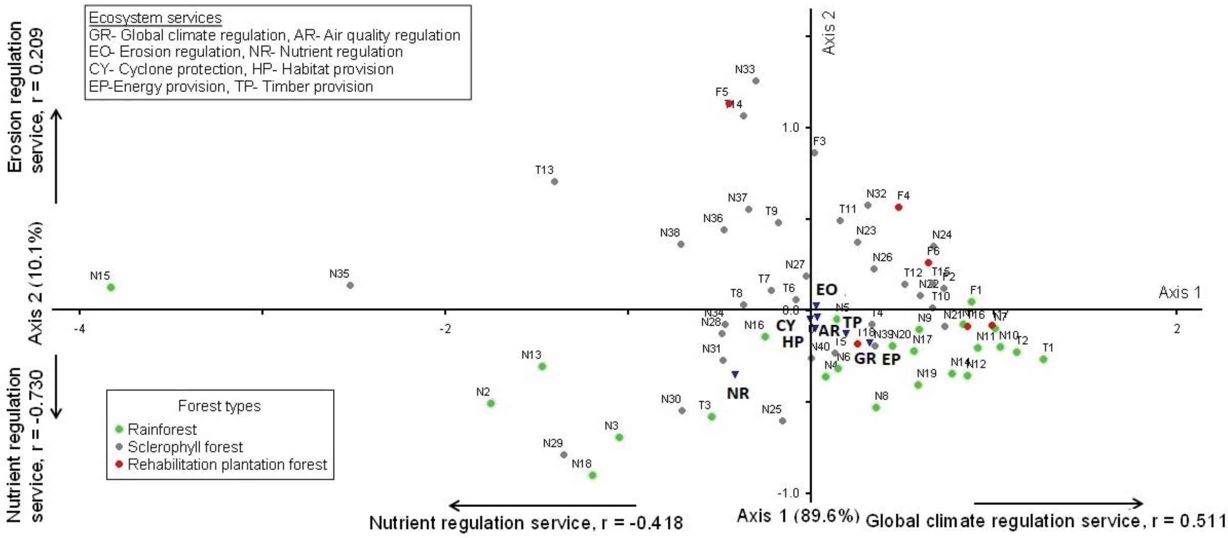

Figure 4.5. Nonmetric multidimensional scaling (NMS) ordination showing the variability of multiple ecosystem services provisioning within and between plots in the three forest types examined. X-axis (axis 1) represents 89.6% (r^2) and y-axis (axis 2) 10.1% (r^2) of the variation. The stress value is 46.41. The combination of letters and numbers indicate only plot coding.

4.3.4. Clustering of multiple ecosystem services in the three forest types

On the basis of the two-way cluster analysis along plots, four groups of plots were identified (Fig. 4.6). The first group was composed of most of the rainforest plots, four rehabilitated plantation forest plots, and eight sclerophyll forest plots, where provision of all eight ecosystem services was high. The second group consisted of mostly sclerophyll forest plots, two rainforest plots, and one rehabilitated plantation forest plot. This group was displaying a low provision of most of the ecosystem services, but high provision of erosion regulation service. The third group was composed mostly of the sclerophyll forest plots, one rainforest plot, and one rehabilitated plantation forest plot. This group showed low provision of all eight ecosystem services. The fourth group consisted of mostly rainforest plots and one sclerophyll forest plot. This group showed lower provision of the erosion regulation service, but high provision of the remaining seven ecosystem services. At the highest level of distance (Fig 4.6), two distinct clusters of plots were found: the first cluster (groups 1 and 4) represented most of the rainforest plots, rehabilitated plantation forest plots, and few sclerophyll forest plots with high provision of most of the ecosystem services. The second cluster (groups 2 and 3) included most of the sclerophyll forest plots, few rehabilitated plantation forest plots, and a few rainforest plots with low provision of most of the ecosystem services (Fig. 4.6).

On the basis of a two-way cluster analysis along multiple ecosystem services, three groups of ecosystem services were identified. The first group consisted of habitat provision, cyclone protection, air quality regulation, and erosion regulation ecosystem services. The second group was composed of global climate regulation, energy provision, and timber provision. The third group contained only one ecosystem service, that is, nutrient regulation. Two distinct clusters were evident at the highest level of distance (Fig 4.6). The first cluster represented seven ecosystem services (groups 1 and 2): global climate regulation, air quality regulation, erosion regulation, cyclone protection, habitat provision, and energy provision. The second cluster (group 3) was represented by only one ecosystem service, that is, nutrient regulation (Fig. 4.6).

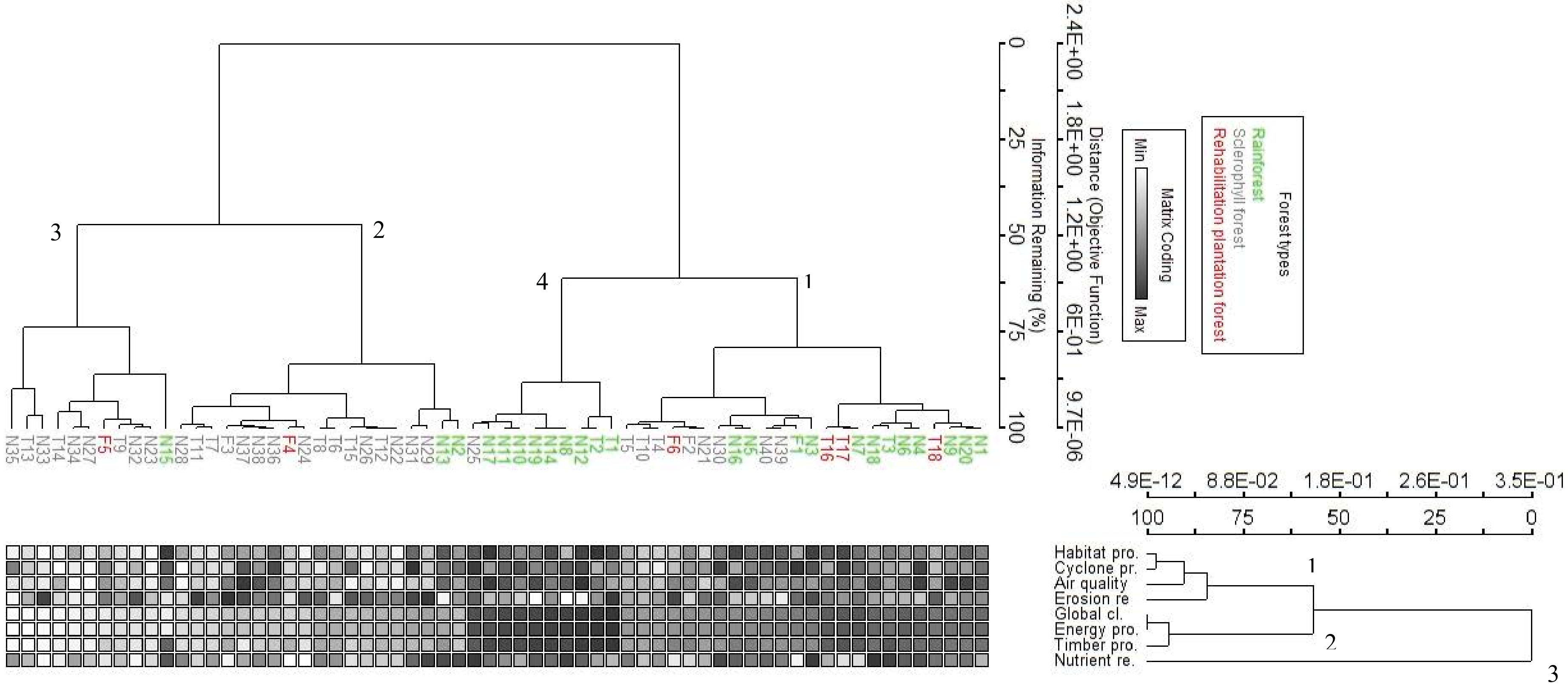

Figure 4.6. A two way cluster analysis dendrogram of plots of three forest types with eight ecosystem services in the Wet Tropics bioregion of Northern Queensland, Australia. The combination of letters and numbers indicate only plot coding.

4.4. Discussion

We found that, on average, most of the examined ecosystem services are significantly higher in rainforests than sclerophyll forests. However, a mixed trend was apparent when we compared rehabilitated plantation forests with rainforests and sclerophyll forests (Fig 4.3). The sharing of ordination space between seven out of eight ecosystem services examined in rainforest plots (Fig 4.5) also showed that rainforests have a higher capacity of providing multiple ecosystem services. These findings are consistent with those of the previous studies, which reported a higher capacity of tropical rainforests than other forest types to supply multiple ecosystem services (Clark and Clark, 2000; Mutoko et al., 2015; Tian et al., 1998). At the plot level, the high provision of some ecosystem services (e.g., air quality regulation) (Figs. 4.4 and 4.6) by rehabilitated plantation forests such as rainforests suggests their capacity to recover and provide a sustainable supply of ecosystem services. Hence, even degraded forests remain important for the supply of certain ecosystem services. In the ordination analysis (Fig. 4.5), the sharing of ordination space between rainforest plots and rehabilitated forest plots also signifies that rehabilitated plantation forests may provide multiple ecosystem services nearly equivalent to natural rainforests; however, the level of degradation might be considered. The studied rehabilitated rainforests were reasonably old (~19 years), reestablished with local native rainforest species, focusing on ecological connectivity objectives, with a significant number of large trees retained after previous degradation of the area. All these factors may have acted together to recover the capacity of the degraded forests after rehabilitation for supply of higher multiple ecosystem services. Our findings are also supported by the findings of Kanowski and Catterall (2010), who reported a higher density of trees and a significant number of large trees in the rehabilitated plantation forests of this region. These results are also consistent with the findings of Edwards et al. (2010) and Edwards et al. (2014), who showed that even degraded tropical rainforest may supply valuable ecosystem services as they can retain ecosystem functioning. In addition to rehabilitation, the recovery of the ecosystem services capacity by the degraded forests may depend on the ecological threshold (Andersen et al., 2009; Scheffer and Carpenter, 2003). In some instances, ecosystems may lose the capacity to supply certain ecosystem services permanently, due to the tremendous ecological pressure and excessive use of ecosystem services (Scheffer and Carpenter, 2003; Villamagna et al., 2013)

At the plot level, wide variations in the provision of all eight ecosystem services were found in plots of the same forest type (Figs. 4.4 and 4.6). This might be due to the differences in the scale of disturbances – such as tropical cyclones – across all forest types in the landscape (Turton, 2008, 2012). For example, more disturbances were recorded in the plots of coastal areas than upland areas of the same forest type, with very fine scale differences in the vegetation composition and structure between the plots of the same forest type (Ash, 1988; Stork and Turton, 2008). This may be due to controlled fire (burns) in the sclerophyll forests (Bowman, 2000; Fensham et al., 2003), which is one of the main determinants of the recruitment of the sclerophyll forest in this region.

Nutrient regulation and global climate regulation ecosystem services appeared to have had a substantial opposite effect with varying scale in the provision of the other six ecosystem services across the three forest types (Fig. 4.5). The supply of more nutrient regulation ecosystem service may result in the reduction of the supply of remaining seven ecosystem services in the three forest types. A possible reason for this may be that nutrient regulation ecosystem service is more closely related to the fallen logs in the forest floor (details in Section 4.2.6.4.). Evidently, more fallen logs on the forest floor indicates less standing biomass, less canopy cover, and more forest disturbances (considered as a proxy), which are likely to reduce the capacity of the forest to provide multiple ecosystem services. By contrast, the supply of more global climate regulation ecosystem service may increase the supply of the remaining examined ecosystem services such as habitat provision, air quality regulation, erosion regulation, and energy provision. This is due to the supply of more global climate regulation ecosystem service being related to more dense forest covers together with more large trees, which is likely to increase the capacity of the forest to supply multiple ecosystem services. These results are comparable to those of Slik et al. (2013), who have reported the important role of larger trees in forests to supply global climate regulation service, and Lewis et al. (2009) and Liddell et al. (2007), who have reported the influence of dense forest cover to supply multiple ecosystem services. In this study, we evaluated the interaction among ecosystem services supply based on vegetation attribute data. Without the interaction among ecosystem services, the supply of multiple ecosystem services is also controlled by the drivers in the forest type (e.g., environmental variable, management, and policy decision) (Bennett et al., 2009). Therefore, together with the interactions among the ecosystem services presented in this study, drivers need to be considered in the decision-making processes.

We found that the supplies of eight examined ecosystem services were divided into distinct groups of dissimilarity (three groups of lower dissimilarity and two groups of higher dissimilarity) (Fig. 4.6). Ecosystem services in one group were closely linked together (synergies) than the separate group (trade-off). Therefore synergies and trade-offs exist in the supply of multiple ecosystem services. The identified synergies and trade-off groups (Fig 4.6) of multiple ecosystem services are comparable with the ecosystem service groups identified from other regions (Baral et al., 2013b; de Groot et al., 2010; Foley et al., 2005; Galicia and Zarco-Arista, 2014). As groups 1 and 2 of multiple ecosystem services supply (Fig 4.6) are synergies, we argue that management interventions targeting the enhancement of supply of at least one ecosystem service from each of the first two groups, such as habitat provision from group 1 and global climate regulation from group 2, will sustain the supply of other examined ecosystem services across the forest types in the landscape. By contrast, as trade-offs are apparent between group 3 and groups 1 and 2, any intervention that will enhance the nutrient regulation service may reduce the supply of other examined ecosystem services in the landscape. We analyzed the trade-offs and synergies in regard to the supply of ecosystem services, and consequently the identified trade-off and synergies groups in this study describe the interactions and patterns of distribution in the supply of ecosystem services. However, it is noted that this may not appropriately describe the use of ecosystem services. Furthermore, potential conflict may arise between the ecosystem services of the same groups (both in synergies and trade-off groups) if the study is arbitrated considering the use of ecosystem services. Synergies and trade-offs among ecosystem services are useful to maintain ecosystem services sustainability in the landscape (Nelson et al., 2009; Raudsepp-Hearne et al., 2010). The synergies and trade-offs among ecosystem services may occur due to interactions and/or drivers (Bennett et al., 2009; Villamagna et al., 2013). We used indicators that were derived from vegetation attributes, and in the quantification process of each indicator, there was an overlap of a few components of vegetation attributes (details in Sections 4.2.4 and

4.2.6), which were likely to have an impact on synergies and trade-offs among ecosystem services.

4.5. Conclusions

Ecosystem services capacity across forest types in a tropical landscape varies and the supply of multiple ecosystem services from rainforests is higher than that in the other two examined forest types. In order to sustain the supply of multiple ecosystem services from the heterogeneous forested landscape of the Wet Tropics bioregion, it is optimal to primarily conserve the rainforests and sclerophyll forests. Rehabilitation of the degraded rainforests may play a considerable supplementary role to sustain the supply of multiple ecosystem services in the long term, if forest restoration effort aims to increase global climate regulation and habitat provision ecosystem services, noting that their synergies have positively affected the other examined ecosystem services.

In addition, in rainforests and sclerophyll forests, where the current supplies of examined ecosystem services are less (except nutrient regulation), management intervention (e.g., ensuring adequate native tree species selection) to sustain the supply of habitat provision and global climate regulation ecosystem services will be useful to sustain the supply of examined multiple ecosystem services.

It is important to consider both social and ecological systems to understand the interaction among ecosystem services (Bennett et al., 2009). As we collected field-based ecological data, we only considered the ecological interface. Future research integrating both social and ecological systems will gain deeper insights into interactions among multiple ecosystem services.

Acknowledgments

The authors acknowledge the partial funding for this research project from Skyrail Rainforest Foundation, Australia; Wet Tropics Management Authority, Australia; and the Graduate Research Scheme of James Cook University, Australia. Their gratitude further extends to volunteers Edison Salas Castelo, Dale Bennett, Darren

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OVERVIEW OF CHAPTER 5

This chapter is based on a paper under review in the journal *Scientific Reports*[7] with minimal formatting changes. I address the fourth objective of the thesis - to compare the carbon storage among rainforests, degraded rainforests and sclerophyll forests, and to determine the drivers of carbon storage in the tropical forested landscape.

In this chapter, I investigate the biomass and carbon storage of rainforests, degraded rainforests and sclerophyll forests in the Wet Tropics bioregion. I also examine the influence of forest structural features and disturbances upon forest biomass and carbon storage. Finally, I discuss mechanisms whereby appropriate site selection and management of degraded forests may allow for potential policy interventions which enhance carbon storage in the tropical forests of this landscape, and additionally may aid biodiversity conservation.

[7] **Alamgir, M.**, Campbell, M.J. Turton, S.M., Pert, P.L., Edwards, W., Laurance, W.F. (in review, first revision submitted). Degraded tropical rainforests possess valuable carbon storage opportunities in a complex, forested landscape. *Scientific Reports*

CHAPTER 5

DEGRADED TROPICAL RAINFORESTS POSSESS VALUABLE CARBON STORAGE OPPORTUNITIES IN A COMPLEX, FORESTED LANDSCAPE

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Wet Tropics rainforest a global carbon pool (Photo credit: Mohammed Alamgir)

ABSTRACT

Tropical forests are major contributors to the terrestrial global carbon pool, but this pool is being reduced via deforestation and forest degradation. Relatively few studies have assessed carbon storage in degraded forests. We surveyed 74 plots covering 37,000 m^2 in total of intact rainforests, degraded rainforests and sclerophyll forests across the greater Wet Tropics bioregion of northeast Australia. We compared aboveground biomass and carbon storage of the three forest types, and the effects of forest structural attributes and environmental factors that influence carbon storage. Some degraded forests were found to store much less aboveground carbon than intact rainforests, whereas others sites had similar carbon storage to primary forest. Sclerophyll forests had lower carbon storage, comparable to the most heavily degraded rainforests. Our findings indicate that, under certain situations, degraded forest may store as much carbon as intact rainforests. Strategic rehabilitation of degraded forests could enhance regional carbon storage and have positive benefits for tropical biodiversity.

5.1. Introduction

Although tropical forests only cover ~6% of the global land surface, they are the largest single repository of above-ground biomass carbon (ABC) stores (Laurance, 2008; Lewis, 2006; Saatchi et al., 2011), containing ~195 petagrams of carbon (PgC) (Liu et al., 2015; Pan et al., 2011). In addition to their total ABC storage, tropical forests are also net carbon sinks (Baker et al., 2005; Lewis et al., 2009; Pan et al., 2011). As a consequence of their significant carbon storage and sink capacity, tropical forests play a critical role in climate change mitigation (Carrasco and Papworth, 2014; Fahey et al., 2009). However, despite this valuable carbon storage and potential climate change mitigation capacity, tropical forests experience high levels of annual deforestation (Achard et al., 2002; Hansen et al., 2013), which has been estimated to have resulted in an annual, global ABC loss of 0.26 $PgCYr^{-1}$ (1993-2012; Liu et al., 2015). Moreover, deforestation of tropical forests is the second greatest contributor of green-house gas emissions to the atmosphere after the burning of fossil fuels (Laurance et al., 1998; Pütz et al., 2014).

In addition to deforestation, much of the remaining tropical forested area experiences various forms of degradation, with the area of degraded tropical forests now estimated to exceed 500 million hectares (Ghazoul et al., 2015). Moreover, it is further estimated that regenerating forests now exceed primary forests as the predominant form of tropical forest cover worldwide (Chazdon, 2014). Degraded tropical forests store less ABC than primary forests (Achard et al., 2004; Bustamante et al., 2015; Laurance et al., 1997) and thus forest degradation results in increased atmospheric CO_2 emissions (Achard et al., 2004; Asner et al., 2005; Bustamante et al., 2015). Alternatively, however, forest regrowth within degraded forests may remove large amounts of carbon from the atmosphere (Pan et al., 2011). As such, it is becoming crucial to determine the impact of forest degradation and regeneration on net CO_2 emissions and overall forest carbon storage capacity (Achard et al., 2004; Bustamante et al., 2015; Ioki et al., 2014; Magnago et al., 2015; Malhi and Grace, 2000).

One problem faced when determining the carbon storage capacity of degraded forest, and thus their net CO_2 emissions, is that landscape-scale factors often determine the impact of forest degradation on carbon storage within complex tropical forest landscapes. These factors include variability in the availability of constraining environmental resources (Cramer et al., 2004; Tian et al., 2000), differences in vegetation composition (Corlett and Primack, 2006; Glenday, 2006), forest structural variation (Usuga et al., 2010; Zhang et al., 2007), and forest management regime (Fahey et al., 2009; Turner et al., 2011). As a whole, it is still unclear what impact each of these factors has on the dynamics of carbon storage in complex tropical forested landscapes (Tian et al., 2000; Zuidema et al., 2013). As such, if the development of optimal management strategies to enhance landscape-scale carbon storage in complex tropical forested landscapes is to occur, determining the impact of individual landscape-scale factors on the carbon storage capacity of degraded forests is of utmost importance (Fearnside, 2004; Ioki et al., 2014).

The Wet Tropics bioregion of northeast Australia is a complex and contested landscape (Stork and Turton, 2008) which is primarily composed of one of the oldest rainforests on earth (Byrne et al., 2011). This region has been described as the second most irreplaceable natural world heritage area (Bertzky et al., 2013), and the sixth most irreplaceable protected area on the planet (Le Saout et al., 2013). The significant biodiversity values of the Wet Tropics bioregion are well documented (Hilbert et al., 2001; UNESCO, 1988; Williams et al., 2003). However, the carbon storage values of the component vegetation types within the Wet Tropics bioregion is yet to be determined ((precluding carbon storage within restoration plantings; (Preece et al., 2012)).

Here we evaluate the biomass and carbon storage of intact closed-canopy forests (hereafter termed rainforest), degraded rainforests and sclerophyll forests within a complex and heterogeneous landscape of the Wet Tropics bioregion. In addition, we examine the influence of forest structural features (e.g. tree size) and disturbance upon forest biomass and carbon storage. We also compare the impact of rainfall and elevational gradients upon the carbon storage of these vegetation types. Finally, we discuss mechanisms whereby appropriate site selection and management of degraded forests may allow for potential policy interventions which enhance carbon storage in the tropical forests of this landscape, and additionally aid biodiversity conservation.

5.2. Materials and methods

5.2.1. The study area

Our study was conducted in the Wet Tropics bioregion northeast Australia (Fig. 5.1). The total area of the bioregion is ca. two million ha (Goosem, 2002), most of which experiences a seasonally wet tropical climate. The total mean annual rainfall ranges from 1200 mm to 4000 mm (although the highest mountain peaks may receive 8000 mm yr^{-1}) and the mean annual temperature ranges from 17°C to 31°C (Goosem, 2002). The elevation of the bioregion ranges from a few metres above mean sea level (msl) to ~ 1000 m although the highest peak within the region is 1622 m. The heterogeneous and complex landscape of the region is dominated by intergrading rainforests and sclerophyll forests with environmentally defined boundaries (Ash, 1988). Approximately, 45% (894,420 ha) (mainly rainforest) of the Wet Tropics bioregion was inscribed on the World Heritage list in 1988 as a property that fulfilled all four natural criteria for listing (Goosem, 2002; UNESCO, 1988).

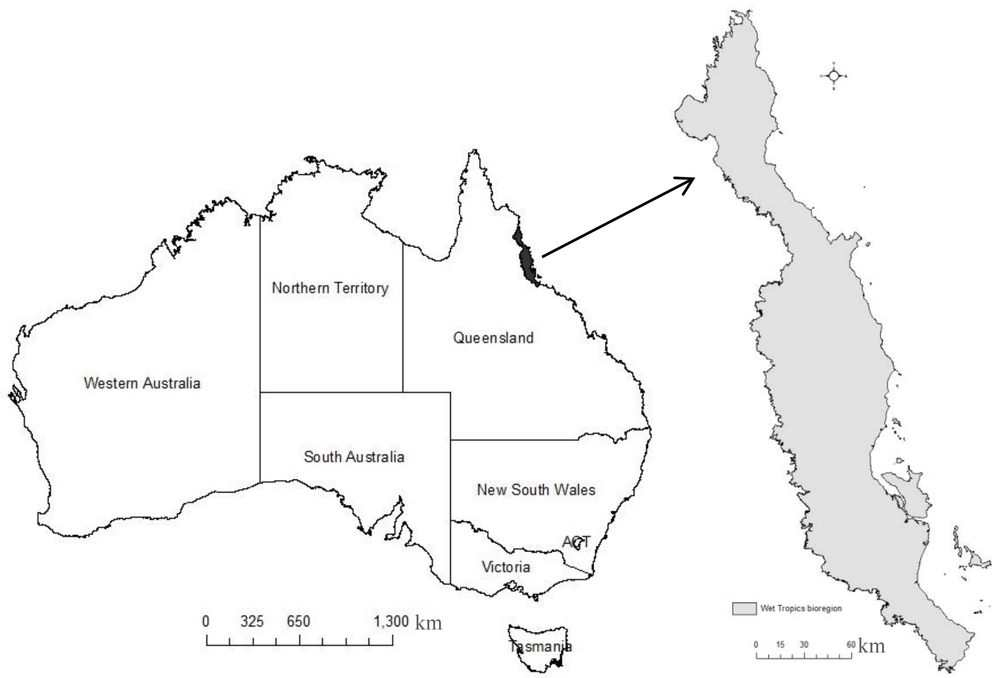

Figure 5.1. The study area- Wet Tropics bioregion, Queensland Australia. The maps were created using Esri ArcMap 10.2. (http://www.arcgis.com).

5.2.2. Data collection

We sampled a total of 74 plots- 29 rainforest, 32 sclerophyll forest and 13 degraded forest sites- over an area of 37,000 m^2 from the Wet Tropics bioregion, Australia. We defined degraded forest as rainforest that was in the process of being rehabilitated after extensive disturbance (e.g. logging or clearing) or were fragmented. Within each site we collected data from a plot of 0.05 ha (50 m × 10 m transect). The sampling points were predetermined prior to field data collection to a) avoid creeks and water bodies; and b) minimize edge effects by maintaining at least 20 m distance between our plots and other land uses. For our data collection we used a transect method which has previously been determined suitable for estimating high densities of trees in rainforests (Preece et al., 2012) and sclerophyll forests (Burrows et al., 2002). We measured the diameter at breast height (dbh) of all trees ≥ 10 cm within our plots to the nearest millimeter. In each of the plots we also counted the number of fallen logs ≥ 10 cm diameter on the forest floor (≤ 1 m above ground level). All the sites were georeferenced and general environmental and landscape features such as slope and elevation recorded. To account for the high diversity of mean annual rainfall and elevation within the Wet Tropics bioregion, we collected data from sites ranging from a few metres above mean sea level (msl) to more than 1000 m, and from sites receiving less than 1000 mm to more than 3500 mm mean annual rainfall. Mean annual rainfall data was determined using long-term records for the region provided by the Wet Tropics Management Authority, Queensland, Australia (WTMA, 2009b).

5.2.3. Estimation of above ground biomass carbon

Preece et al. (2012) compared the accuracy of biomass estimation methods for forests within the Wet Tropics bioregion and concluded that the Chave et al. (2005) allometric provided the best and most reliable estimate for the region. As such, we estimated above ground biomass (AGB) following Chave's allometric equation (Chave et al., 2005). To convert AGB into biomass carbon storage we used a conversion factor of 0.47 which is the recommended value from the Intergovernmental Panel for Climate Change for tropical forests (IPCC, 2006). In addition, wood density estimates were calculated using the reported default value for Australian tropical forests of 0.5 g cm^{-3} (500 kgm^{-3}) (Department of Climate Change and Energy Efficiency, 2010). Consequently, AGB was calculated using equation (1)

$$AGB = \rho \times exp(-1.499 + 2.148 ln(dbh) + 0.207 (ln(dbh))^2 - 0.0281 (ln(dbh))^3) \quad (1)$$

where AGB is measured in kg, dbh is measured in cm, and ρ is wood density measured in g cm^{-3}.

Above ground biomass estimates were then converted to carbon estimates using equation (2)

$$Carbon = AGB \times 0.47 \quad (2)$$

5.2.4. Statistical analyses

All statistical analyses were conducted in IBM SPSS 20, PCORD 6 and R (R Core Team, 2015). We used independent Kruskal-Wallis tests (2-tailed, $\alpha = 0.05$) to compare above ground biomass, above ground carbon storage and forest structural attributes (number of stems ≥ 10cm dbh, average tree dbh, fallen logs (≥ 10 cm diameter), and CV of tree dbh) between the rainforest, degraded forest and sclerophyll forest types. A NMS (Nonmetric Multidimensional Scaling) ordination was performed (in PCORD 6) to investigate the plot based variation in above ground carbon storage in relation to the other examined attributes of the examined forests. In the NMS ordination analysis we utilized *Sorensen* and *Orthogonal Principal Axis* (rotation).

The significant explanatory variables dictating the carbon storage of plots were determined using a binomial generalized linear model (GLM) with a logit link function, followed by a backwards, stepwise regression comparison in R. Prior to creating the global model and candidate model comparisons we performed data exploration and checked for (and removed) correlated predictor variables following the protocol of Zuur et al. (2010). We selected *a priori* a global model in which the carbon storage of plots was a function of: elevation (m), slope (degree), number of fallen logs (≥ 10cm diameter), canopy cover (%), mean annual rainfall (mm), tree diameter breast height (cm), tree abundance and forest type (rainforest, degraded forest or sclerophyll). The best model was then determined through a backwards stepwise model comparison whereby nested models were compared using the drop1 function and AIC model values, and the best model was that which contained only significant variables and the lowest AIC model value.

5.3. Results

5.3.1. Structural variation across the forest types

We counted and measured a total of 1438 trees in rainforests, 1193 trees in degraded forests and 693 trees in the sclerophyll forests. There was a significant difference in the number of trees ≥ 10 cm dbh between the examined forest types (χ^2 = 18.269, df = 2, $p_{2\text{-tailed}} < 0.001$), with a pairwise post hoc comparison, showing that rainforests (RF) had significantly more trees than degraded forest (DF) and sclerophyll forests (SF) (RF-DF $p_{2\text{-tailed}} = 0.013$; RF- SF $p_{2\text{-tailed}} < 0.001$) (Fig. 5.2). However, there was no significant difference in the number of trees between degraded forests and sclerophyll forests ($p_{2\text{-tailed}} = 1.00$) (Fig. 5.2). The average tree dbh (cm) was significantly different between the three forest types (χ^2 = 16.295, df = 2, $p_{2\text{-tailed}} < 0.001$), with rainforests and degraded forests possessing significantly larger trees than sclerophyll forests (RF- SF $p_{2\text{-tailed}} = 0.002$; DF- SF $p_{2\text{-tailed}} = 0.004$) (Fig. 5.2). Analogously, the number of fallen logs (≥ 10 cm diameter) per ha was significantly different between the forest types (χ^2 = 15.406, df = 2, $p_{2\text{-tailed}} <0.001$), with the post hoc pairwise comparison again finding that rainforests and degraded forests possessed more fallen logs than sclerophyll forests (RF- SF $p_{2\text{-tailed}} = 0.006$; DF- SF $p_{2\text{-tailed}} = 0.002$) (Fig. 5.2). Finally, the co-efficient of variation (CV) of tree dbh was also significantly different among the forest types (χ^2 = 13.689, df = 2, $p_{2\text{-tailed}} = 0.001$) which was again driven by the significantly larger values of CV of tree dbh in the rainforest and degraded forests compared to the sclerophyll forest (RF- SF $p_{2\text{-tailed}} < 0.001$; DF- SF $p_{2\text{-tailed}} = 0.046$) (Fig. 5.2).

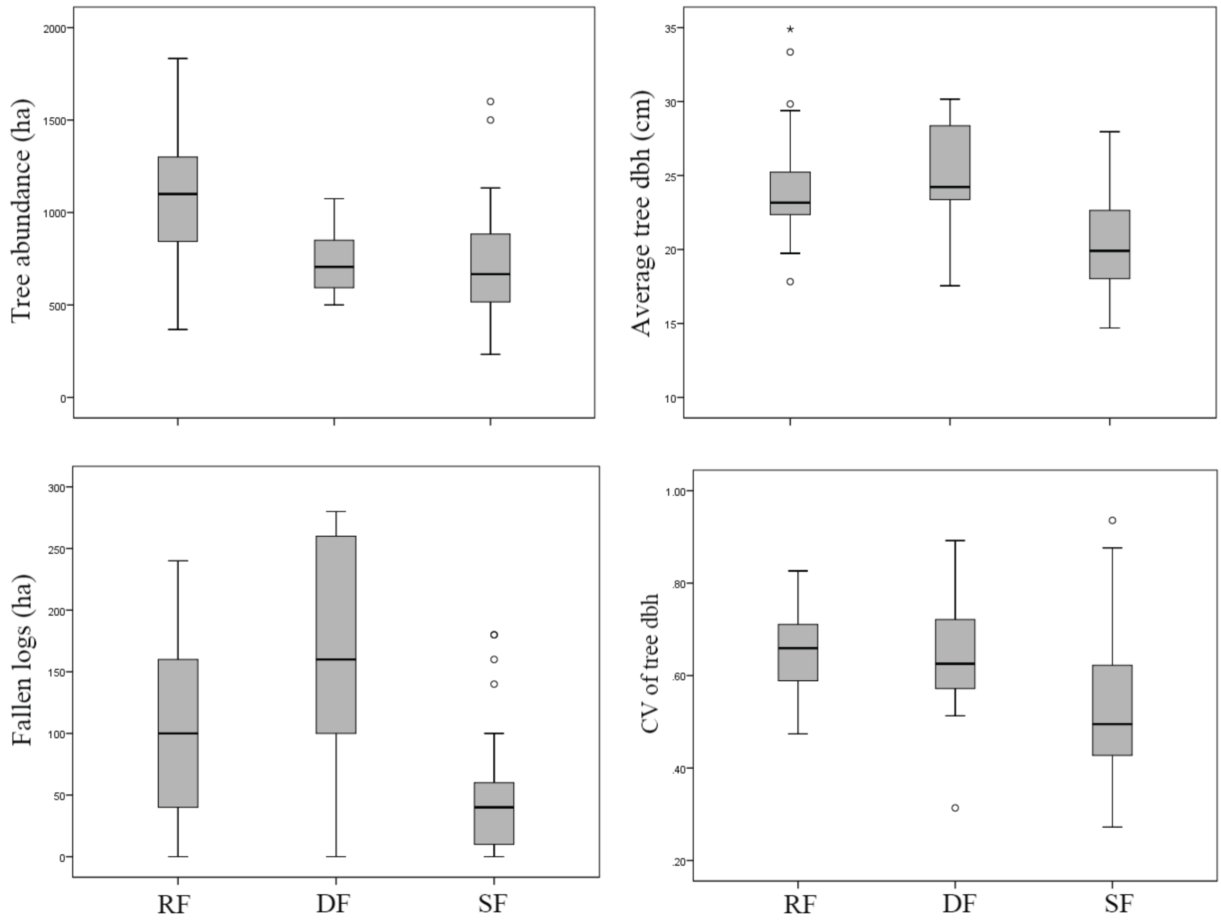

Figure 5.2. The average number of trees (≥ 10cm dbh) per ha, average tree diameter at breast height (cm), average number of fallen logs (≥ 10cm diameter) and average coefficient of variance (CV) of tree diameter at breast height (cm) in the examined plots of rainforest RF, n=29, degraded forest DF, n=13 and sclerophyll forest SF, n=32 within the Wet Tropics bioregion of northeast Australia.

5.3.2. Above ground biomass and carbon stock

There was a significant difference between the amount of above ground biomass (Mg ha^{-1}) and above ground carbon (Mg ha^{-1}) stored within the examined rainforests, degraded forests and sclerophyll forests ($\chi^2 = 33.064$, df = 2, $p_{2\text{-tailed}} < 0.001$ in each case). Using a pairwise post hoc comparison, we found that there was significant more above ground biomass (Mg ha^{-1}) and above ground carbon stored (Mg ha^{-1}) within the rainforests and the degraded forests than within the sclerophyll forest ($p_{2\text{-tailed}} < 0.001$ in each case) (RF- SF $p_{2\text{-tailed}} <0.001$; DF- SF $p_{2\text{-tailed}} = 0.009$). However, there was no significant difference in the amount of above ground biomass and above ground carbon stored in the rainforests and degraded forests (RF- DF $p_{2\text{-tailed}} = 0.442$) (Fig. 5.3).

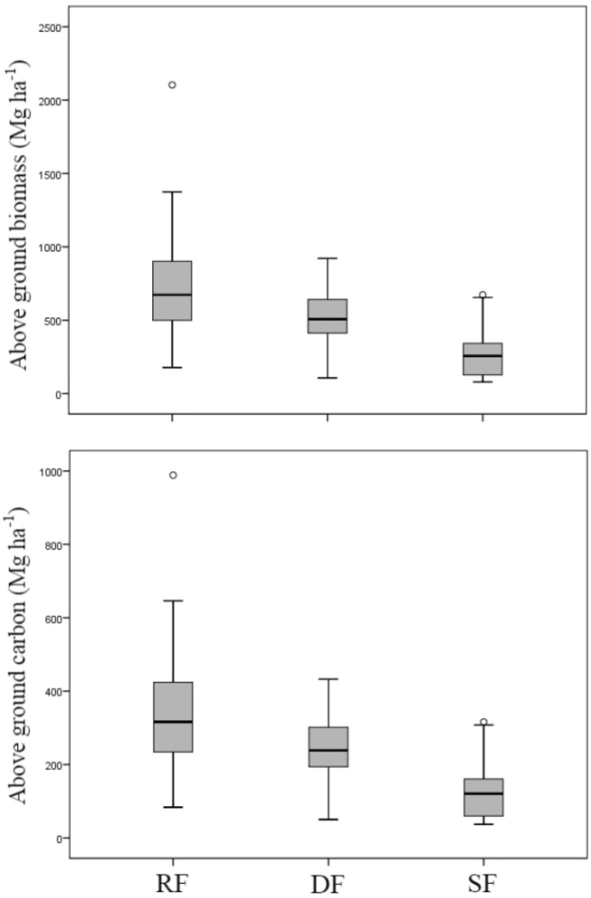

Figure 5.3. The above ground biomass (Mg ha^{-1}) and above ground carbon (Mg ha^{-1}) stored within the examined rainforest RF, n=29, degraded forest DF, n=13, and sclerophyll forest SF, n=32 vegetation types of the Wet Tropics bioregion of northeast Australia.

5.3.3. Variation and distribution in carbon storage of the examined vegetation types

The Nonmetric multidimensional scaling (NMS) ordination of above ground carbon storage of plots in variable space explained 96.3% cumulative variation in the examined data, where the x-axis (axis 1) represented 87.7% and y axis (axis 2) 8.6% of the variation (Fig 5.4). The x-axis showed a strong correlation with the number of trees and above ground carbon storage variables (r = 0.929, and 0.681 respectively) whilst the y axis showed a strong correlation with the average tree dbh and above ground carbon storage variables (r = -0.820, and -0.510 respectively) (Fig 5.4).

Most of the rainforest plots had higher levels of above ground carbon storage than those of sclerophyll forests (Fig. 5.4). The above ground carbon storage in the degraded forest plots varied from high levels similar to those of rainforests through to low levels similar to plots within sclerophyll forests (Fig. 5.4). As such, rainforest plots and sclerophyll forest plots occupied somewhat distinct ordination spaces, the degraded forest plots intergraded between the two; though they were, in general, more similar to those of the rainforest plots (Fig 5.4).

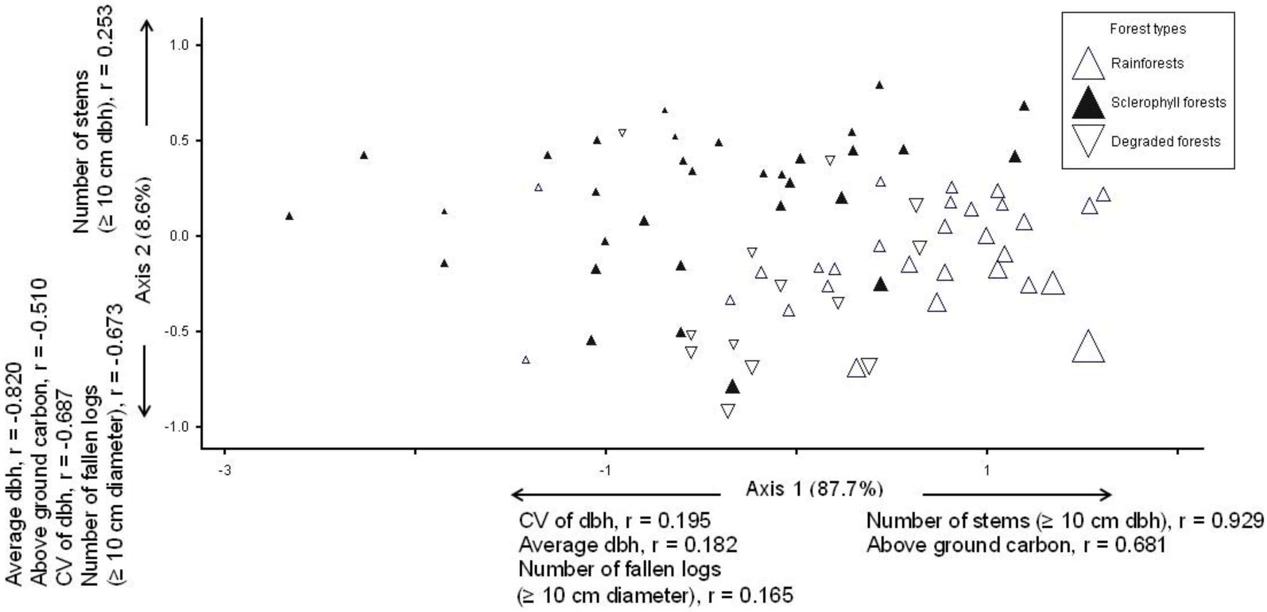

Figure 5.4. Nonmetric multidimensional scaling (NMS) ordination showing the variability of above ground carbon storage in relation to other forest attributes of plots located in rainforests (n= 29), degraded forests (n=13), and sclerophyll forests (n=32) of the Wet Tropics bioregion of northeast Australia. The size of the dots indicates relative quantity of above ground carbon storage. The x-axis (axis 1) represents 87.7% (r^2) and the y-axis (axis 2) 8.6% (r^2) of the described variation.

5.3.4. Factors determining carbon stock

The backwards, stepwise, negative binomial generalized linear model (GLM) process identified three significant explanatory variables for determining the carbon storage of plots ($R^2 = 0.888$; null deviance minus residual deviance/null deviance): the number of fallen logs per plot, average tree diameter breast height per plot and the tree abundance per plot (Figs. 5.5a, b, c and Table 5.1). All of these explanatory variables displayed a positive correlation with the carbon storage (Mg ha^{-1}) of plots (Figs. 5.5a, b, c and Table 5.1).

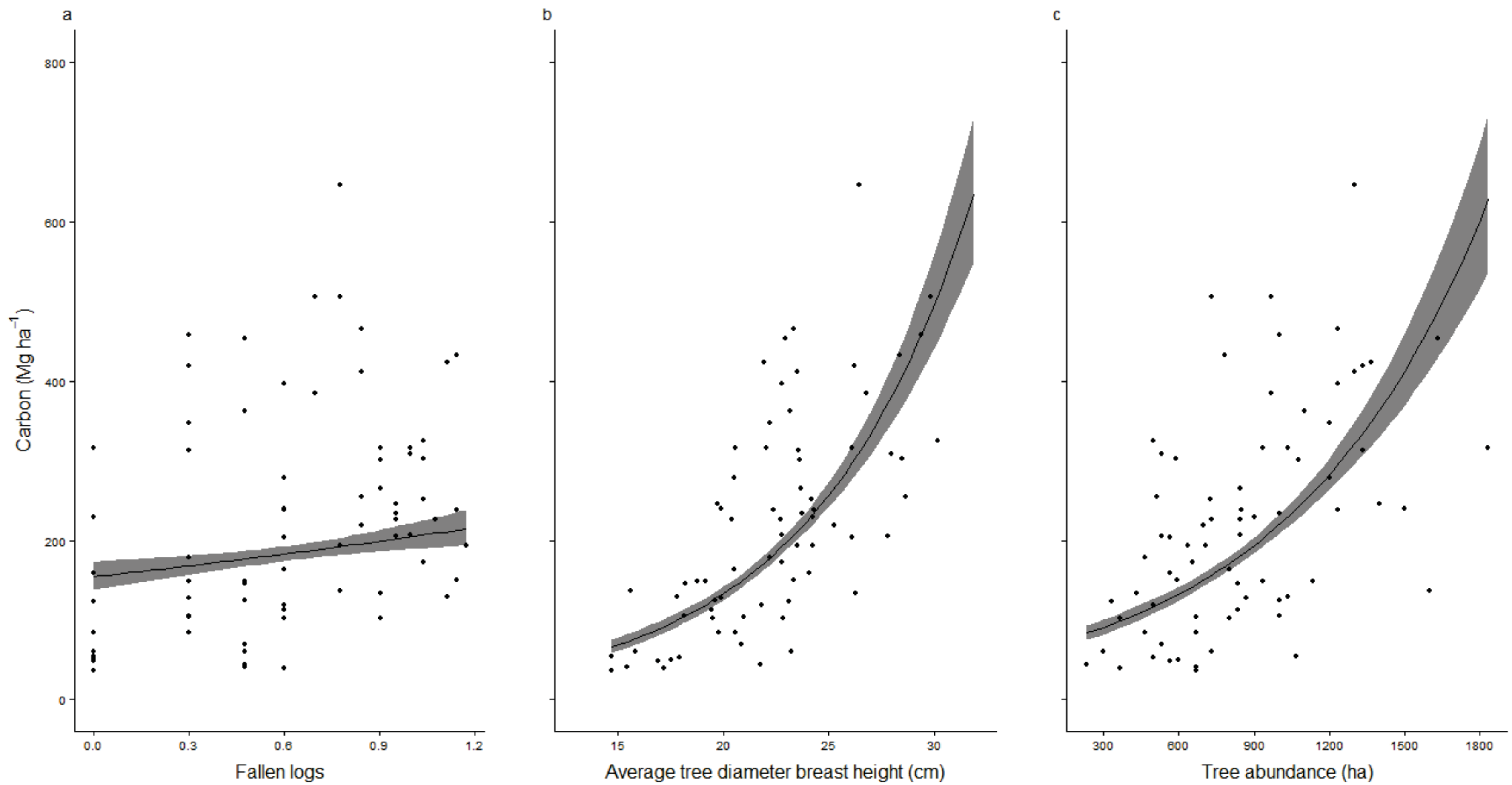

Figure 5.5. The significant relationship between a) fallen logs (≥ 10cm diameter) ha^{-1} (log transformed) , b) average tree diameter breast height (cm) and c) tree abundance and carbon storage (Mg ha^{-1}) of the examined plots, within the Wet Tropics bioregion of northern Queensland, Australia. Filled circles represent the plot (50m×10m) values. The trend line was constructed using a binomial GLM with logit link function and shaded areas represent the 95% confidence intervals.

Table 5.1. Generalized linear model result for describing carbon storage (Mg ha^{-1}) of the examined forests. Only the explanatory variables found to be significant according to their drop1 model comparison values are shown. The fallen logs variable was log 10 transformed prior to the analysis.

	Estimate	SE	Z value	P(>|z|)
Intercept	1.073	0.1785	6.007	< 0.001^
Fallen logs (≥ 10cm diameter) (log 10 transformed)	0.2513	0.0779	3.223	0.001#
Tree DBH (cm)	0.1307	0.0072	17.939	< 0.001^
Tree abundance	0.0012	7.646e-05	1.128	< 0.001^

(*denotes significance where $p < 0.05$, # $p < 0.01$ and ^ $p < 0.001$)

5.4. Discussion

We found that rainforests within the Wet Tropics bioregion store the highest levels of above ground biomass carbon (ABC) of the three examined vegetation types, and when these forests experience degradation their storage capacity is reduced (Figs. 5.3 and 5.4). However, the reduction in ABC storage values of plots in degraded forests compared with those of rainforest was not significant (Fig. 5.3) although degraded forest plots did display considerably more variation in ABC storage (Fig. 5.4). We suggest that it is likely that the examined degraded forests are in an advanced stage of regeneration given the lack of significant difference between their average tree size (dbh) and that of the rainforest. Moreover, the examined degraded forests on a global spectrum still store proportionately high values of ABC (241.04 ± 27.09 Mg ha^{-1}) when compared with other degraded tropical forests. For example, our reported values of ABC for the examined degraded forests (Fig. 5.3) are higher than those of Ioki et al. (2014) who estimated 52.18–229.11 Mg ha^{-1}, and 136.00–382.59 Mg ha^{-1} ABC for the highly degraded and moderately degraded tropical rainforests of northern Borneo, and for those of Usuga et al. (2010) who reported 99.6 Mg ha^{-1} and 85.7 Mg ha^{-1} ABC storage from Tropical Pine and Teak forest plantations in Colombia. Nonetheless, our observation of a non-significant decline in the ABC storage capacity of degraded forests compared to non-degraded rainforests is supported by findings reported elsewhere from within the tropics (Achard et al., 2004; Bustamante et al., 2015; Laurance et al., 1997; Pregitzer and Euskirchen, 2004).

Much of the decline in ABC storage within the degraded forests we examined appears to be due to increased disturbance. For instance, degraded forests when compared with non-degraded rainforests were found to display fewer trees (Fig. 5.2) and possess a higher number of fallen logs (Fig. 5.2). Furthermore, many of the degraded forests we examined were fragmented, a process which is known to result in higher levels of forest disturbance and tree loss through an increased susceptibility to wind damage (Laurance and Curran, 2008) and an altered microclimate on forest edges (Magnago et al., 2015; Williams-Linera et al., 1998). Moreover, both wind damage and microclimatic alterations within forests degraded by fragmentation are known to result in the disproportionate loss of large trees, especially on forest edges (Laurance et al., 2000; Laurance et al., 1997; Oliveira et al., 2008). This loss of large trees can significantly alter the carbon storage capacity of degraded forests as large trees are known to drive tropical forest ABC storage (Slik et al., 2013). In addition, forest fragmentation is known to increase wind damage susceptibility which may be particularly pertinent to lowland forests of this geographic region as they are exposed to regular cyclonic impacts, with the greater Wet Tropics bioregion experiencing 45 recorded east-to-west moving tropical cyclone impacts over the period 1858–2011 (Turton, 2008, 2012). Therefore, strategies that minimize the disturbance of degraded forests and especially forest edges, might allow for enhanced ABC storage through successional recovery of the tree community and re-instatement of resilience to natural disturbances, particularly within larger fragments of forest. For instance, employing wind disturbance mitigation strategies such as wind-buffer plantings along the forest edges (Goosem and Tucker, 2013) of the degraded forests of the region could substantially assist in decreasing forest disturbance (Laurance and Curran, 2008) and thus increase the ABC storage of these forests.

Sclerophyll forests within the study region had the smallest average tree size (dbh) (Fig. 5.2) and stored the least ABC of the three examined vegetation types (Fig. 5.3). The small average size (dbh) of trees within the sclerophyll forests compared to the other vegetation types is unsurprising given the less productive environmental envelope this vegetation type occupies (i.e. lower rainfall) and the fact that sclerophyll forests are pyrophytic and as such recruitment events are often determined by fire events and the time intervals between these (Bowman, 2000). Previous work (Ash, 1988) has suggested that one of the main determinants of the distribution of the rainforest and sclerophyllous vegetation types within the examined region may be fire, especially given the pyrophobic nature of the rainforest vegetation (Ash, 1988; Bowman, 2000; Fensham et al., 2003). Therefore, management practices that aim to optimize ABC storage within the Wet Tropics bioregion should, where practicable and appropriate, focus on the exclusion of fire from the landscape to allow for the rainforest vegetation type to persist.

Assuming the conversion of degraded forest plots to sclerophyll forest by fire can be avoided, assisted restoration of these forests within the Wet Tropics bioregion may be an effective management strategy to allow for significant net ABC storage gains. This is suggested as the degraded forests within the region vary in their ABC storage capacity from values at a low end similar to sclerophyllous vegetation through to those of comparable non-degraded rainforests (Fig. 5.4). In particular, restoration of localized factors which support the retention of large trees and increase tree abundance would significant increase ABC storage across all forest types and within the degraded forest in particular (Figs. 5.4 and 5.5, Table 5.1; Slik et al. (2013)). Although an increase in the number of fallen logs (as a proxy for disturbance) was also found to increase the ABC storage capacity of the examined forests (Fig. 5.5), it is highly likely (given previous research on forest disturbance (Laurance et al., 2006; Nascimento and Laurance, 2004)) that increased disturbance within these forests would result in an asymptote and eventual negative relationship occurring between this and ABC storage. As such, utilizing intermediate disturbance to attain an increase ABC storage of forests in the studied region would be problematic and impractical.

Finally, the Wet Tropics bioregion is a highly contested landscape (Stork and Turton, 2008). Consequently, within this landscape, multi-value land usage strategies may maximize the likelihood of degraded forest retention. For instance, as well as their significant ABC storage values (Fig 5.3), tropical rainforests are known to house the zenith of terrestrial biodiversity (Gibson et al., 2011). Additionally, remnant, fragmented and degraded forests can provide an important biodiversity repository for many complex tropical landscapes (Arroyo-Rodríguez and Mandujano, 2006; Edwards and Laurance, 2013; Magnago et al., 2015). Consequently, degraded forest management within the Wet Tropics bioregion provides considerable opportunities for integrating ABC storage values with biodiversity conservation. In particular, the uplands of the Wet Tropics bioregion, as well as providing an area of low disturbance and thus optimal locations for ABC storage in both rainforests and degraded forests (Fig. 5.4), are also a known "hotspot" for endemism and diversity of numerous biota (Williams and Pearson, 1997; Williams et al., 1995; Yeates et al., 2002) many of which are under threat (Costion et al., 2015; Williams et al., 2003). Moreover, recent studies of forest restoration within this region suggest that secondary and degraded forest restoration may be passively enhanced through selection of sites in close proximity to primary forest (Sloan et al., 2015), although if maximal biodiversity outcomes are to be gained specific species may need to be actively restored (Shoo et al., 2015).

5.5. Conclusions

To maximise ABC storage within the complex landscape of the Wet Tropics bioregion, it is optimal to conserve primary rainforests at sites that experience low levels of disturbance. Additionally, although degraded forests do not store as much ABC as non-degraded rainforest they play a considerable supplementary role in ABC storage and given appropriate management (i.e. disturbance minimization through fire exclusion and edge buffer plantings) and sufficient recovery, they can store as much ABC as non-degraded rainforest. Any additional ABC storage provided by degraded forests will come through the accumulation of additional carbon from the atmosphere and thereby contribute to climate change mitigation. In addition, if degraded forests in close proximity to primary forests can be restored and supplementarily seeded with selected tree species they may also provide additional and considerable biodiversity conservation capacity.

Acknowledgments

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OVERVIEW OF CHAPTER 6

This chapter is based on a paper prepared for submission to the journal *Ecological Indicators* [8] with minimal formatting changes. In this chapter I address objective number five of the thesis: to determine the likely effects of climate change on ecosystem services supplied from rainforests and sclerophyll forests.

Utilizing forest vegetation data I spatially assess the supply of ecosystem services from rainforests and sclerophyll forests of the Wet Tropics bioregion. Through *Climate Future Tools for Australia*, I project future climate change for the Wet Tropics landscape under two RCPs (Representative Concentration Pathway) (RCP6.0 and RCP8.5). I use Net Primary Productivity (NPP) as a surrogate to project the supply of examined ecosystem services under future climate change scenarios.

[8] **Alamgir, M.**, Turton, S.M., Macgregor, C.J., Pert, P.L. (in prep.). Climate change effects on forest ecosystem services. *Ecological Indicator*

CHAPTER 6

CLIMATE CHANGE EFFECTS ON FOREST ECOSYSTEM SERVICES

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Sclerophyll forest in the Wet Tropics (Danbulla State Forests, Tenaro Dam site)- tree mortality most likely from drought (Photo credit: Mohammed Alamgir)

ABSTRACT

Global climates are changing at a faster rate than anticipated, threatening the supply of many ecosystem services; however, the scale and magnitude of climate change effects on the supply of forest ecosystem services is poorly understood. After collecting and analysing forest vegetation data in the Wet Tropics bioregion of northeast Australia we evaluated the spatial distribution of seven ecosystem services supplied from rainforests and sclerophyll forests in the landscape: global climate regulation, erosion regulation, cyclone protection, habitat provision, energy provision and timber provision. Then, using automatically calibrated regression models among NPP (Net Primary Productivity, NPP - the net exchange of energy between ecosystem and environment) and each examined ecosystem service, we demonstrated the likely trend for future supply of those seven ecosystem services under two climate change scenarios – RCP6.0 (Representative Concentration Pathway, RCP) and RCP8.5 for two time periods 2050 and 2070. We found that the supply of the most of the ecosystem services from rainforests may be reduced with future climate change scenarios; however, uncertainty exists on the supply of ecosystem services from sclerophyll forests in the bioregion. The ecosystem services supply from upland rainforests is likely to be more negatively affected than lowland rainforests. We also found that the scale and magnitude of climate change effects on ecosystem services will vary with the following three factors: i) type of ecosystem service; ii) source of supply, i.e. forest type; and iii) climate change scenarios. We conclude that an ecosystem service based adaptation approach would be extremely useful in maintaining sustainable supply of ecosystem services under climate change. This study will be useful for natural resource management planning and climate adaptation planning that is concerned with the sustainable supply of ecosystem services under future climate change scenarios.

Keywords: NPP; elevation gradient; rainforest; sclerophyll forest

6.1. Introduction

The supply of 15 of the 24 globally recognised ecosystem services are in various levels of decline (Millennium Ecosystem Assessment (MA, 2005)). On the whole, climate change is one of the major drivers forcing the decline in the supply of various ecosystem services (MA, 2005; Rasche, 2014; Schröter et al., 2005; Shaw et al., 2011). However, the scale and magnitude of climate change effects on different forested ecosystems and various ecosystem services are not clear. Very recently, the Intergovernmental Panel on Climate Change (IPCC) published four plausible climate change scenarios – RCP (Representative Concentration Pathway) 2.0 (lowest emission scenario), RCP4.0, RCP6.0 (intermediate emission scenarios) and RCP8.5 (high emission scenarios), with a wide range of projections for rainfall change and temperature increase over this century (IPCC, 2014a). Therefore, it can be hypothesised that climate change effects on the supply of ecosystem services will vary with the RCP scenario the world tracks in the future. As a consequence of escalating greenhouse emissions (IPCC, 2007, 2014a), it is likely that the supply of many ecosystem services will be more impacted due to climate change in this century, at a rate of change unprecedented for many hundreds of thousands of years. Other processes such as land use change may act synergistically with climate change, adding further to the loss of ecosystem services.

Along with climate change scenarios (RCPs), the level and nature of climate change effects on ecosystem services may vary with ecosystem types (Schröter et al., 2005), and among ecosystem services (Rasche, 2014; Shaw et al., 2011). For example, as a consequence of both climate change and land use change, Schröter et al. (2005) reported a projected decline of certain forest-based ecosystem services from different European ecosystems. Shaw et al. (2011) showed a projected decline of certain forest-based ecosystem services from a landscape of California. Furthermore, Rasche (2014) found that the supply of most of the tree-based ecosystem services will be reduced from temperate forests at various levels due to climate change. Yet, how climate change will affect the supply of ecosystem services from tropical forests has not been fully explored.

Tropical forest ecosystems are the single most important terrestrial repository to supply multiple ecosystem services (Costanza et al., 1997). Human community well-beings are largely influenced by the sustained supply ecosystem services (MA, 2005). Therefore, any negative effects of climate change on the ecosystem services supplied from tropical forests will substantially undermine community cohesion and development. Hence, climate adaptation pathways need to be developed and applied to maintain the sustainable supply of ecosystem services in tropical forest landscapes (Alamgir et al., 2014b; de Bremond and Engle, 2014; Lawler, 2009). For effective climate adaptation, a detailed understanding on the likely effects are required (Lawler, 2009). The commonly used methods to evaluate the supply of ecosystem services are either unsuitable for the tropical forest ecosystems or unsuitable to predict the supply of most of the ecosystem services under different climate change scenarios. For example, InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) is widely applied for tradeoffs analysis under land use change scenarios (Nelson et al., 2009); GVDM (Global Vegetation Dynamic Model), widely used for carbon and water cycles (Shaw et al., 2011), and also for the temperate forests (Schröter et al., 2005).

NPP (Net Primary Productivity, the net exchange of energy between ecosystem and environment), can potentially be used as a surrogate to evaluate the distribution of certain ecosystem services (Egoh et al., 2008). Along with ecosystem services, NPP usually varies with temperature and rainfall (Del Grosso et al., 2008; Jiang et al., 1999). Prediction of NPP in a tropical forested ecosystem is also possible under different climate change scenarios (Del Grosso et al., 2008). Therefore, the relationship between NPP and ecosystem services can potentially be used to predict the supply of ecosystem services from tropical forest ecosystems under different climate change scenarios.

The Wet Tropics bioregion in northeast Australia is a unique landscape consisting of various forest types while dominated by rainforests and sclerophyll forests (Stork and Turton, 2008). The rainforest in this landscape is one of the oldest rainforest types on the planet (Stork et al., 2011) and is identified as the sixth most irreplaceable natural protected area considering the habitat of all species found in this region (Le Saout et al., 2013). The biodiversity values of this rainforest are under severe threat due to climate change (Costion et al., 2015; Hilbert et al., 2001; Williams et al., 2003) but the fate of ecosystem services is unknown, although the Wet Tropics bioregion provides several ecosystem services which are critical from the local to global scales (Alamgir et al., 2016). The overall aim of our study is to provide a rigorous scientific understanding about climate change effects on the ecosystem services of this bioregion.

In this study, we evaluate the spatial distribution of seven ecosystem services supplied from rainforests and sclerophyll forests of the Wet Tropics bioregion using vegetation attributes: global climate regulation, air quality regulation, erosion regulation, cyclone protection, habitat provision, energy provision and timber provision. Then, using automatically calibrated regression models between NPP and ecosystem services we show the trend of supply of these seven ecosystem services under two future climate change scenarios (RCP6.0 and RCP8.5) for two time periods, 2050 and 2070. Our study will be useful for natural resource management and climate adaptation planning focusing on ecosystem services in this region, and will contribute to a global understanding of climate change effects on tropical forests ecosystem services.

6.2. Materials and methods

6.2.1. The study area

We conducted this study in the Wet Tropics bioregion of northeast Australia (Fig. 6.1), which is a heterogeneous forested landscape dominated by rainforests and sclerophyll forests (Ash, 1988). The total area of this bioregion is approximately two million ha (Goosem, 2002) and approximately 45% (894,420 ha, mainly rainforests) of this area was inscribed on the World Heritage List in 1988 (UNESCO, 1988). The rainforests of the Wet Tropics are biologically diverse having relatively close canopies, while the sclerophyll forests have relatively open canopies (Stork and Turton, 2008; Tracey, 1982; Webb, 1959; WTMA, 2014). The sclerophyll forests are dominated by *Acacia* and *Eucalyptus* species (Tracey, 1982) while, biodiversity in the rainforests are unique in the world (Le Saout et al., 2013).

The Wet Tropics bioregion is one of 89 bioregions in Australia. Each bioregion is different in climate, geology, landforms pattern, ecological features and biological communities (Department of Environment, 2015). The Wet Tropics bioregion experiences a seasonally wet tropical climate – mean annual rainfall ranges from 1200 to 4000 mm, and mean annual temperature ranges from 17°C to 31°C (Goosem, 2002;

Trott, 1996). Elevation ranges from a few metres above mean sea level (msl) to around 1000 m, although the highest peak within the region is 1622 m.

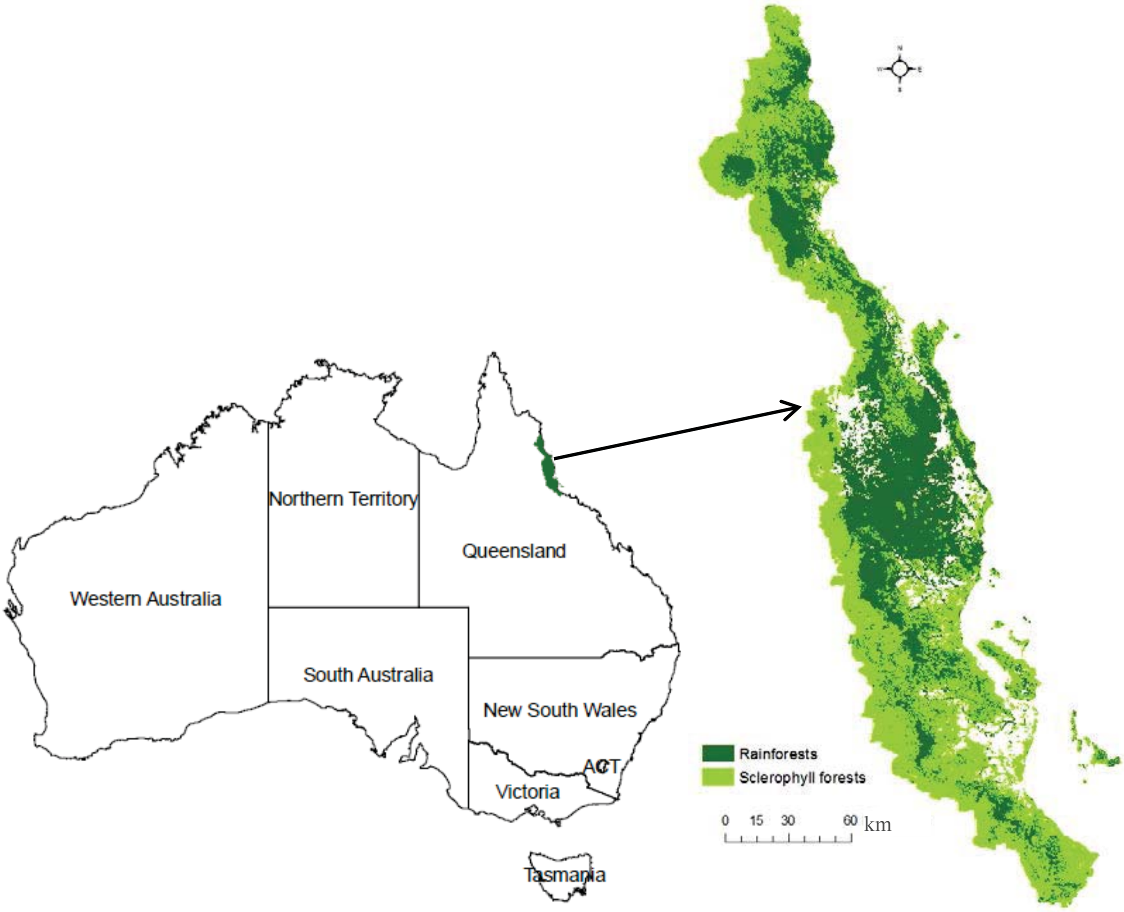

Figure 6.1. The study area- Wet Tropics bioregion, northeast Australia with distribution of rainforests and sclerophyll forests.

6.2.2. Sampling and data collection

We sampled 24 rainforest sites (13 sites found in mesophyll forests, nine in notophyll forests, and two in disturbed rainforests), and 34 sclerophyll forest sites (32 from sclerophyll forests and woodlands, and two from the disturbed sclerophyll forests) in the Wet Tropics bioregion. Each plot was 0.05ha in size (50m×10m) and was located on a map prior to entering the field to avoid creeks and other water bodies, and a minimum 50m distance established from other land uses to avoid edge effects. Considering the strong environmental gradient, our plots were distributed from a few metres to more than 1000m above msl.

We used a modified transect method for tree data collection within 0.05ha plots (Alamgir et al., 2016). Large trees (≥ 20cm dbh) were measured in the 0.05 ha plots, medium trees (≥ 10 cm dbh) and small trees (≥ 2.5cm) were measured in the 0.03 and 0.015 ha subplots respectively. The diameter at breast height (dbh) of trees was measured using a dbh tape, and height of the representative trees (2 to 3 trees from each subplot) was measured using a Forestry Pro Laser Range Finder Hypsometer, and canopy cover was measured using a convex spherical crown densitometer. Along the centre line of transects we counted the number of fine and coarse woody debris that intersected the centre line ≤ 1m high. We also measured canopy cover, ground cover (%) from 1m high at three $1m^2$ subplots placed at 5m, 25m and 45m. Each plot was spatially referenced and attributes such as slope, soil, and altitude were also recorded.

6.2.3. Assessment of ecosystem services

Various frameworks have been developed for the classification of ecosystem services (e.g. (Bagstad et al., 2013; Boyd and Banzhaf, 2007; de Groot et al., 2010; de Groot et al., 2002; Fisher et al., 2009; MA, 2005; van Oudenhoven et al., 2012)). MA provided one of the pioneer framework, that has been commonly used in ecosystem services research (e.g. (Baral et al., 2013a; Burkhard et al., 2012; Harrison et al., 2014; Pert et al., 2010; Schneiders et al., 2012; Sohel et al., 2014)). Therefore, we utilized the framework of MA (MA, 2005) to classify selected ecosystem services (Table 6.1). Detail protocol for the assessment of examined ecosystem services were illustrated in Alamgir et al. (2016) however summary are presented in Table 6.1.

Table 6.1. Methods used for the assessment of selected ecosystem services along with their attributes and indicators.

Ecosystem service component	Attribute	Indicator	Assessment methods
Regulating ecosystem services			
Global climate regulation	Ecosystem plays an important role in global climate regulation by sequestering greenhouse gases	Sequestered atmospheric CO_2 by above ground tree biomass (CO_2 equ. Mg ha^{-1})	Alamgir et al. (2016)
Air quality regulation	The capacity of ecosystem to remove toxic and other elements from the atmosphere	Tree canopy cover (%)	Alamgir et al. (2016)
Erosion regulation	Vegetative cover plays an important role in soil retention	Stratified vegetation cover index	Zhongming et al. (2010)
Cyclone protection*	The presence of forest ecosystems can dramatically reduce the damage caused by cyclones	Coefficient of variance (CV) of tree diameter at breast height (dbh)	Alamgir et al. (2016)
Provisioning ecosystem services			
Habitat provision	Importance of ecosystem to provide habitat for species (particularly fauna) and natural biodiversity.	Multi-criteria index	Alamgir et al. (2016)
Energy provision	Presence of trees or plants with potential use as energy source	Above ground tree biomass (AGB) (Mg ha^{-1})	Chave et al. (2005)
Timber provision	Presence of species with potential use for timber	Tree basal area (BA) (m^2 ha^{-1})	Alamgir et al. (2016)

The ecosystem service components, attributes and indicators are based on de Groot et al. (2010); MA (2005); Burkhard et al. (2012); and de Groot et al. (2002); Alamgir et al. (2016). * We use term cyclone protection rather than storm protection (which was used in MA, 2005) considering local preferences.

The spatial analysis was conducted in ESRI ArcGIS 10.2 and statistical analysis was conducted in IBM SPSS 20.

6.2.4. Modelling climate change

Tools have yet to be developed to model climate change at a local scale in Australia, utilising the IPCC's (Intergovernmental Panel on Climate Change) AR5 (Assessment Report five) (IPCC, 2014a) RCPs 2.6, 4.5, 6.0 and 8.5. A few web-based exploration tools like Terra Nova (https://terranova.org.au/Tools) are available these are more suitable for Natural Resource Management (NRM) practitioners rather than researchers. In our study, we have utilised CSIRO[9]'s CMIP5 Global Circulation Model – newly developed Climate Future Tools for Australia's NRM regions (http://www.climatechangeinaustralia.gov.au). We model seasonal temperature change

[9] The Commonwealth Scientific and Industrial Research Organisation (CSIRO) is Australia's national science agency.

and seasonal rainfall change in the Wet Tropics cluster for a range of time periods until the end of this century, based on the IPCC's AR5 RCPs: RCP2.6, 4.5, and 8.5. The Wet Tropics cluster consists of the whole Wet Tropics bioregion with some additional areas based on catchment boundaries. The Climate Future Tools has been developed for larger NRM regions of Australia so for convenience we used the larger Wet Tropics Cluster region (McInnes et al., 2015) rather than Wet Tropics bioregion for our climate change modelling.

6.2.5. Climate change impacts assessment

We estimated current and projected NPP (g m^{-2} $year^{-1}$) for RCP6.0 and RCP8.5 for 2050 and 2070 for each plot using the well-known Miami Model (equation 1) (Lieth, 1975). We then utilized automatically calibrated regression models to assess the relationship between NPP and each examined ecosystem service (details in supplementary materials). Finally, we used those regressions to project the future supply of each ecosystem service under RCP6.0 and RCP8.5 for 2050 and 2070 time period. We did not use RCP2.6 and RCP4.5 in the projection assuming that RCP2.6 is unrealistic, and RCP4.5 and RCP6.0 show nearly same trend of climate change. The current and projected annual rainfall under RCP6.0 and RCP8.5 for 2050 and 2070 for each plot was downscaled from WorldClim (www.worldclim.org) using an ensemble of CMIP5 models. In the projection, we found a very few unrealistic values, therefore, we have not considered values which projected more than 200% of change on either side of the baseline (positive or negative); subsequently we have removed those values from the systems.

$$NPP = 3000*(1-e^{-0.000664*R}) \qquad (1)$$

Where, R = annual rainfall (mm)

A number of methods are available in the literature to estimate NPP utilizing climatic data (e.g. (Del Grosso et al., 2008; Lieth, 1975)). As the forested ecosystems of the Wet Tropics bioregion are largely regulated by rainfall (Hilbert et al., 2001; Stork and Turton, 2008; Webb, 1959), we utilized the Miami Model, which calculated the NPP as function of annual rainfall (equation 1) (Lieth, 1975).

6.3. Results

6.3.1. Spatial distribution of ecosystem services supplied

The overall distribution patterns of the supply of ecosystem services in the study area are displayed in Fig. 6.2. All of the examined ecosystem services were unevenly distributed and differed considerably both within the rainforests and sclerophyll forests. Considering global climate regulation, air quality regulation, cyclone protection, habitat provision, energy and timber provision, most of the rainforest plots exhibited higher values than sclerophyll forests, while erosion regulation values were found to be higher in most of the sclerophyll forest plots than rainforest plots. Our results showed that rainforest plots in high elevations supplied more global climate regulation, air quality regulation, energy provision and timber provision ecosystem services than rainforest plots in low elevations.

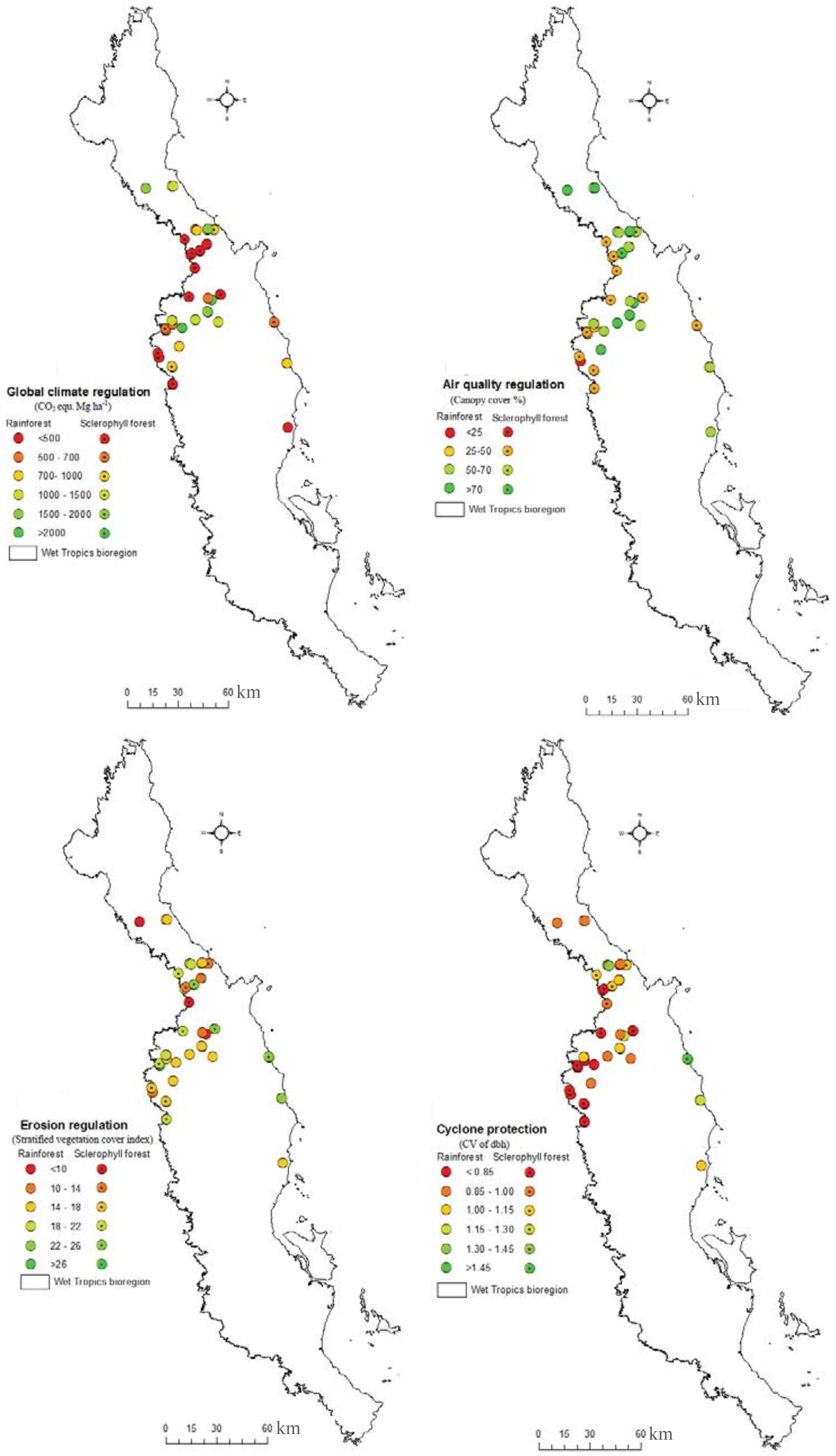

Figure 6.2. The spatial distribution of ecosystem services in rainforest and sclerophyll forest plots in the Wet Tropics bioregion northeast Australia.

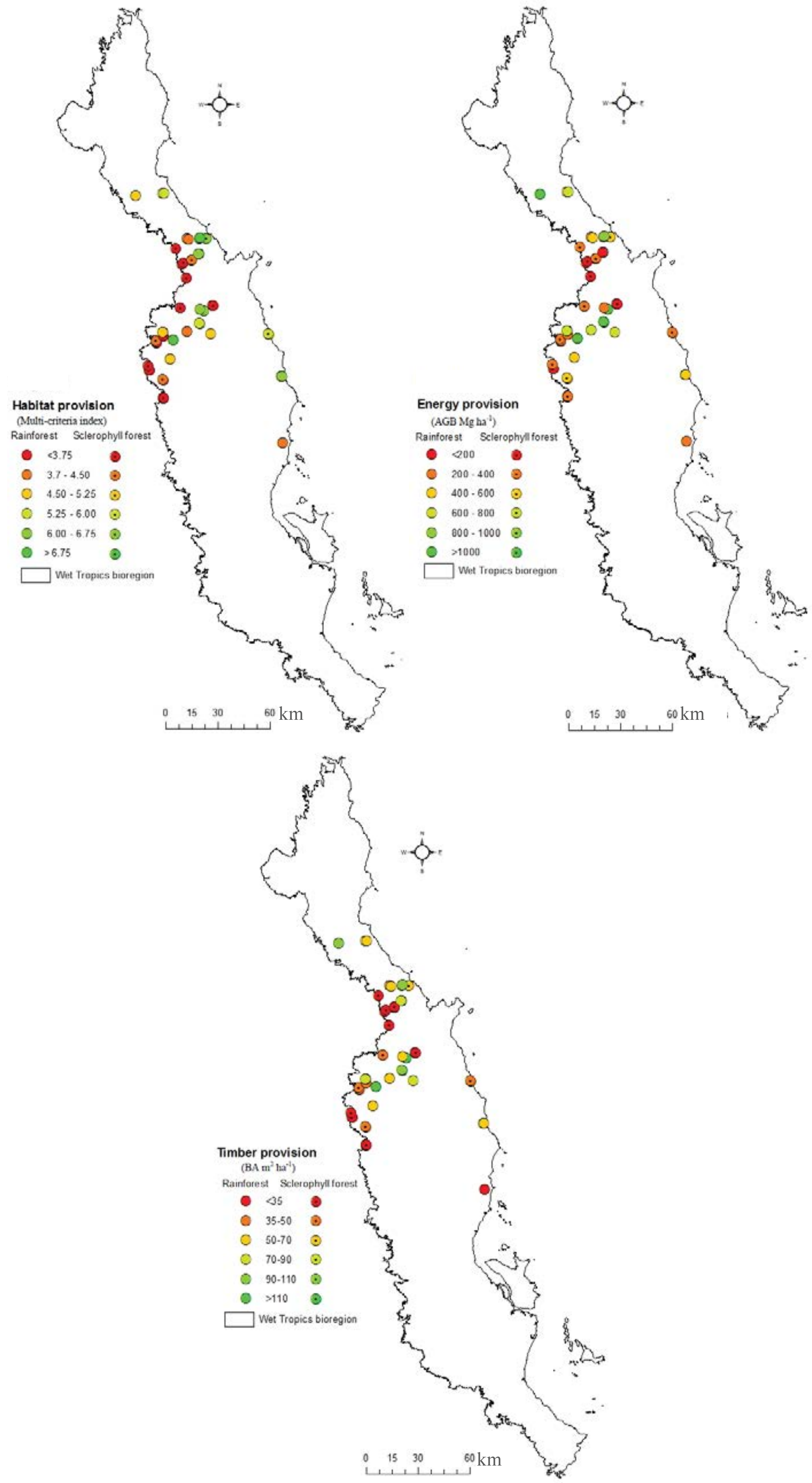

Figure 6.2. Continued- The spatial distribution of ecosystem services in rainforest and sclerophyll forest plots in the Wet Tropics bioregion northeast Australia.

6.3.2. Climate change trend

Global Circulation Models showed that for RCP8.5 (the highest emission pathway) summer, autumn, winter and spring mean temperatures in the Wet Tropics are likely to increase (considering baseline mean for 1986-2005) by nearly 1.5°C, more than 2°C and nearly 3°C by 2050 (2040-49), 2070 (2060-79) by the end of this century (2080-99) respectively (Fig. 6.3, top panel). The model also showed that total rainfall (comparing baseline mean for 1986-2005) in autumn, winter and spring is likely to decrease with RCP2.6, 4.5. A more significant decrease in spring (September-November) and greater anomalies in summer (December-March) rainfall with RCP8.5 were also revealed for the Wet Tropics region (Fig. 6.3, bottom panel).

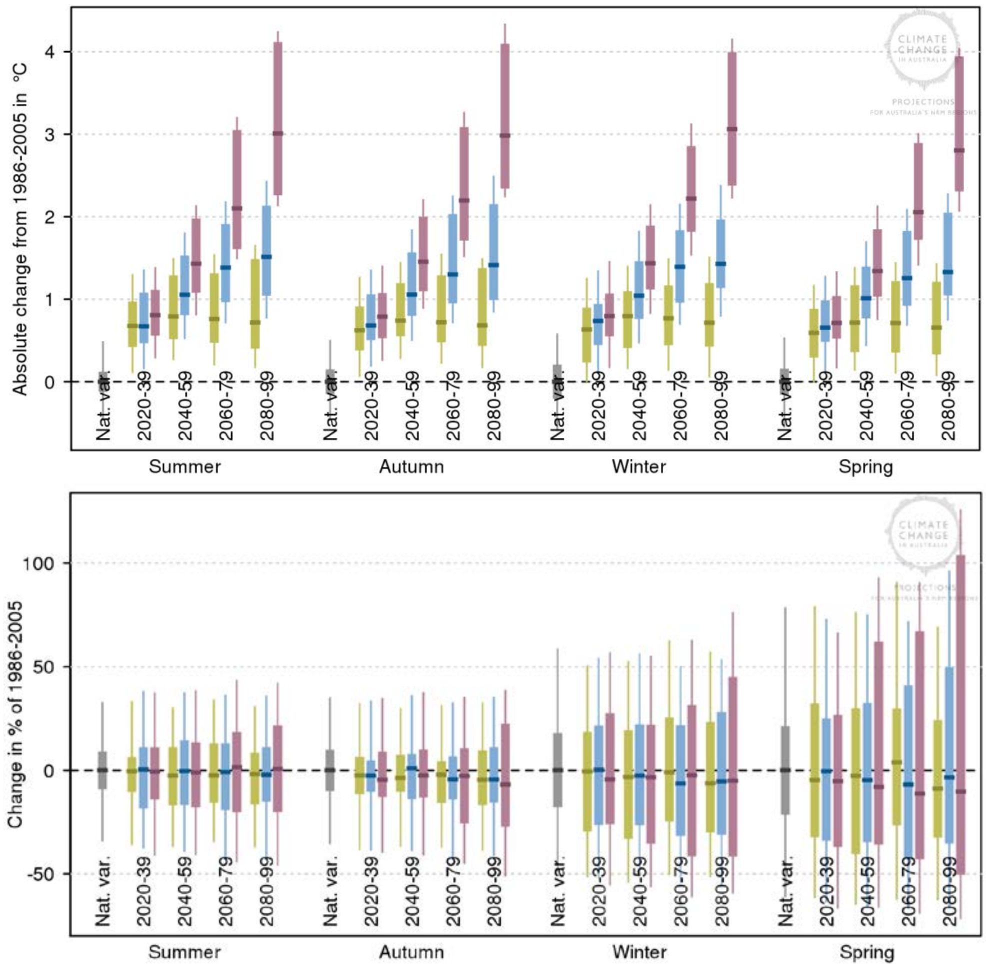

Figure 6.3. Climate change modelling for the Wet Tropics Cluster region, Australia. Top panel- seasonal temperature change; bottom panel-seasonal rainfall change. Left to right bars for each period indicates RCP2.6, 4.5, and 8.5. Horizontal line in each bar represents the median value. CMIP5 Global Circulation Model using CSIRO's Climate Future Tools.

6.3.3. Projected changes to the supply of ecosystem services

The supply of global climate regulation service from rainforests was found to be reduced from 37% by 2050 under RCP6.0 to 40% by 2070 under RCP8.5; this compared with an increased supply from 60 to 57% respectively from sclerophyll forests. Similarly, a decreasing trend in the supply of ecosystem services from rainforests was also projected for air quality regulation: -21to -24 %, cyclone protection: -10 to -17%, habitat provision: -24 to -28%, energy provision: -37 to -41% and timber provision: -31 to -33% by 2070 under RCP6.0 and RCP8.5 respectively (Fig.6.4). This contrasts with an increasing trend in the supply of ecosystem services from sclerophyll forests, e.g. energy provision (61 to 57%) and timber provision (42 to 38%) by 2070 under RCP6.0 and RCP 8.5 scenarios. Erosion regulation is the only ecosystem service that was found to increase from rainforests (9 to 8 %) by 2070 under RCP6.0 and 8.5, respectively. The changes of air quality regulation, erosion regulation, cyclone protection and habitat provision ecosystem services were found to be uncertain as projections were spread from positive to negative considering 25th to 75th percentiles spreads across sclerophyll forest plots for both RCP6.0 and RCP8.5 and for both time periods 2050 and 2070 (Fig. 6.4).

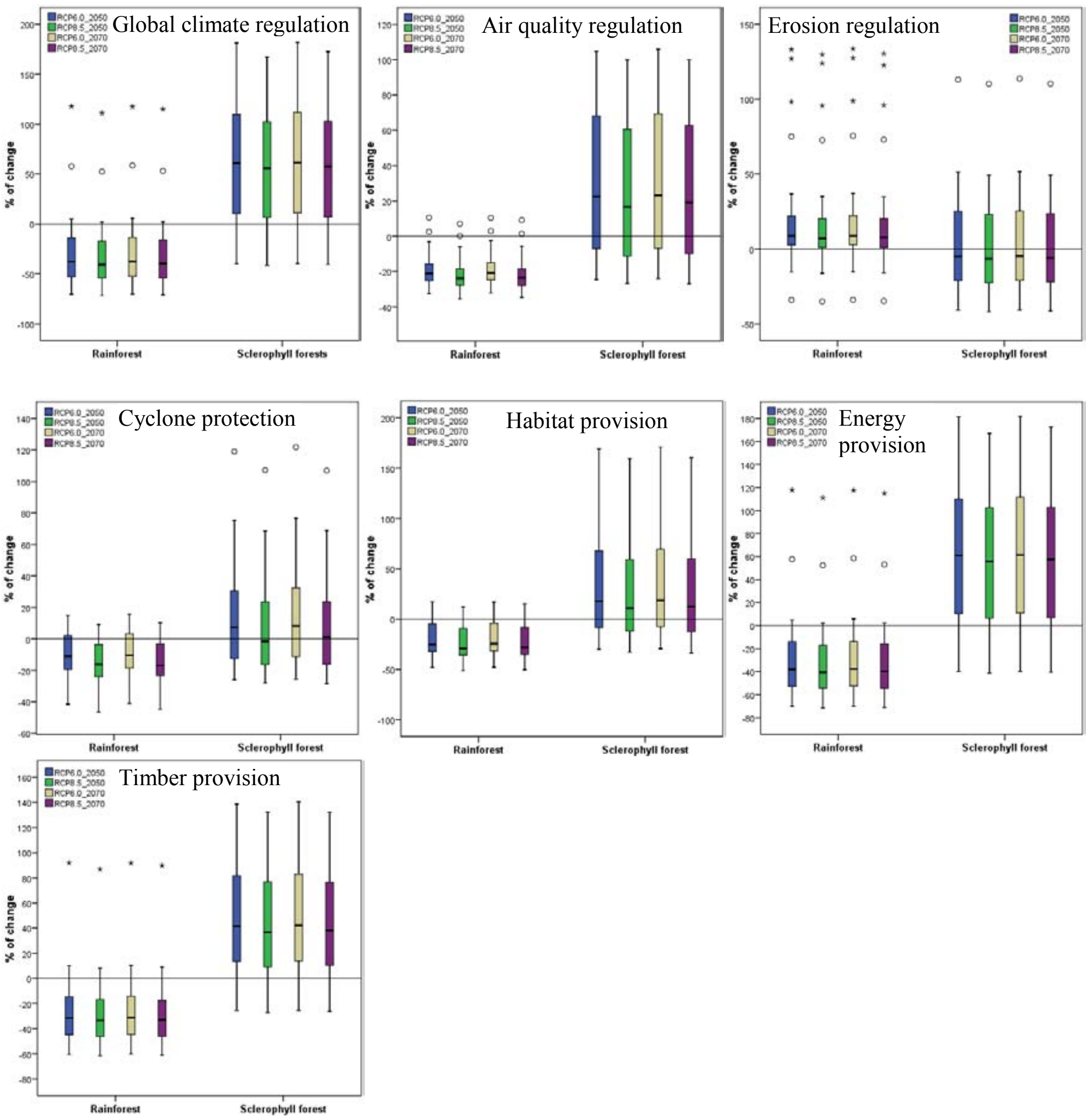

Figure 6.4. The predicted effects of climate change on ecosystem service supplies from rainforests and sclerophyll forests considering RCP6.0 and RCP8.5 for 2050 and 2070. Middle line = median; upper edge = 75th percentile; lower edge = 25th percentile; whisker caps = variability outside the quartiles; circles and stars = outliers.

6.3.4. Projected changes along the elevational gradient

The projection of air quality regulation and cyclone protection ecosystem services from rainforest plots both showed a clear trend of increasing reduction rates from low to high elevational gradients under two climate change scenarios (RCP6.0 and RCP 8.5) by 2050 and 2070 (Fig. 6.5). Global climate regulation, habitat provision, energy provision and timber provision ecosystem services from rainforest plots showed a relatively higher reduction rate in the mid elevation plots (e.g. >750 m to <900m);

however, they showed a higher reduction rate in high elevation than low elevational plots under RCP6.0 and RCP8.5 (Fig. 6.5). In the sclerophyll forest plots, only habitat provision ecosystem service showed a relatively lower increasing rate in high elevation plots than low elevation plots (gradually reached to the negative side in the higher elevation plots) under both climate change scenarios RCP6.0 and RCP 8.5 by 2050 and 2070 whereas no trend was projected in the supply of remaining ecosystem services from sclerophyll forest plots (Fig. 6.5).

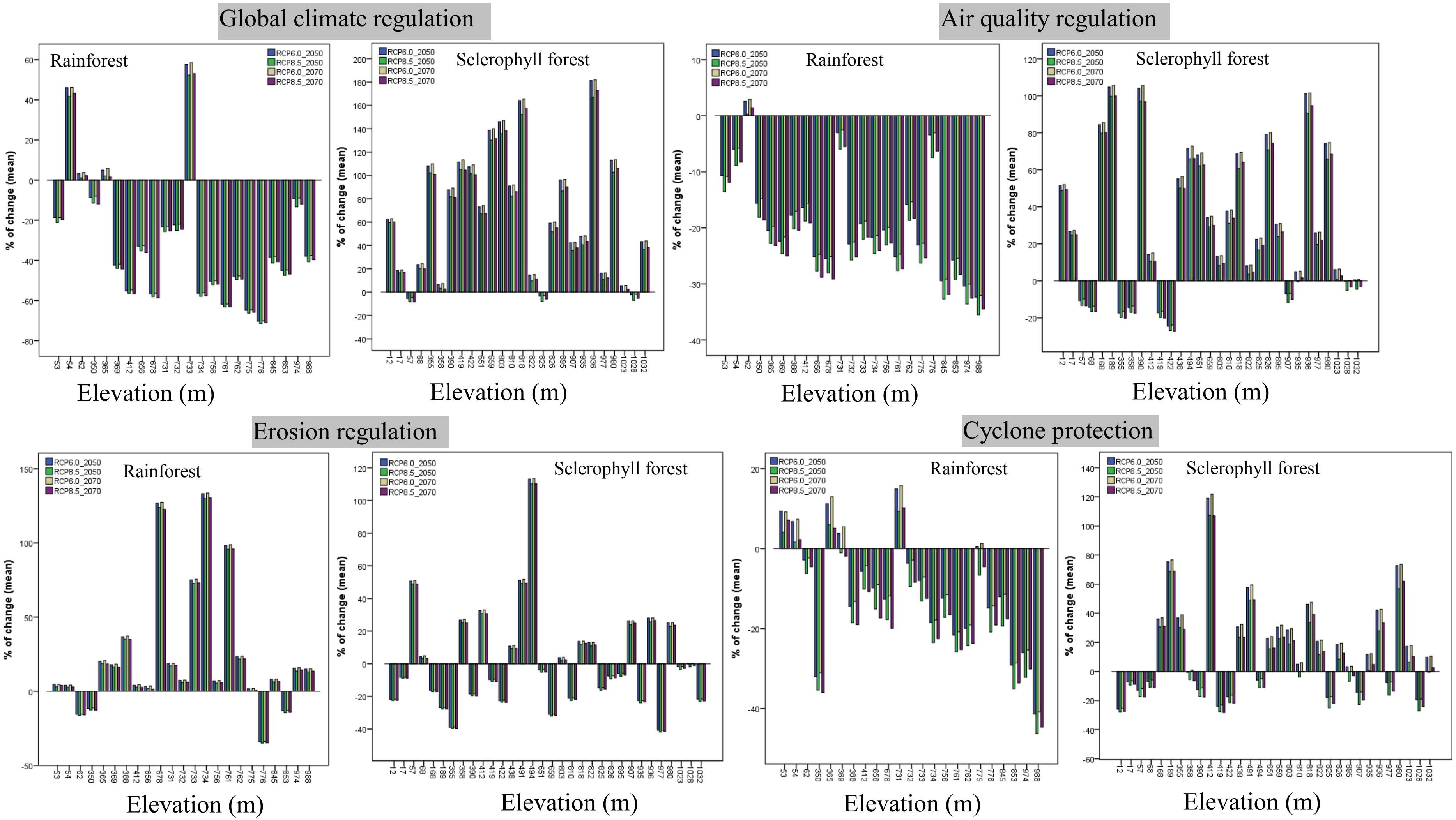

Figure 6.5. The predicted effects of climate change on ecosystem service supplies from rainforest and sclerophyll forest plots in the Wet Tropics northeast Australia along the elevation considering RCP6.0 and RCP8.5 for 2050 and 2070.

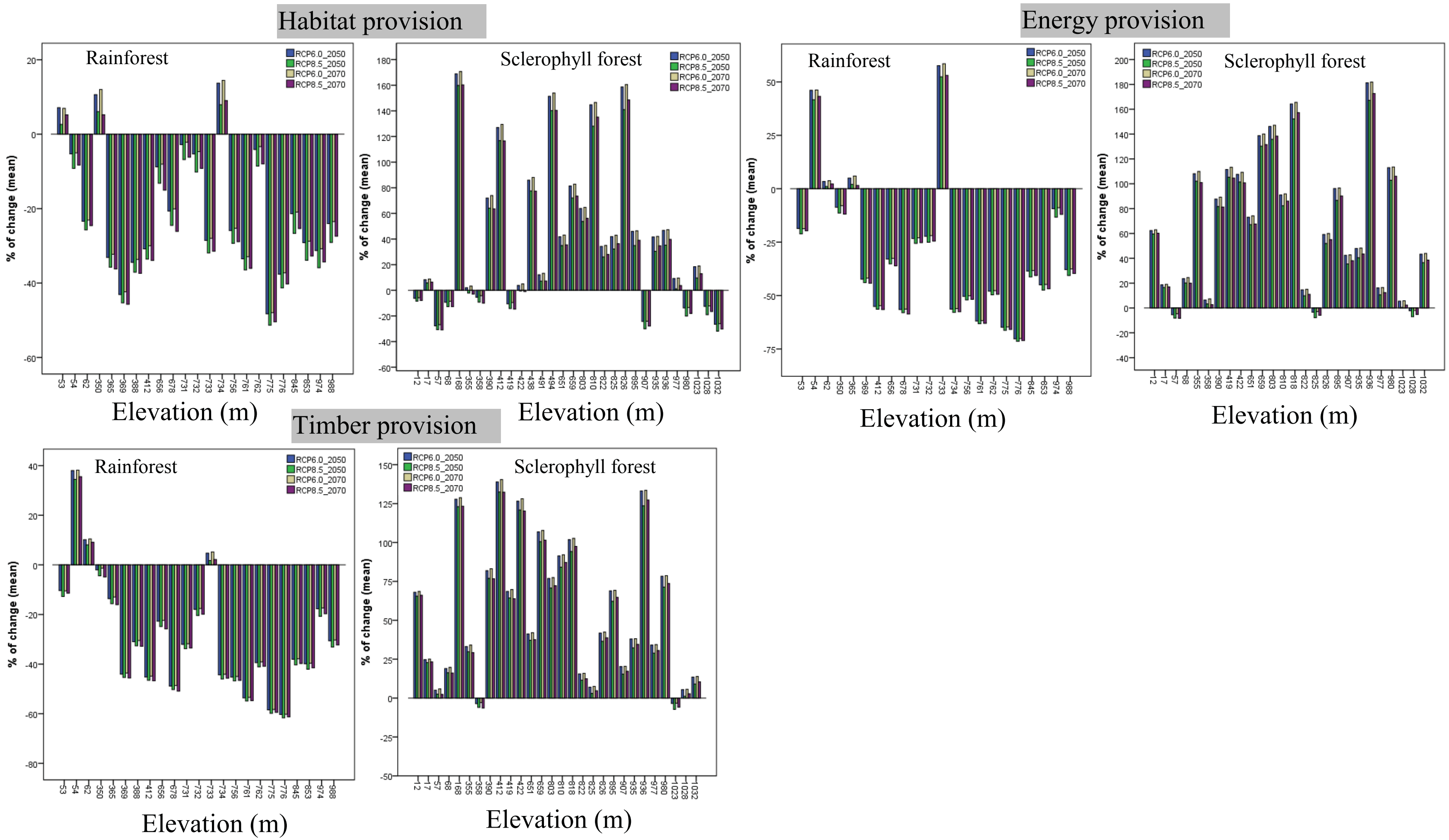

Figure 6.5. Continued-The predicted effects of climate change on ecosystem service supplies from rainforest and sclerophyll forest plots in the Wet Tropics northeast Australia along the elevation considering RCP6.0 and RCP8.5 for 2050 and 2070.

6.4. Discussion

The rate of negative effects of climate change on the supply of ecosystem services are likely to be more pronounced under the RCP8.5 scenario than RCP6.0 scenario, and the total negative effect will increase with time, i.e. will be greater negative effect by 2070 than 2050 (Figs. 6.4 and 6.5). A wide variation in the projection of seasonal annual rainfall under RCP6.0 and RCP8.5 by 2050 and 2070 was revealed from CMIP5 models, ranging from a 50% increase to 50% decrease (Fig. 6.3). As RCP8.5 is the worst case scenario of greenhouse gasses concentration in the atmosphere (IPCC, 2014a), so it has projected more drying trend than RCP6.0 and also more drying over time (Fig. 6.3). Therefore, it is likely that increased drying will produce more drastic impacts on the supply of ecosystem services from the rainforests in this region. Similarly, Rasche (2014) reported a significant negative impact on tree-based ecosystem services due to drought. More negative impacts on the forest-based ecosystem services with higher greenhouse gas emission scenarios were also projected from other regions (Rasche, 2014; Schröter et al., 2005). Therefore, our findings are comparable with previous findings and it may be inferred that higher pathways of greenhouse gas emissions are likely to have a substantially negative impact on the supply of ecosystem services from tropical forest landscapes.

Although ecosystem services from most of the rainforest plots showed that these forest types are likely to be negatively affected by climate change, sclerophyll forest plots in the same bioregion did not show any clear trend about the likely effects (Figs. 6.4 and 6.5). The ecosystem functioning and processes of the rainforests of this region are greatly regulated by the rainfall (Hilbert et al., 2001; Stork and Turton, 2008; Webb, 1959) and more rainfall favours the rainforests while less rainfall favours the sclerophyll forests (Hilbert et al., 2001). This could be the main reason for the projections of less ecosystem services from rainforests and relatively more ecosystem services (although uncertain) from sclerophyll forests. Therefore reduced rainfall projection for both climate change scenarios (RCP6.0 and RCP8.5) may disrupt functioning and processes within rainforest ecosystems, and hence reduce the sustained supply of rainforest ecosystem services. Furthermore, the rainforests of this region are growing in a certain rainfall range. The projected rainfall reduction in some plots of the rainforest may go beyond that limit, which are also likely to reduce the supply of ecosystem services from rainforests. Additionally, NPP is usually higher in the rainforest areas due to higher rainfall; reduced rainfall may potentially reduce the NPP in the rainforest areas (Ju et al., 2007; Matsushita et al., 2004; Mickler et al., 2002; White et al., 1999). In the NPP projection, we used Miami Model (details in section 6.2.5) which is based on rainfall projection. The sclerophyll forests of the Wet Tropics bioregion usually grow in lower rainfall areas, hence they are used to growing in relatively dry conditions with occasional fires, which might be the reason for the projected increase of ecosystem services supply in some of the sclerophyll forest plots. In some parts of the sclerophyll forest areas, climate change models have projected a small increase in the rainfall (due to spatial variation in the projection), which may have a positive impact to increase the NPP and subsequent supply of ecosystem services.

The magnitude and direction of climate change effects (positive or negative) on different ecosystem services (from the same forest type) are likely to be different under the same climate change scenario and time period (Figs.6.4 and 6.5). Our results are supported by previous research that reported the variable impact of the same climate change scenario on different ecosystem services supplied from the same forest type (Rasche, 2014; Schröter et al., 2005; Shaw et al., 2011). The supply of global climate regulation, energy provision, timber provision and habitat provision ecosystem services were projected to be more reduced than the remaining examined ecosystem services from rainforests under both climate change scenarios (Figs.6.4 and 6.5). Global climate regulation and habitat provision ecosystem services from the rainforest of the Wet Tropics bioregion are the two most important ecosystem services that are being used in local to global scale approaches for climate change mitigation and biodiversity conservation (Alamgir et al., 2016). The likely reduction of global climate regulation ecosystem services due to climate change has also been reported previously from other regions (Ju et al., 2007; Shaw et al., 2011; White et al., 1999). Therefore, our results are aligning with previous findings. The projected increasing trend of global climate regulation, energy provision and timber provision ecosystem services from the sclerophyll forest plots indicate the important capabilities of the sclerophyll forests under drying conditions than rainforests in this region, which were also reported by Hilbert et al. (2001).

The upland rainforests of the Wet Tropics bioregion are the hotspots of multiple ecosystem services (Alamgir et al., 2016). Most of the ecosystem services from the upland rainforest plots are likely to be more negatively affected than lowland rainforest plots under RCP6.0.and RCP8.5 by 2050 and 2070 (Fig. 6.5). This may be due to the combined effects of three factors: i) upland rainforests of the Wet Tropics are very environment sensitive, ii) ecosystem structure of the upland rainforests are largely controlled by the high amount of annual rainfall – including cloud stripping, and iii) the greater drying climate projection for the upland rainforest areas than lowland rainforest areas. It is reported that upland rainforests in the Wet Tropics bioregion are more vulnerable to climate change compared with lowland rainforests (Foster, 2001; Ostendorf et al., 2001). The potential biodiversity loss from the upland rainforests for this bioregion has also been published (Costion et al., 2015; Williams et al., 2003). Our study reported a potential loss of ecosystem services supply from the rainforests of the Wet Tropics bioregion which is likely to be more pronounced in the upland rainforests.

6.5. Conclusions

From this study it is evident that the distributions of different ecosystem services are different not only between different forest types but also among the plots of the same forest type. A significant change of climate in the Wet Tropics bioregion has been predicted for this century, and the trend and magnitude of climate change impacts (positive and negative) on the different ecosystem services originating from same forest type and also from different forest types will be different under same climate change scenarios and time periods. Therefore, an adaptation approach focusing on ecosystem services urgently needed for the Wet Tropics bioregion.

On the whole, we can conclude that most of the ecosystem services supplied from the rainforests are likely to be reduced in the future due to climate change, while uncertainty exists in case of sclerophyll forests, although only a few ecosystem services have shown to be increased from sclerophyll forests. Along with forest type, the scale and trend (positive or negative) of climate change effect on an ecosystem service will vary with climate change scenarios and time period under consideration.

We utilized NPP as a surrogate to project the supply of ecosystem services. This approach was unable to address many ecological interactions between forested ecosystems ecosystem services, and climate change. Projection using more sophisticated models would provide more precise future trend of the supply of ecosystem services under climate change.

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CONCLUSIONS AND RECOMMENDATIONS

Through this doctoral research, I have contributed to knowledge gaps in the ecosystem service science through: (i) delivering a comprehensive assessment of various ecosystem services across different tropical forest types at the unit and landscape levels, and providing an in-depth understanding of the interactions among multiple ecosystem services in the forested landscape; and (ii) providing evidence of climate change effects on the supply of forest ecosystem services. To achieve the stated aim, I collected forest vegetation attributes data from the Wet Tropics bioregion in northeast Australia, and utilized current spatial datasets about forest distribution for the bioregion. Furthermore, I arranged an expert workshop at James Cook University. In Table A, I provide the key contributions from this doctoral research to knowledge gaps in ecosystem service science against each objective, and describe the implications for sustainable management of ecosystem services from tropical forested landscapes.

Table A. Objectives, key contribution to knowledge gaps and implications to sustain ecosystem services supply from tropical forested landscape.

Objectives	Key contributions to knowledge gaps	Implications
<u>Objective 1:</u> to assess the supply of multiple ecosystem services from rainforests, sclerophyll forests and rehabilitated plantation forests, and how they are spatially distributed across the landscape.		
	➢ Overall, rainforests supply higher multiple ecosystem services followed by rehabilitated plantation forests and sclerophyll forests. ➢ The hotspots of multiple ecosystem services were found in the upland rainforests followed by lowland rainforests and upland sclerophyll forests. ➢ The rehabilitated forests may provide some ecosystem services nearly equivalent to the rainforest. ➢ Ecosystem services supply from the sclerophyll forests are lower, most likely due to their lower structural diversity. ➢ Along the forest structure and type, elevation, rainfall and temperature gradients are the main determinant factors for the quantity of ecosystem services supplied.	• Active conservation of rainforests needs to occur, with more emphasis placed on protecting upland rainforests. • Rehabilitated forests are also worth protecting for ecosystem services supply, and increasing structural diversity in the sclerophyll forests may also enhance ecosystem services supply. • In case of limited resources and options for protection of rainforests, the most important ecosystem services for the community should be identified, and subsequently the hotspots of that ecosystem service supply should be identified and protected, as ecosystem services supply widely varies across forest types and environmental gradients.

Objectives	Key contributions to knowledge gaps	Implications
Objective 2: to investigate the interactions among multiple ecosystem services and how the supply of one ecosystem service is influenced by others.		
	➢ Among the eight examined ecosystem services, increased supply of global climate regulation ecosystem service was found having potentially positive effects on the supply of remaining examined ecosystem services while nutrient regulation ecosystem service supply was emerged having potentially negative effects on the supply of remaining examined ecosystem services. ➢ Although sclerophyll forest plots and rainforest plots occupied in the distinct ordination space but most of the rehabilitated plantation forest plots shared the ordination space with rainforests. ➢ In the higher level of the distance matrix, two distinct clusters were evident among seven ecosystem services. The first cluster represented seven ecosystem services global climate regulation, air quality regulation, erosion regulation, cyclone protection, habitat provision and energy provision, and the second cluster was represented only by nutrient regulation.	• Where the current supplies of examined ecosystem services are lower (except nutrient regulation), management intervention to ensure the supply of habitat provision and global climate regulation ecosystem services will be useful to maximize the supply of multiple ecosystem services. • Along with rainforest and sclerophyll forest conservation, rehabilitated plantation forests may play a considerable supplementary role to maximize the supply of multiple ecosystem services in the heterogeneous forested landscape.
Objective 3: to evaluate the capacities of different Land Use and Land Covers (LULC) to supply ecosystem services, and to identify the spatial congruence between ecosystem services and biodiversity in the landscape.		
	➢ Key and high-potential multiple ecosystem services supply are heterogeneous in the landscape and reduce due to forest disturbances. ➢ A spatial congruence was found between high-potential biodiversity	• If the ecosystem services under consideration are forest-based (e.g. global climate regulation, air quality regulation, and habitat provision) then the integration of high-potential multiple ecosystem

Objectives	Key contributions to knowledge gaps	Implications
	and high-potential global climate regulation ecosystem service in the intact rainforest areas. ➢ A spatial divergence was revealed between high-potential biodiversity and high-potential global climate regulation ecosystem service in the sclerophyll and other disturbed and low tree abundance forested areas. ➢ A spatial congruence exists between certain high-potential multiple ecosystem services (mainly forest-based) and high-potential biodiversity.	services supply and biodiversity conservation is possible in the tropical forested landscape. • Management intervention priorities should focus on increasing tree abundance both in non-tree vegetated land cover areas and within disturbed forested areas to increase the high-potential multiple ecosystem services supply at the landscape level, that will also ensure high-potential biodiversity values in the landscape
<u>Objective 4:</u> to compare the carbon storage among rainforests, degraded rainforests and sclerophyll forests and to determine the driver of carbon storage in the tropical forested landscape.		
	➢ Above ground biomass carbon storage is highest in primary rainforests. ➢ Above ground biomass carbon storage decreases as primary rainforest incurs disturbance. ➢ Degraded forests store more above ground carbon than sclerophyll forests and can store as much as primary rainforest. ➢ Large trees and tree abundance are the major drivers of carbon storage in the tropical forested landscape.	• To maximise above ground carbon storage within the complex landscape of the Wet Tropics bioregion, it is optimal to conserve primary rainforests at sites that experience low levels of disturbance. • Restoring degraded rainforest could result in increased above ground carbon storage in the landscape. • Above ground carbon storage and biodiversity conservation goals can potentially be integrated.

Objectives	Key contributions to knowledge gaps	Implications
<u>Objective 5:</u> to determine the likely effects of climate change on ecosystem services supplied from rainforests and sclerophyll forests		
	➢ Supply of most of the ecosystem services is likely to be reduced from rainforests due to climate change; however, uncertainty exists in case of sclerophyll forests. ➢ The ecosystem service supply from upland rainforests is likely to be more negatively affected than lowland rainforests due to climate change. ➢ The scale and magnitude of climate change effects on ecosystem services likely to vary with three factors- i) ecosystem service type, ii) source of supply i.e. forest type, iii) climate change scenarios	• Adaptation approaches focusing on ecosystem services are urgently needed for the Wet Tropics bioregion especially for the rainforest ecosystem services.

Future Research

In this research I only examined eight ecosystem services using forest vegetation attribute data. Among terrestrial biomes, tropical forests are the most important sources of a number of other ecosystem services such as pollination, ground water recharge, and water purification. Distribution, interaction, processes and climate change effects on these remaining ecosystem services are unknown. It would be interesting to extend this research approach to examine other ecosystem services supplied from the forests of the Wet Tropics bioregion.

In this research I identified the supply and hotspots of several ecosystem services across the landscape. My study did not include the identification of users of those ecosystem services. If it is possible to identify the users of ecosystem services supplied from this forested landscape that will be helpful for more effective natural resource management planning.

The central concept of ecosystem services is human well-being. In this study I evaluated ecosystem services using ecological attribute data. Therefore, we know the spatial distribution of ecosystem services and how they are ecologically important to sustain the supply of other ecosystem services. It would be very much interesting to examine how human communities living in this region rank important ecosystem services supplied from the forested landscape, and which areas are important for them and for particular ecosystem services. It would then be possible to overlay these maps of ecological importance and social importance of ecosystem services supply to determine the congruence and divergence between ecological preferences and social preferences.

This research has potential to link with species distribution modeling studies that are currently being undertaken for the bioregion. It would be interesting to make ecosystem services supply map under different climate change scenarios and compare these maps with biodiversity maps under different climate change scenarios.

Finally, as my research has revealed the trends of climate change effects on ecosystem services, it would be useful to extend this research to determine the appropriate adaptation options for each ecosystem services under future climate change scenarios.

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Appendices

Appendix S1: Articles that were reviewed in the meta-analysis (Chapter 1).

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Appendix S2: Supplementary material (Chapter 2) - forest description (Tracey, 1982; Webb, 1959; WTMA, 2014).

No.	Forest	Description
1	Mesophyll forest	The rainforests with dominant canopy leaf blade length of more than 12.5 cm, generally occur in very wet to moist low land.
2	Microphyll forest	The rainforests with dominant canopy leaf blade length of less than 7.5cm generally occur in wet highland areas.
3	Notophyll forest	The rainforests with dominant canopy leaf blade length of 7.5-12.5cm. This is the most extensive rainforests in this region ranging from foot hills to upland.
4	Sclerophyll forest	Eucalyptus forests and wood land. This class includes: Sclerophyll forest-*Acacia* dominant, Sclerophyll woodland and forest-*Acacia* dominant, Sclerophyll woodland and forest-*Allocasuarina* or *Casuarina* dominant, Sclerophyll woodland and forest-*Eucalyptus* dominant (including *Corymbia*), Sclerophyll woodland and forest-*Eucalyptus* dominant (including *Melaleuca*), Sclerophyll woodland and forest-*Lophostemon* dominant, Sclerophyll woodland and forest-*Melaleuca* dominant, Sclerophyll woodland and forest-*Syncarpia* dominant.
5	Heath & shrubland	Open to close scrub and heath. This class also includes: Shrublands with emergent trees and low shrub by woodlands with *Melaleuca viridiflora*, *Melaleuca minutifolia* subsp. *monantha*, *Acacia flavescens*, *Grevillea glauca*, *Grevillea coriacea*, *Petalostigma pubescens*, *Corymbia clarksoniana*, *Corymbia dallachiana*, *Eucalyptus crebra*, *E. portuensis*, *Allocasuarina littorali*
6	Disturbed original rainforest	Disturbed areas that originally supported rainforest vegetation.
7	Disturbed original sclerophyll forest	Disturbed areas that originally supported sclerophyll vegetation.
8	Rehabilitated plantation forest	Plantation of native trees for restoration.

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Appendix S3: Supplementary material 1 (Chapter 3). Description of land cover category (Tracey, 1982; Webb, 1959; WTMA, 2014).

Land cover	Description
Mesophyll forest	The rainforests with dominant canopy leaf blade length of more than 12.5 cm, generally occur in very wet to moist low land. This class includes all sub classes of mesophyll forests –complex mesophyll vine forests, mesophyll vine forests, and semi deciduous mesophyll vine forests.
Microphyll forest	The rainforests with dominant canopy leaf blade length of less than 7.5cm, generally occur in wet highland areas. This class includes all subclasses of microphyll forests- simple microphyll vine forests, microphyll vine thickets, deciduous microphyll thickets, and microphyll fern thickets.
Notophyll forest	The rainforests with dominant canopy leaf blade length of 7.5-12.5cm. This is the most extensive rainforests in this region ranging from foot hills to upland. This class includes all subclasses of notophyll forests-complex notophyll vine forests, notophyll vine forests, notophyll vine thickets, simple notophyll vine forests, semi-evergreen notophyll vine forests, semi-evergreen notophyll vine thickets, and semi-deciduous notophyll vine forests
Sclerophyll forest	Eucalyptus forests and wood land. This class includes: Sclerophyll forest-*Acacia* dominant, Sclerophyll woodland and forest-*Acacia* dominant, Sclerophyll woodland and forest-*Allocasuarina* or *Casuarina* dominant, Sclerophyll woodland and forest-*Eucalyptus* dominant (including *Corymbia*), Sclerophyll woodland and forest-*Eucalyptus* dominant (including *Melaleuca*), Sclerophyll woodland and forest-*Lophostemon* dominant, Sclerophyll woodland and forest-*Melaleuca* dominant, Sclerophyll woodland and forest-*Syncarpia* dominant.
Mangrove forest	Mangrove Forest. Main components: medium closed mangrove forest (*Rhizophora* spp., *Bruguiera* spp., etc) and scrub (*Avicennia eucalyptifolia*, *Ceriops* spp.)
Heath & shrubland	Open to close scrub and heath. This class also includes: Shrublands with emergent trees and low shrub by woodlands with *Melaleuca viridiflora*, *Melaleuca minutifolia* subsp. *monantha*, *Acacia flavescens*, *Grevillea glauca*, *Grevillea coriacea*, *Petalostigma pubescens*, *Corymbia clarksoniana*, *Corymbia dallachiana*, *Eucalyptus crebra*, E. *portuensis*, *Allocasuarina littorali*
Disturbed original rainforest	Extremely disturbed areas that originally supported rainforest vegetation. The original canopy has been entirely removed and the original vegetation type cannot be determined from the present species composition.
Disturbed original sclerophyll forest	Extremely disturbed areas that originally supported sclerophyll vegetation, and including areas that have regrown from complete clearing. The original canopy has been entirely removed and the original vegetation type cannot be determined from the present species composition.
Grassland, fernland & sedgeland	This class includes: fernland, fernland-sedgeland, grassland, grassland and sedgeland, sedgeland'
Unvegetated (natural)	Unvegetated, largely rocks, alluvial deposits
Rehabilitation plantation_ native species	Plantation of native trees for restoration
Acacia dominant shrubland	Low woodlands to tall shrublands dominated by Acacia spp. on residuals.
Vegetation complexes and mosaic	Low to medium woodland, open forest, closed forest, or mosaics of all three. Complex of open to closed shrublands, low to medium woodlandds and forests and grasslands.
Mineral extraction sites	Quarry or bare areas resulting from mining activities
Open shrublands-exotics dominant	Non-remnant vegetation dominated by exotics-areas of significant vegetation coverage with a high percentage of exotic species in the canopy and/or the understory and ground cover.

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Appendix S4: Supplementary material 2 (Chapter 3). List of ecological integrity and ecosystem service components with rationales (after Burkhard et al. 2012; de Groote et al., 2010; MA, 2005).

Components	Rationales
Ecological integrity	
Abiotic heterogeneity	The provision of suitable habitats for different species, for functional groups of species and for processes is necessary for functioning of ecosystems
Biodiversity	The presence and absence of selected species, groups of species and species composition
Biotic water flows	Referring to the water cycling affected by plant processes in the system
Metabolic efficiency	Referring to the amount of energy necessary to maintain a specific biomass also serving as a stress indicator for the system
Energy capture	The capability of ecosystems to enhance the input of usable energy. In ecosystems the captured energy is used to build up biomass (by primary production) and structures
Reduction of nutrient loss	Referring to the irreversible output elements of the systems, the nutrient budget and matter flows
Storage capacity	Referring to the nutrients, energy and water budgets of the systems and the capacity to store them when available and to release them when needed
Regulating ecosystem services	
Local climate regulation	Changes in land cover can locally affect temperature, wind, radiation and precipitation
Global climate regulation	Ecosystem can play an important role in climate by either sequestering or emitting greenhouse gasses
Flood protection	Natural elements dampening extreme flood events
Groundwater recharge	The timing and magnitude of runoff, flooding, and aquifer recharge can be strongly influenced by changes in land cover, including in particular, alterations that change the water storage potential of the system such as the conversation of the wetlands or the replacement of the forests with croplands or croplands with urban areas
Air quality regulation	The capacity of ecosystems to remove toxic and other elements from the atmosphere
Erosion regulation	Vegetative cover are useful to prevent landslides and soil retention
Nutrient regulation	The capacity of ecosystems to carry out (re) cycling nutrients
Water purification	Ecosystems possess the capacity to purify water but can also be a source of impurities in fresh water
Pollination	Ecosystem changes affect the distribution, abundance and effectiveness of pollinators. Wind and bees are in charge of the reproduction of a lot of culture plants
Cyclone protection	The presence of coastal ecosystems such as mangroves can dramatically reduce the damage caused by cyclones
Provisioning ecosystem services	
Livestock	Keeping of edible animals
Fodder	Cultivation and harvest of animal fodder
Capture fisheries	Catch of commercially interesting fish species, which are accessible for fisherman
Wild foods	Harvest of e.g. berries, mushrooms, wild animal hunting or fishing
Timber	Presence of trees or plants with potential use of timber
Wood fuel	Presence of trees or plants with potential use as fuel
Energy (biomass)	Presence of trees or plants with potential use as energy source
Biochemical and medicines	Production of biochemical and medicine
Freshwater	Presence of fresh water
Habitat	Importance of ecosystem to provide habitat for species, natural biodiversity (particularly endemic species)
Cultural ecosystem services	

Recreation and aesthetic values	Visual qualities of the area (scenery, scenic beauty).
Intrinsic value of biodiversity	The value of nature and species themselves, beyond economic or human benefits

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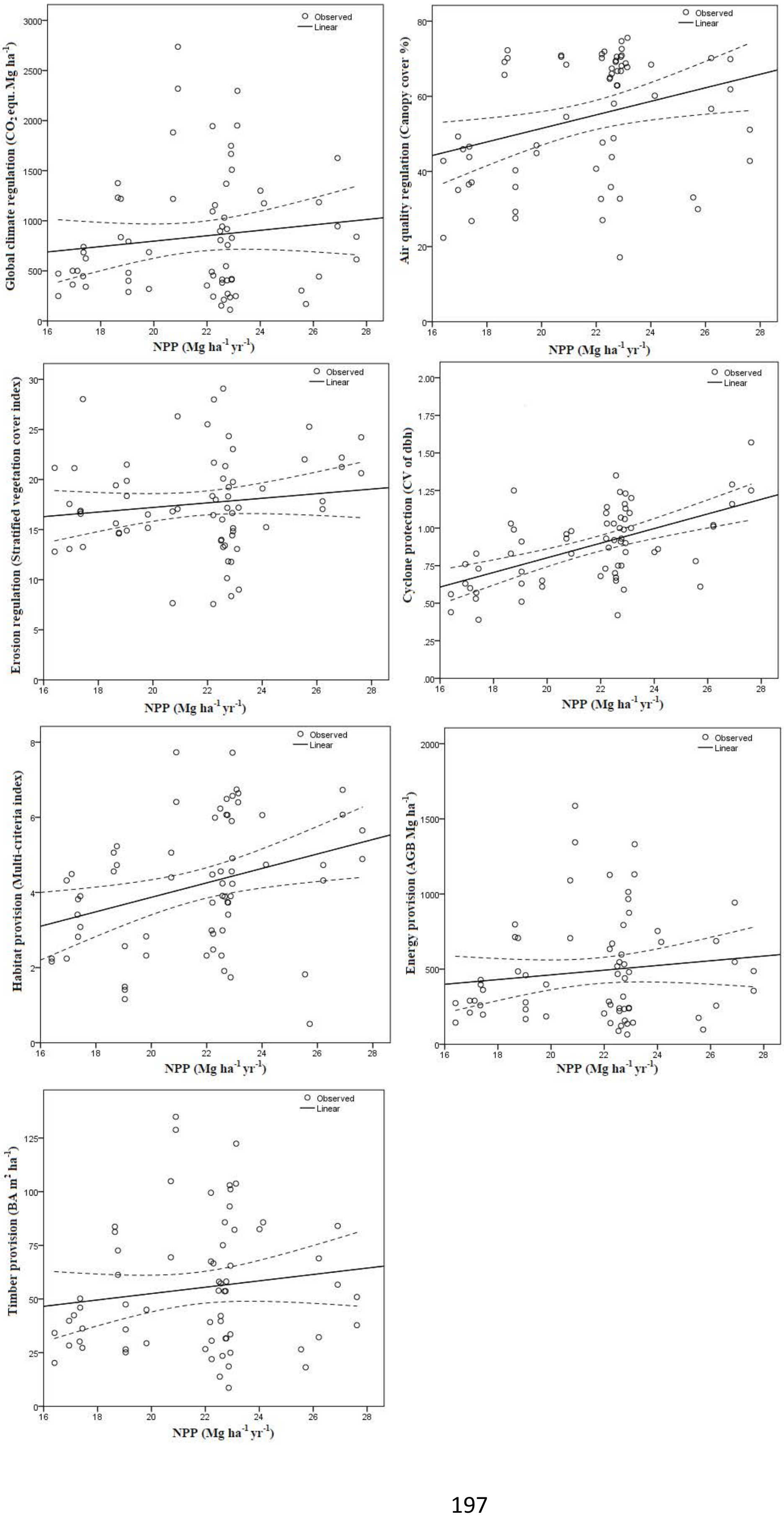
Appendix S5: Supplementary material (Chapter 4). Plot details.

Plot no.	Plot sites	Forest types	Geology	Elevation (m)	Mean annual rainfall (mm)	Mean annual temperature (OC)
1	Clump mountain national park 1	Rainforests	Fine sedimentary	53	3115	23.70
2	Clump mountain national park 2	Rainforests	Fine sedimentary	54	3115	23.70
3	Mount lewis national park 1	Rainforests	Fine sedimentary	678	1767	22.60
4	Mount lewis national park 2	Rainforests	Fine sedimentary	656	1767	22.60
5	Curtain fig national park 1	Rainforests	Alluvium	734	2030	20.70
6	Curtain fig national park 2	Rainforests	Alluvium	732	2030	20.70
7	Danbulla national park 1	Rainforests	Alluvium	761	2223	20.80
8	Danbulla national park 2	Rainforests	Gneiss	733	2131	20.70
9	Moresby range national park 1	Rainforests	Gneiss	62	3423	23.70
10	Crater lakes national park 1	Rainforests	Alluvium	756	2171	20.60
11	Crater lakes national park 2	Rainforests	Alluvium	762	2171	20.60
12	Mount hypipamee national park 1	Rainforests	Alluvium	988	1478	19.50
13	Mount hypipamee national park 2	Rainforests	Fine sedimentary	974	1478	19.50
14	Kuranda national park 1	Rainforests	Gneiss	369	2177	23.10
15	Kuranda national park 2	Rainforests	Gneiss	365	2177	23.10
16	Wongabel state forest 1	Rainforests	Alluvium	775	1797	20.70
17	Wongabel state forest 2	Rainforests	Alluvium	776	1797	20.70
18	Mowbray national park 1	Rainforests	Gneiss	54	2048	25.10
19	Barron gorge national park 1	Rainforests	Gneiss	388	2209	22.70
20	Barron gorge national park 2	Rainforests	Gneiss	412	2221	22.70
21	Goldsborough vally state forest 2	Rainforests	Gneiss	731	2460	20.90
22	Herberton range forest reserve 1	Rainforests	Andesite	845	1463	20.00
23	Herberton range forest reserve 2	Rainforests	Andesite	853	1463	20.00
24	Restoration plantation 3	Rainforests	Sands	350	2102	23.10
25	Kuranda state forest 1	Rehabilitated plantation	Granite	455	2118	22.60
26	Kuranda state forest 2	Rehabilitated plantation	Granite	459	2133	22.80
27	Goldsborough vally state forest 1	Rehabilitated plantation	Granite	730	2426	20.80
28	Restoration plantation 2	Rehabilitated plantation	Granite	349	2144	23.10
29	Restoration plantation 1	Rehabilitated plantation	Granite	350	2144	23.10
30	Restoration plantation 4	Rehabilitated plantation	Granite	364	2102	23.10
31	Millstream falls national park 1	Sclerophyll forests	Alluvium	826	1517	20.30
32	Millstream falls national park 2	Sclerophyll forests	Alluvium	825	1517	20.30
33	Moresby range national park 2	Sclerophyll forests	Gneiss	68	3423	23.70
34	Little mulgrave national park 1	Sclerophyll forests	Gneiss	189	2933	23.80
35	Little mulgrave national park 2	Sclerophyll forests	Gneiss	168	2874	23.40
36	Russell river national park 1	Sclerophyll forests	Siliceous	17	3817	24.00
37	Russell river national park 2	Sclerophyll forests	Siliceous	12	3817	24.00
38	Herberton range national park 1	Sclerophyll forests	Fine sedimentary	980	1311	19.60
39	Herberton range national park 2	Sclerophyll forests	Fine sedimentary	977	1311	19.60
40	Smithfield conservation park 2	Sclerophyll forests	Laterite	68	2139	24.20
41	Smithfield conservation park 1	Sclerophyll forests	Laterite	57	2139	24.20
42	Davies creek national park 1	Sclerophyll forests	Fine sedimentary	494	2163	22.30
43	Davies creek national park 2	Sclerophyll forests	Fine sedimentary	491	2163	22.30
44	Dinden state forest 1	Sclerophyll forests	Gneiss	412	2115	22.70
45	Dinden state forest 2	Sclerophyll forests	Gneiss	438	2095	22.40
46	Ravenshoe forest reserve 1	Sclerophyll forests	Andesite	818	1517	20.30
47	Ravenshoe forest reserve 2	Sclerophyll forests	Alluvium	810	1517	20.30
48	Herberton range state forest 2	Sclerophyll forests	Fine sedimentary	1032	1274	19.20
49	Mowbray national park 2	Sclerophyll forests	Gneiss	68	2086	24.60
50	Dinden national park 1	Sclerophyll forests	Fine sedimentary	422	2176	22.50
51	Dinden national park 2	Sclerophyll forests	Fine sedimentary	419	2176	22.50
52	Baldy mountain state forest 1	Sclerophyll forests	Andesite	803	1627	20.60
53	Baldy mountain state forest 2	Sclerophyll forests	Andesite	822	1627	20.60
54	Danbulla state forest 1	Sclerophyll forests	Fine sedimentary	651	2024	21.30
55	Danbulla state forest 2	Sclerophyll forests	Fine sedimentary	659	1990	21.10
56	The bluff state forest 1	Sclerophyll forests	Fine sedimentary	895	1253	20.20
57	The bluff state forest 2	Sclerophyll forests	Fine sedimentary	907	1253	20.20
58	The bluff forest reserve 1	Sclerophyll forests	Andesite	935	1192	19.80
59	The bluff forest reserve 2	Sclerophyll forests	Andesite	936	1192	19.80
60	Tumuolin state forest 1	Sclerophyll forests	Andesite	1023	1301	19.40
61	Tumuolin state forest 1	Sclerophyll forests	Alluvium	1028	1301	19.40
62	Bilwon state forest 2	Sclerophyll forests	Gneiss	390	2035	22.90
63	Kuranda 1	Sclerophyll forests	Gneiss	355	2102	23.10
64	Kuranda 2	Sclerophyll forests	Gneiss	358	2089	23.10

Appendix S6: Supplementary material (Chapter 6). Automatically calibrated regression models between ecosystem services and NPP. Dashed line indicates 95% confidence intervals of mean.



Appendix S7: Reprint of Alamgir, M., Pert, P.L., Turton, S.M. (2014) A review of ecosystem services research in Australia reveals a gap in integrating climate change and impacts on ecosystem services. International Journal of Biodiversity Science, Ecosystem Services & Management 10 (2), 112-127.

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REVIEW ARTICLE

A review of ecosystem services research in Australia reveals a gap in integrating climate change and impacts on ecosystem services

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Ecosystem services (ES) are the benefits people obtain from ecosystems. A substantial part of human well-being is dependent on the sustainable flow of ES. Climate change, economic growth and an increasing human population has placed greater pressures on global ES. Australia's ecosystems are among the most vulnerable sectors to climate change. Hence, a comprehensive review is necessary to explore ES research that integrates climate change impacts. Our review reveals that ES research in Australia, stimulated in the early 2000s, has continued to increase consistently after the Millennium Ecosystem Assessment. Australian ES research has primarily focused on the impact of land-use change and management, policy and governance issues, but less on the impact of climate change on ES. Climate change models show that climate will threaten most of the main ES in Australia by 2050. For the sustainable management of these ES – incorporating climate change – ecosystem and ES specific adaptations are suggested as the best sustainable policy tools for the future. Therefore, further research needs to incorporate climate change and ES for evidence-based sustainable management of Australia's ES. We provide the following recommendations for future ES research: (i) evaluating the extent and trend of climate change impacts on ES through consideration of different climate change scenarios; (ii) preparing vulnerability maps of important ES that are likely to be sensitive to climate change and (iii) developing ecosystem and ES specific adaptations to climate change that involve key stakeholders.

Keywords: climate change; ecosystem services; adaptation; stakeholder involvement; land-use change; Australia

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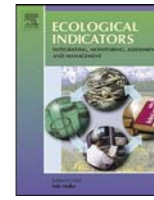

Assessing regulating and provisioning ecosystem services in a contrasting tropical forest landscape

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Global climate regulation

ABSTRACT

Ecosystem services are the bridge between nature and society, and are essential elements of community well-being. The Wet Tropics Australia, is environmentally and biologically diverse, and supplies numerous ecosystem services. It contributes to the community well-being of this region, Australian national economy and global climate change mitigation efforts. However, the ecosystem services in the region have rarely been assessed undermining strategic landscape planning to sustain their future flow. In this study, we attempted to: (i) assess the quantity of five regulating ecosystem services – global climate regulation, air quality regulation, erosion regulation, nutrient regulation, and cyclone protection, and three provisioning ecosystem services – habitat provision, energy provision and timber provision across rainforests, sclerophyll forests and rehabilitated plantation forests; (ii) evaluate the variation of supply of those regulating and provisioning ecosystem services across environmental gradients, such as rainfall, temperature, and elevation; (iii) show the relationships among those ecosystem services; and (iv) identify the hotspots of single and multiple ecosystem services supply across the landscape. The results showed that rainforests possess a very high capacity to supply single and multiple ecosystem services, and the hotspots for most of the regulating and provisioning ecosystem services are found in upland rainforest followed by lowland rainforest, and upland sclerophyll forest. Elevation, rainfall and temperature gradients along with forest structure are the main determinant factors for the quantity of ecosystem services supplied across the three forest types. The correlation among ecosystem services may be positive or negative depending on the ecosystem service category and vegetation type. The rehabilitated plantation forests may provide some ecosystem services comparable to the rainforest. The results demonstrated disturbance regimes (such as tropical cyclones) may have influenced the usual spatial trend of ecosystem service values. This study will assist decision makers in incorporating ecosystem services into their natural resource management planning, and for practitioners to identify the areas with higher values of specific and multiple ecosystem services.

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