THE UNIVERSITY OF QUEENSLAND

AUSTRALIA

Environmental Consequences of Land use Changes for Bioenergy Crop Production at a Regional Scale

Saori Miyake

Bachelor of Home Economics, Master of Town Planning (M.TPL)

A thesis submitted for the degree of Doctor of Philosophy at

The University of Queensland in 2013

The School of Geography, Planning and Environmental Management

Abstract

Sustainability has become a focus of global bioenergy (especially biofuel) policy and research over the past few years. Due to the rapid expansion of demand for global bioenergy and increased sustainability requirements, land use change associated with bioenergy crops (hereafter referred to as 'bioenergy-driven land use change') has emerged as an important research topic. The use of degraded land, marginal and abandoned agricultural lands (hereafter referred to as 'underutilised agricultural land') has been proposed for non-food and lignocellulosic crop production. However, the implications and consequences of the use of these lands remain highly uncertain. This was a study to evaluate the regional environmental effects of the use of underutilised agricultural land for bioenergy production using a spatial modelling approach and land use change scenarios in a case study region—the Burnett River catchment, Australia.

The aim of this research was to evaluate whether land use change scenarios that involve bioenergy crop production on underutilised agricultural land can enhance regional-scale environmental outcomes when compared with current land uses, and to provide recommendations and suggestions for future land use options. The scenarios assessed in this research, including the selection of bioenergy crops (and the associated management practices) and land use change pathways, were developed to minimise potential environmental impacts. In line with four research objectives based on the aim, a review of past studies relating to bioenergy-driven land use changes was conducted to better understand their dynamics and their effects both in the past, and in future projections (Objective 1, Chapter 2). A spatially explicit evaluation framework was developed to evaluate regional scale environmental consequences associated with land use changes (Objective 2, Chapter 3). The evaluation framework focused on the issues of water quantity and quality (and soil erosion), and terrestrial biodiversity, the impacts of which are commonly experienced regardless of geographical region. It was tested in a case study region in subtropical Queensland, Australia (Objective 3, Chapter 5), and then applied to six bioenergy-driven land use change scenarios to quantify the impacts and compare the results with a baseline (2005/06) and with other scenarios (Objective 4, Chapter 6). The findings from the evaluation were synthesised as recommendations for future bioenergy sustainability research community and policy, and also as a basis for decision making concerning sustainable land use options for future bioenergy production (Chapter 6 and 7).

The results of the land use change scenario evaluation indicated that bioenergy crop scenarios could benefit regional environmental quality in the case study region only when:

(i) open grazing areas (pastures) were used as the plantation site,

(ii) native woody perennial bioenergy crops were used (e.g. Pongamia [Millettia pinnata] or Short Rotation Coppice [SRC] eucalyptus species), and

(iii) the new plantations were under low intensity management (similar to conventional forested grazing areas or conventional forestry).

The results also suggested that current bioenergy policy—simply limiting crop production to underutilised agricultural land—will not necessarily enhance environmental sustainability in bioenergy production. They also indicated that future policy could address more detailed prescriptions including very careful site planning and management strategies. These included: (i) selection of the most suitable crops and site locations (i.e. the land use change pathway most suited to a region); (ii) stringent control of native vegetation clearing and consideration of the local ecology when deciding on a location (e.g. appropriate distance from watercourses and conservation areas, spatial configuration of native vegetation and local ecology); and (iii) identification and implementation of the optimum management intensity. These are perhaps the most significant findings of the study and they have added important knowledge to current bioenergy research and policy.

Another important contribution of this research was the evaluation framework that offers a methodology potentially applicable to various regions for land use change scenario evaluation and could support for decision making on bioenergy land use at the regional scale. The need for such an evaluation framework is becoming more important, as there is an increasing need worldwide for decision making related to land use change for bioenergy crops. While land use change scenarios vary significantly between countries and regions, the evaluation framework could be tailored to specific regions in future applications. Future development of the framework could also be undertaken in accordance with the development of sustainability indicators by accredited certification organisations/initiatives. A well-planned and integrated bioenergy industry can have major environmental, economic and social benefits for a region.

Declaration by author

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

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

I acknowledge that an electronic copy of my thesis must be lodged with the University Library and, subject to the General Award Rules of The University of Queensland, immediately made available for research and study in accordance with the *Copyright Act 1968*.

I acknowledge that copyright of all material contained in my thesis resides with the copyright holder(s) of that material. Where appropriate I have obtained copyright permission from the copyright holder to reproduce material in this thesis.

Publications during candidature

Peer-reviewed paper:

Miyake, S., Renouf, M., Peterson, A., McAlpine, C., Smith, C., 2012, 'Land-use and environmental pressures resulting from current and future bioenergy crop expansion: A review', *Journal of Rural Studies*, vol. 28, no. 4, pp. 650-658.

Conference presentations:

Miyake, S., 2011a, 'Framework for evaluating the environmental consequences of bioenergy-driven land-use changes at local and regional scales'. *Bioenergy Australia 2011 conference*. 23-25 November 2011.Sunshine Coast, Queensland, Australia.

Miyake, S., 2011b, 'Framework for evaluating the environmental consequences of bioenergy-driven land-use changes at local and regional scales'. *IEA bioenergy workshop on quantifying and managing land use effects of bioenergy*. 19-21 September 2011, Campinas, Brazil.

Poster presentation:

Miyake, S., 2010, 'Land-use and environmental pressures resulting from the recent and future bioenergy crop expansion (poster presentation)'. *Bioenergy Australia 2010 conference*. 8-10 December 2010. Sydney, Australia.

Publications included in this thesis

Publication 1: Incorporated as Chapter 2

Miyake, S., Renouf, M., Peterson, A., McAlpine, C., Smith, C., 2012, 'Land-use and environmental pressures resulting from current and future bioenergy crop expansion: A review', *Journal of Rural Studies*, vol. 28, no. 4, pp. 650-658.

Contributor	Statement of contribution
Saori Miyake (Candidate)	Research design and implementation (80%), Wrote and edited the paper (80%)
Ann Peterson	Edited the paper (5%)
Marguerite Renouf	Advice on research design (10%), Edited the paper (5%)
Clive McAlpine	Advice on research design (10%), Edited the paper (5%)
Carl Smith	Edited the paper (5%)

Publication 2 and 3: Incorporated as Chapter 3

Miyake, S., 2011a, 'Framework for evaluating the environmental consequences of bioenergy-driven land-use changes at local and regional scales'. *Bioenergy Australia 2011 conference*. 23-25 November 2011.Sunshine Coast, Queensland, Australia.

Miyake, S., 2011b, 'Framework for evaluating the environmental consequences of bioenergy-driven land-use changes at local and regional scales'. *IEA bioenergy workshop on quantifying and managing land use effects of bioenergy*. 19-21 September 2011, Campinas, Brazil.

Contributor	**Statement of contribution**
Saori Miyake (Candidate)	Research design and implementation (65%), Presentation (100%)
Carl Smith	Advice on research design (20%)
David Waters	Advice on research design (10%)
Clive McAlpine	Advice on research design (5%)

Contributions by others to the thesis

Contributor	**Statement of contribution**
Saori Miyake (Candidate)	Research design, implementation, writing and editing (65%)
Ann Peterson	Advice on research design and editing (10%)
Carl Smith	Advice on research design and editing (7.5%)
Marguerite Renouf	Advice on research design and editing (7.5%)
Clive McAlpine	Advice on research design and editing (5%)
David Waters	Advice on hydrological modelling (5%)
Judith Millington	Proofreading of the thesis

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

None

Acknowledgements

This thesis would never have been completed without many people's advice, support, and encouragement. It was one of the most challenging four and a half years of my life, but the most worthwhile experience at the same time. I was always lucky to get help throughout the journey.

First, I would like to express my sincere gratitude to my thesis supervisory team: Ann Peterson; Carl Smith; Marguerite Renouf; and Clive McAlpine. I believe that I could not have finished my thesis without Ann's continuous support, encouragement, and patience in reviewing and editing my writing so many times. Carl's strong expertise and novel ideas, backed up by his knowledge and experience helped me tremendously when finalising the methodology. Without Marguerite's inspiration and encouragement, I would never have imagined myself starting this journey, which would eventually take me to new chapter in life. Her thorough and critical comments were always the greatest assets that have enhanced the overall quality of my work. Thanks to Clive's high expertise in local ecology and extensive experience in scientific research and successful RHD student supervision, I have been more confident that I was on the right track since he joined my supervisory team.

I also received enormous support from people outside the team. I would particularly like to express my thanks to David Waters (Principal Scientist, Queensland Department of Natural Resources and Mines [DNRM]), who helped me with hydrological modelling. This was the hardest part in my project, and David spent a great deal of time teaching and explaining it to me, checking what I did, providing data and parameters, and enabling communications with other agencies. I also express my thanks to David Burton (Senior Spatial Information Officer, DNRM) who provided me with various technical support and advice whenever problems occurred in learning and implementing the models. Noam Levin (Senior Lecturer, Department of Geography, The Hebrew University of Jerusalem) and my colleague Alvaro Salazar Perez also helped me a great deal in GIS and other spatial modelling (*Fragstats*, and IDRISI) when I was had tremendous problems with the technology.

I would like to thank Leonie Seabrook (Postdoctoral Research Fellow, GPEM, UQ), Malcolm Wegener (Honorary Research Fellow, School of Agriculture and Food Sciences, UQ), and John Armston (Senior Scientist [remote sensing], Queensland Department of Science, Information Technology, Innovation and the Arts [DSITIA]) for technical advice and discussions when I was establishing the methodology.

Throughout my project, the Remote Sensing Centre currently at the Queensland DSITIA helped me in acquisition of crucial spatial data, in particular: Tony Gill (currently Remote Sensing Scientist/Programmer, NSW Department of Premier and Cabinet), Dan Tindall (Principal Scientist [Remote Sensing], DSITIA), and Craig Shephard (A/Principal Scientist [Remote Sensing], DSITIA). Without the latest data and your technical advice, I would not have been able to complete the evaluations in my thesis.

My gratitude also extends to bioenergy researchers: Annette Cowie (Director, Rural Climate Solutions University of New England/ NSW Department of Primary Industries) for her introduction to the international bioenergy research community; Peter Gresshoff (Director and Professor) and Paul Schott (Research Fellow) at the ARC Centre of Excellence for Integrative Legume Research, UQ, for their expertise and various data on Pongamia; David Lee (A/Professor, Forest Industries Research Centre, University of the Sunshine Coast) for his knowledge and advice on eucalypt species; Birka Wicke (Assistant Professor, the Copernicus Institute of Sustainable Development, Utrecht University) for her support and valuable information on cutting-edge European research in bioenergy sustainability; and Ian O'Hara (Principal Research Scientist, Centre for Tropical Crops and Biocommodities, QUT) for his information on sweet sorghum trials in Queensland.

During the past years I made a few trips to the case study region, and I acknowledge that they became very meaningful for my research thanks to Ian Crosthwaite (Senior Agronomist, BGA AgriServices), Damien O'Sullivan (Senior Extension Officer, former Queensland Department of Employment, Economic Development and Innovation [DEEDI]), and Cathy Mylrea (Burnett Mary Regional Group).

I would also like to thank the staff at Geography, Planning and Environmental Management (GPEM) for their daily support, particularly Alan, Jurgen Tineke, Nivea, Judy, Claire, and Lorraine. Their daily administrative support greatly facilitated the progress of my research.

Lastly, I would like to thank my family in Japan for their belief in my reckless challenge in Australia over the years, and all my friends I met during my journey who have shared so many precious experience and moments. Thanks very much my beautiful friends, Lara, Birte, Julie, Izumi, Ian, Nikola, Jackie, Aki, Chrystal, Kelsey, Nivea, Melissa, Francisco, Aude, Heidi, Andrew, Juliane, Miranda, Missaka, Cintia, Martina, Payal, Ezgi, Merlin, John and Lauren for your friendship, support and great conversations over lunch time, weekends and evenings that were often inspiring, lit up my heart and coloured my life and memories of Brisbane.

Keywords

bioenergy, biofuels, land use change, deforestation, abandoned agricultural land, marginal and degraded land, eucalypts, Pongamia

Australian and New Zealand Standard Research Classifications (ANZSRC)

ANZSRC code: 120504, Land Use and Environmental Planning, 40%
ANZSRC code: 070104, Agricultural Spatial Analysis and Modelling, 40%
ANZSRC code: 050209, Natural Resource Management, 20%

Fields of Research (FoR) Classification

FoR code: 0502, Environmental Science and Management, 90%
FoR code: 0703, Crop and Pasture Production, 10%

Table of Contents

List of Figures

List of Tables

List of Abbreviations

ABARES	The Australian Bureau of Agricultural and Resource Economics and Sciences
BCV_{LU}	Biodiversity conservation value corresponding to native vegetation or land use class
BMRG	The Burnett Mary Regional Group
BRS	The Bureau of Rural Science
CA	Class Area
dLUC	Direct land use change
EMC/DWC	Event Mean Concentration/Dry Weather Concentration
FPC	Foloage Projective Cover
FU	Function Unit
GBR	Great Barrier Reef
GHG	Greenhouse gas
ha	hectare
iLUC	Indirect land use change
LCA	Life Cycle Assessment
LPI	Largest Patch Index
MSA	Mean Species Abundance
NP	Number of patches
QLUMP	Queensland Land Use Mapping Program
RE	Regional Ecosystem
SRC	Short Rotation Coppice
SLATS	Statewide Landcover and Trees Study
TN	Total nitrogen
TP	Total phosphorous
TSS	Total suspended solids

Chapter 1: INTRODUCTION

1.1 Background

There is a growing trend worldwide to look for renewable energy sources. Bioenergy is one alternative to fossil fuel. Since the mid-2000s, a growing number of governments have encouraged the production and use of bioenergy, especially biofuels, due to the potential benefits for climate change mitigation, improved energy security and regional development (IEA 2004; European Commission 2009; U.S. EPA 2010). As a result, global fuel ethanol production doubled to approximately 92,000 million litres between 2005 and 2009, while global biodiesel production increased four-fold to 17,200 million litres during the same period (OECD & FAO 2012). By 2020, global biofuel production is expected to reach 155,000 million litres for fuel ethanol and 41,900 million litres for biodiesel (OECD & FAO 2012).

Bioenergy encompasses a wide range of energy products including bioelectricity and heat, liquid and gaseous fuels (e.g. biogas, methanol, ethanol, butanol, biodiesel and bio-oil) and biochar, which is produced from biomass sources including agricultural crops and their residues, forests and their residues, grasses, wood processing waste, woody weeds, oil-bearing plants, animal manures, sewage and the organic fraction of municipal solid waste (Diesendorf 2007). This research focused on bioelectricity and biofuels (mainly ethanol and biodiesel used for transport fuel)[1] generated from agricultural and forest crops, plants and their residues. The scope of this research was defined by the fact that the electricity and transport sectors have recently been responsible for the majority of greenhouse gas (GHG) emissions in developed countries, and these countries are currently the world's major emitters of GHG. For example, these sectors combined are currently responsible for 61% of the total GHG emissions in the United States of America (U.S.A.) (U.S. EPA 2013), almost half of the emissions in Australia (Commonwealth of Australia 2013) and also almost half of those in the European Union (EU) (EEA 2013). In addition, the energy demand for these sectors is expected to multiply worldwide over the next decades due to the fast-growing economies of countries such as China and India.

[1] Biofuels incorporate bioethanol and biodiesel in both a pure form and as blends with fossil fuels.

Bioenergy was initially thought to have large potential environmental and social advantages over fossil fuels. However, the production and use of biofuels is now the subject of lively debate about its sustainability. A rapid increase in biofuel production from conventional agricultural crops has led to an array of environmental and social concerns. Its advantages and disadvantages over fossil fuels have been discussed widely over the past several years. Previous environmental assessments in the 2000s predominantly investigated the energy balance and GHG intensity of biofuels from different feedstocks, mostly using environmental life cycle assessment (LCA) methodology (Quirin et al. 2004; von Blottnitz & Curran 2007; Worldwatch Institute 2007; Zah et al. 2007; OECD & International Transport Forum 2008). One of their most significant findings was to clearly identify limitations for biofuels sourced from conventional agricultural food crops, using existing conversion pathways, namely 'first-generation' biofuels. Thereafter the sustainability of biofuels started to be questioned in the international arena (Doornbosch & Steenblik 2007) To date, corn/maize, wheat, sugarcane, sugar beet, sweet sorghum and cassava have been major crops used for fuel ethanol production, while soybeans, canola/rapeseed, and oil palm have been used for biodiesel production (de Vries et al. 2010). As these crops are key food and feed crops, growing demand for biofuel was blamed for the 2007-2008 global food crisis in the escalating 'food versus fuel' debate (Mitchell 2008).

In response, technological developments are focused on non-food feedstock and improving production efficiencies to mitigate the pressures on land resources associated with bioenergy crop production. They include second-generation lignocellulosic conversion technologies and biofuels from third generation high yield algae. However, technological breakthroughs and high production costs have been the greatest challenges for commercialisation of these technologies (Sims et al. 2008; Singh & Olsen 2011). Despite recent bioenergy policy indicating the phase-out of first-generation biofuels (e.g. EU by 2020), they still need to serve as a bridge to the full commercialisation of other emerging technologies. In parallel, there has been research and development in high yield dedicated energy crop and plant species for future bioenergy feedstocks such as non-food oil crops, including Pongamia (*Milletta pinnata*), jatropha (*Jatropha curcus*) and lignocellulosic crops such as. switchgrass (*Panicum virgatum*), miscanthus (*Miscanthus x giganteus*), short rotation coppice (SRC) willow and poplar. However, they also require substantial areas of land for cultivation. Consequently, bioenergy crop production will place large demands on land resources in the next decades.

Rounsevell and Reay (2009) analysed past studies of land use change at global, European and national scales (i.e.UK), and found that most scenarios indicated increased use of agricultural land for bioenergy crop production in the future. To support this argument, a substantial land area is estimated to be required to cultivate crops to meet the ambitious biofuel targets set by the U.S.A., EU and fast growing countries such as Brazil, India and China, where energy and transport fuel demand is expanding rapidly (Bhattacharya et al. 2003; Sudha et al. 2003; IEA 2004; FAO 2008; Field, Campbell & Lobell 2008; Keyzer, Merbis & Voortman 2008; Sanderson & Adler 2008; Lapola, Priess & Bondeau 2009). To meet global biofuel targets, the International Energy Agency (IEA) (2011) provides the most recent estimates, that around 65 million hectares will be required by 2030 and 105 million hectares by 2050.

This is expected to result in land use changes on a global scale. A large number of studies (Gibbs et al. 2008; Nepstad et al. 2008; Searchinger et al. 2008; Al-Riffai, Dimaranan & Laborde 2010; Tyner et al. 2010) have suggested a link between recent bioenergy policies and targets, increased demand for crop expansion, and adverse land use changes (hereafter referred to as 'bioenergy-driven land use change'). In developing regions, this includes large-scale deforestation and expansion of oil palm plantations in Southeast Asia (Koh & Wilcove 2008; Danielsen et al. 2009; Koh et al. 2011), and soybean expansion in the Amazon basin and the Cerrado savanna in Brazil (Fearnside 2001b; Morton et al. 2006; Barona et al. 2010). These changes have attracted international attention. Ambitious bioenergy targets could result in the conversion of existing forests, grasslands, wetlands and arable lands into bioenergy agriculture on a global scale. The serious implications for bioenergy-driven land use change, suggested by past studies, include unintended GHG emissions from land clearing (so-called indirect land use change [iLUC]), decline in water quantity and quality, soil quality, biodiversity, social welfare and wellbeing at different geographical scales.

However, the implications remain uncertain, especially in an emerging debate with regard to iLUC effects. Land use change has been categorised as direct land use change (dLUC) and iLUC. dLUC involves changes to the land use on the site being used for bioenergy crop production, such as conversion of forests to oil palm plantations for biodiesel production. iLUC is the unintended effects that occur elsewhere as a consequence of the displacement of the land uses that give way to bioenergy crop production (Searchinger et al. 2008; IEA Bioenergy 2009; Berndes, Bird & Cowie 2010). The indirect land use displacement often occurs in one or several countries outside the original country, due to land resource constraints in the original country and also to the impacts of globalised markets (e.g. Al-Riffai et al., 2010; Bauen et al., 2010; Searchinger et al., 2008; Tyner et al., 2010). The U.S.A. and the EU, as part of their most recent bioenergy policies, have started tackling the issue of iLUC by introducing sustainability criteria and GHG accounting methods that incorporate emissions from iLUC (European Commission 2009; U.S. EPA 2010). There are currently extensive research efforts being undertaken to quantify the iLUC impacts worldwide. However, estimating iLUC is very complicated. The methodologies are still in their infancy, and results vary significantly between studies (Prins et al. 2010; Di Lucia, Ahlgren & Ericsson 2012). Therefore considerable work is required to tackle uncertainty in a wide range of factors (Di Lucia, Ahlgren & Ericsson 2012; Wicke et al. 2012).

1.1.1 Regional-scale environmental consequences of bioenergy-driven land use change

Bioenergy sustainability comprises broad issues of socio-economic and environmental sustainability related to feedstock production, processing, distribution, and use (Efroymson et al. 2013). Bioenergy-driven land use change effects are major consequences of feedstock production, which potentially involve both environmental and socio-economic impacts. Until recently, there has not been a comprehensive framework to assess the sustainability of bioenergy production systems using a range of environmental and socio-economic indicators and thus to move the discussion beyond the carbon/GHG debate. Research on environmental indicators and assessment frameworks for biofuel environmental sustainability is being developed in the U.S.A. (Zhang et al. 2010; McBride et al. 2011; Efroymson et al. 2013) and Europe (Langeveld et al., 2012; van Dam et al., 2009) in accordance with the recent developments in sustainability requirements and certification schemes for biofuels in international markets (e.g. the EU's Renewable Energy Directive [RED]).

A stronger emphasis is being placed on the environmental impacts of bioenergy production due to the importance of climate change in the international and national political arena. A large number of studies assessed the various environmental impacts of bioenergy crop production on different geographical areas and scales using a diversity of research methods. Until the late 2000s, the majority of this research focused on life cycle assessment (LCA) to compare the GHG and energy balance and emissions of different bioenergy feedstocks (e.g. fossil fuels, first generation and next-generation biofuels) (Quirin et al. 2004; von Blottnitz & Curran 2007; Worldwatch Institute 2007; Zah et al. 2007; Menichetti & Otto 2009; Williams et al. 2009). These LCA studies confirmed that the bioenergy feedstock/crop type and the associated management practices were key factors in determining the environmental outcomes of bioenergy production. Although commonly used, a key weakness of LCA was the assessment of spatial issues, including land use changes (Petersen 2008). While there have been recent improvements in its evaluation of land use changes, such as consequential life style analysis (CLCA) and its combination with the economic general equilibrium model which evaluates the impacts from iLUC (Kloverpris, Baltzer & Nielsen 2010), the LCA is still widely used in current studies to estimate GHG emissions and impacts on climate change.

In contrast, other environmental impacts of bioenergy crop production, such as those on water, soil and biodiversity, have received much less attention in the literature. These aspects are important when considering regional environmental impacts, but research in this area has emerged only in the past few years. Such studies include evaluation of the impacts/benefits of large-scale bioenergy crop production on water quality (Schilling et al. 2008; Ng et al. 2010; Wu & Liu 2012) and biodiversity (Bellamy et al. 2009; Wilcove & Koh 2010; Rowe et al. 2011), due to growing concerns over the expanding demand for biofuel feedstock. These studies showed that environmental impacts depend on various factors, such as the land use before the conversion, crop type/feedstock, geographical location, soil, climate and agricultural practice. For example, the conversion to lignocellulosic crops, such as perennial grasses and SRC generally results in lower environmental impacts than conventional crops. The conversion could potentially provide various benefits, if implemented and managed properly. These include reduced soil erosion/sediment (Wu & Liu 2012), enhanced water quality (Schilling et al. 2008; Dimitriou et al. 2009; Ng et al. 2010) and enhanced biodiversity in agricultural landscapes (Bellamy et al. 2009; Rowe, Street & Taylor 2009; Rowe et al. 2011; Baum, Bolte & Weih 2012; Langeveld et al. 2012).

1.1.2 Spatially explicit frameworks to evaluate the environmental sustainability of bioenergy crop production

Van Dam et al. (2009), van der Hilst et al. (2012a) and Langeveld et al. (2012) emphasised the importance of understanding the regional scale environmental impacts of bioenergy crop production. They identified this as a significant knowledge gap and this highlights the need for a comprehensive and integrated framework for evaluating regional-scale sustainability consequences associated with bioenergy-driven land use changes, and the presentation of the sustainable land use options at a regional scale for decision makers. A spatially explicit modelling approach based on Geographic Information System (GIS) is a promising methodology to achieve these objectives, since it has strong advantages in the analysis of spatial biophysical data, such as location, scale/area, topography, hydrology, and vegetation cover.

Since 2010 there have been important advances in identifying a spatially explicit approach in bioenergy sustainability research, especially in land use change modelling (Hillman & Verburg 2010, 2011; Koh & Ghazoul 2010; Mehaffey, Smith & van Remortel 2012; van der Hilst et al. 2012b), and to a lesser extent in developing environmental assessment frameworks which address and quantify multiple environmental effects in an integrative way (Zhang et al. 2010; van der Hilst et al 2012a). The assessment framework of van der Hilst et al. (2012a) was prepared for the European context to quantify the environmental effects of scenarios for bioenergy production from miscanthus and sugar beet using eight indicators – GHG emissions, water quantity and quality, soil quality, and biodiversity. The framework is highly relevant to policy and decision making concerning future bioenergy crop production in the European context. However, the methods used often required high levels of expertise, outcomes from extensive field trials and detailed environmental data. This may limit the application of this framework to regions outside Europe, including developing countries, which currently are experiencing high land use and environmental stress from bioenergy crop expansion, and where urgent solutions are required.

In the U.S.A. Zhang et al. (2010) explored trade-offs between bioenergy production and multiple environmental effects - biomass yield, GHG emissions, erosion, nitrogen (N) and phosphorous (P) losses. The strength of this framework is the flexible structure that suggested it could be linked to other aspects, such as economic analysis and evaluation of biodiversity. However, modelling of biodiversity effects has been a common challenge (Zhang et al. 2010), and biodiversity concerns generally have not been integrated in these frameworks (Stoms et al. 2012).

In the Australian context, the CSIRO has also modelled the economic viability and environmental impacts of eucalypt species for biomass production at a regional scale using spatially explicit modelling approach (Bryan, Ward & Hobbs 2008; Bryan, King & Wang

2010). The focus of their environmental assessment was, however, again restricted to carbon mitigation and region-specific issues such as salinity and wind erosion. The framework did not capture indicators highly relevant to regional environmental issues such as hydrology and biodiversity.

1.1.3 Underutilised agricultural land

Recent bioenergy policies and studies from the U.S.A. and Europe suggest the use of 'underutilised agricultural lands' for future bioenergy crop production to minimise the impacts of bioenergy-driven land use change such as: competition for land with food production; the release of GHG (mainly carbon from the soil and biomass); impacts on water, soil and biodiversity (Hoogwijk et al. 2005; Hill et al. 2006; Campbell et al. 2008; Field, Campbell & Lobell 2008; Wiegmann, Hennenberg & Fritsche 2008; European Commission 2009). More importantly, this is a commonly suggested solution to avoid the risk of iLUC (Bryngelsson & Lindgren 2013).

The definition of 'underutilised agricultural land' varies widely by country and is dependent on local conditions (Dale et al., 2010). However, in this research the term was defined as (Figure 1.1) lands that are not in production for more than a certain period of time [2] or are not suitable for food production for reasons, such as poor soil fertility, and unsuitable topographic and climate conditions. It includes lands referred to as 'abandoned agricultural land' and 'marginal' or 'low productivity' agricultural lands, and/or part of 'degraded land' (Hoogwijk et al. 2005; Hill et al. 2006; Campbell et al. 2008; FAO 2008; Fargione et al. 2008; Field, Campbell & Lobell 2008; Wiegmann, Hennenberg & Fritsche 2008; Wicke 2011). Several studies have already investigated the potential of marginal lands (Worldwatch Institute 2006; FAO 2008; Gopalakrishnan et al. 2009; Wicke et al. 2011) and abandoned agricultural land for biofuel crop production (Campbell et al. 2008; Field, Campbell & Lobell 2008; Reijnders & Huijbregts 2009) at different geographical scales. These studies estimated that from 100-1,000 million hectares of marginal, degraded and abandoned lands are potentially available for future biofuel production globally (Ackom, Mabee & Saddler 2010). However, the sustainability of these lands is controversial. Firstly, the availability and the potential of these lands may be much smaller than these estimates (Fritsche, Sims & Monti 2010). Secondly,

[2] Fallow land is land set aside for some time to improve soil fertility for certain periods (generally ranging from 1-5 years) before it is cultivated again. Underutilised agricultural land is not in production for more than a certain period of time due to factors that can affect productivity.

introduction of commercial-scale bioenergy crop production on these lands is unfeasible due to lack of incentives to grow crops on these lands (Bryngelsson & Lindgren 2013). Thirdly, the environmental outcomes from the use of these lands requires further research (Wicke et al. 2012), since degraded or marginal lands could require significant inputs of water and nutrients to maintain their productivity (Robertson et al. 2008; Fritsche, Sims & Monti 2010; Wicke 2011), and may have high conservation and biodiversity values, particularly if these lands have not been in production for a long period (Bowen et al. 2007; Robertson et al. 2008; Reijnders & Huijbregts 2009). The socio-economic outcomes are also questionable. In India, Africa and other developing regions, marginal land (often referred to as 'wasteland' in relevant studies) is an important part of the livelihood of small farmers and the rural poor for their day-to-day survival (Rajagopal 2007; FAO 2008; Matondi, Havnevik & Beyene 2011; van der Horst & Vermeylen 2011). For example, livestock production on these lands is the key to the rural economy in the agriculturally marginal areas in Africa (Matondi, Havnevik & Beyene 2011).

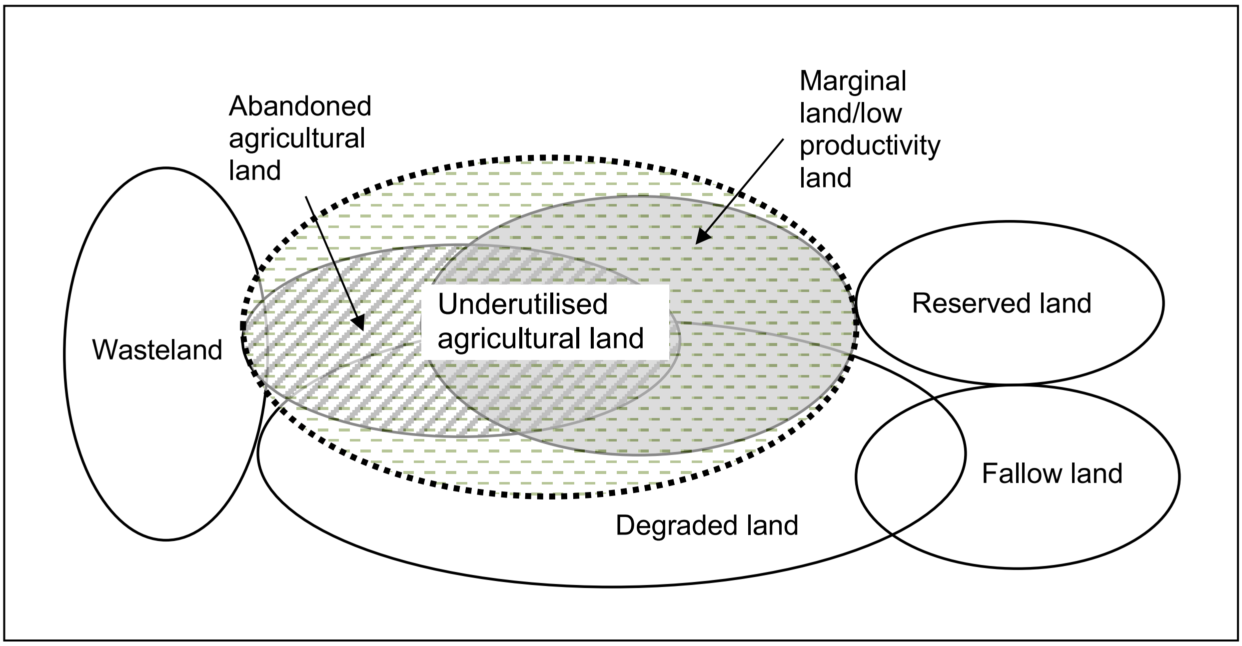


Figure 1.1 Key elements of 'underutilised agricultural land' (adapted and modified from Wiegmann et al., [2008]).

The sustainability of underutilised agricultural land is an important emerging area of research. However, its use as a means of mitigating the impacts of bioenergy crop production has not been well researched or evaluated. Thus this requires further evaluation (Wicke 2011), the results of which will assist decision-makers to better understand the potential impacts of the conversion of underutilised agricultural lands to bioenergy crop production. There is a need for more effective evaluation techniques using a comprehensive evaluation framework that addresses a range of appropriate environmental sustainability indicators and tools relevant to a regional scale.

1.2 Problem statement

Bioenergy-driven land use change and its sustainability is an emerging research area. This has been developing rapidly over the past few years due to the latest international bioenergy policies that specify sustainability requirements and certification for the bioenergy production process. A number of studies have projected bioenergy-driven land use changes in the next decades, and all have expressed concerns about the inevitable consequences. However, scientists are still exploring methodologies to evaluate the impacts and consequences of bioenergy-driven land use changes at different spatial scales, and decision makers are searching for the best policy options to minimise the impacts. This research focuses on the environmental sustainability issues surrounding bioenergy-driven land use changes and addresses the following knowledge gaps:

1. the need for a methodology to quantify and compare the regional-scale environmental sustainability of bioenergy crop production and its associated land use changes (bioenergy-driven land use change) in a more integrative way;
2. the need for a spatially explicit framework, which is less complex, but more flexible and appropriate for evaluating preferred land use options for future bioenergy crop production within the broader context of environmental sustainability; and
3. the limited research that has been undertaken in relation to the environmental impacts of using underutilised agricultural land relative to current and other land use options as a means of mitigating bioenergy-driven land use change effects.

Overall, the problems that this research addressed are: the need to better understand the environmental consequences of bioenergy-driven land use changes at different spatial scales; the limited understanding of the environmental sustainability of bioenergy crop production beyond energy balance and GHG emissions; the need for an evaluation framework for investigating the impacts of different land use change scenarios at regional scale; and the need for further research on the sustainability of the use of underutilised agricultural lands for bioenergy crop production. Research that addresses these deficiencies will contribute to our understanding of the environmental sustainability of bioenergy crop production and future land use options.

1.3 Research question

This research addressed the following research questions:

1. How, and to what extent will conversion of land from existing land uses to bioenergy agriculture affect environmental qualities at the regional spatial scale?
2. Compared to current land uses, what are the environmental impacts of using 'underutilised agricultural lands' to produce bioenergy crops, and how and under what conditions can the conversion of these lands for crop production deliver better environmental sustainability outcomes?

1.4 Research aim and objectives

In previous studies, it has been hypothesised that the use of underutilised agricultural lands for bioenergy production will be more environmentally sustainable than alternative land use scenarios as there will be less competition with lands used for food production and other uses, fewer carbon emissions from conversion from forest and grassland soils, and fewer impacts on natural resources and ecosystems (see 1.1.3). Thus the aim of this research is to evaluate whether land use change scenarios that involve bioenergy crop production on underutilised agricultural land can enhance regional-scale environmental outcomes when compared with current land uses, and to provide recommendations and suggestions for future land use options. The scenarios assessed in this research, including the selection of bioenergy crops (and the associated management practices) and land use change pathways, are developed so as to minimise the potential environmental impacts. Issues related to socio-economic sustainability fall outside the scope of this research despite this being an important area of research. The focus is on regional-scale environmental impacts, especially impacts on natural resources and ecosystems.

The objectives of this research are to:

- **Objective 1**: *Review the environmental and land use pressures resulting from the global increase in bioenergy production.*

Rationale: An important component of this research was to better understand the patterns and dynamics of bioenergy-driven land use change and its consequences. In Objective 1, this was achieved through a review, synthesis and analysis of the literature on the land use changes and impacts on the environment that have been occurring and are expected to occur as a result of an increase in bioenergy demand and production.

- **Objective 2:** *Develop a framework for evaluating the environmental consequences of bioenergy-driven land use change at the regional scale.*
  Rationale: Due to the limited research on a comprehensive framework for evaluating the environmental consequences of bioenergy-driven land use change and the lack of understanding of regional environmental consequences, an evaluation framework was developed for application at the regional scale through literature review, development of conceptual diagrams, and selection of key indicators and tools.

- **Objective 3:** *Test the effectiveness of the environmental evaluation framework by applying it to a selected region that has experienced land use changes due to crop production.*
  Rationale: The framework developed in Objective 2 was tested through application to a case study catchment, located in Queensland, Australia, where land use change to agricultural production has occurred previously. This objective included discussion of the strengths and limitations of the framework using key evaluation criteria for testing the appropriateness of the environmental indicators, the appropriateness of the selected models and tools, and the overall effectiveness of the framework from a user's perspective.

- **Objective 4:** *Develop land use change scenarios that incorporate the use of 'underutilised agricultural land' in a case study region, and apply the evaluation framework to evaluate the regional scale environmental consequences of the scenario in relation to current land use.*
  Rationale: In order to determine whether the use of underutilised agricultural lands for woody perennial bioenergy crop production will bring more sustainable outcomes, the framework was applied to different land use change scenarios developed in the case study catchment (Objective 3) to evaluate and compare regional scale environmental consequences.

1.5 Significance of this research

This research produced two main outcomes: (i) a spatially explicit framework for evaluating the consequences of bioenergy-driven land use changes that pulls together the results of each indicator; and (ii) evaluations and recommendations on more sustainable land use change options for future bioenergy crop production, including the evaluation of the environmental sustainability of the use of underutilised agricultural land.

Firstly, this evaluation framework took a spatially explicit approach to the environmental consequences of bioenergy-driven land use changes, in order to provide an overview of changes in the environmental qualities due to particular land use changes using maps and diagrams. Involving several key indicators for regional-scale environmental impacts, it provided a radar chart that synthesised the results of each indicator for different bioenergy-driven land use change scenarios. The recent global trends in environmental and land use policy and decision-making have indicated a greater reliance on decentralised arrangements at the local and regional scales. Thus this evaluation framework was designed to assist policy makers at the regional level to better understand the potential impacts of land use changes on natural resources and ecosystems. To achieve this purpose, it was also designed to be less complex, more flexible than existing frameworks, and potentially applicable to various regions.

More importantly, this evaluation framework was developed to evaluate the environmental sustainability of bioenergy crop production on underutilised agricultural land relative to current and other land use change options. The results of this evaluation framework will contribute to knowledge on bioenergy sustainability and inform bioenergy and land use policy makers in future direction setting. It will provide information on the preferable options for the location/land use and scale of future bioenergy crop production, and assist decision makers to develop sustainable land use and bioenergy policies and planning at a regional scale. In the last part of this research (Chapter 6), recommendations on more sustainable land use change options for future bioenergy crop production will be derived from the results of the land use change scenario analysis in an application to a case study region in Australia. The discussion will address the benefits and challenges associated with using underutilised agricultural land, and how and under which conditions it can provide environmental benefits and reduced degradation at the regional level.

1.6 Thesis structure

The thesis contains six core chapters, followed by a summary and conclusions (Figure 1.2).

Chapter 2 addresses Objective 1. The chapter includes a review and analysis of the patterns and dynamics of the bioenergy-driven land use changes that have been documented in past studies. The review was global in context, but focused on four countries/regions: Brazil; Indonesia and Malaysia; the U.S.A.; and the EU. This was based on the significant differences in the characteristics between these countries/regions. Land use change pathways were developed, and opportunities for sustainable land use options for bioenergy production were identified, based on the major findings from the literature review.

Chapter 3 addresses Objective 2, and explains the key processes associated with developing the spatially explicit environmental evaluation framework. These involved: development of conceptual diagrams to understand key environmental issues associated with bioenergy-driven land use changes, and cause and effect relationships with the interrelations of each factor involved in the issues; and selection of key indicators and tools that were incorporated into the evaluation framework. The key indicators were selected based on the results of the literature review, conceptual diagrams, and expert opinion. The tools incorporated into the evaluation framework were selected against a set of selection criteria.

Chapters 4 and 5 align with Objective 3 in this research. Chapter 4 provides justification of the case study selection for this research and background and key elements of the Burnett River catchment in Queensland, Australia, as a case study region. They included climate, land cover, vegetation, land use and its change, water quality and biodiversity in the catchment. This was followed by the application of the environmental evaluation framework to past land use changes in the catchment. Based on this result, Chapter 5 evaluates the effectiveness of the environmental evaluation framework against a set of key evaluation criteria to refine and validate the evaluation framework. The strengths and limitations of the framework are also discussed in this chapter.

Chapter 6 addresses Objective 4, which evaluated and compared the regional scale environmental consequences of bioenergy-driven land use change scenarios based on the use of underutilised agricultural land, through the application of the environmental evaluation framework. The land use change and management scenarios were developed based on the current land use of the Burnett River catchment and the existing land suitability analysis. Scenarios for the conversion of underutilised agricultural land to bioenergy crop production were compared against the current land use and also against alternative land use change scenarios to evaluate the relative sustainability of bioenergy crop production on these lands.

Finally, Chapter 7 summarises the main findings of the thesis. It also discusses the limitations of the research and makes recommendations for future research.

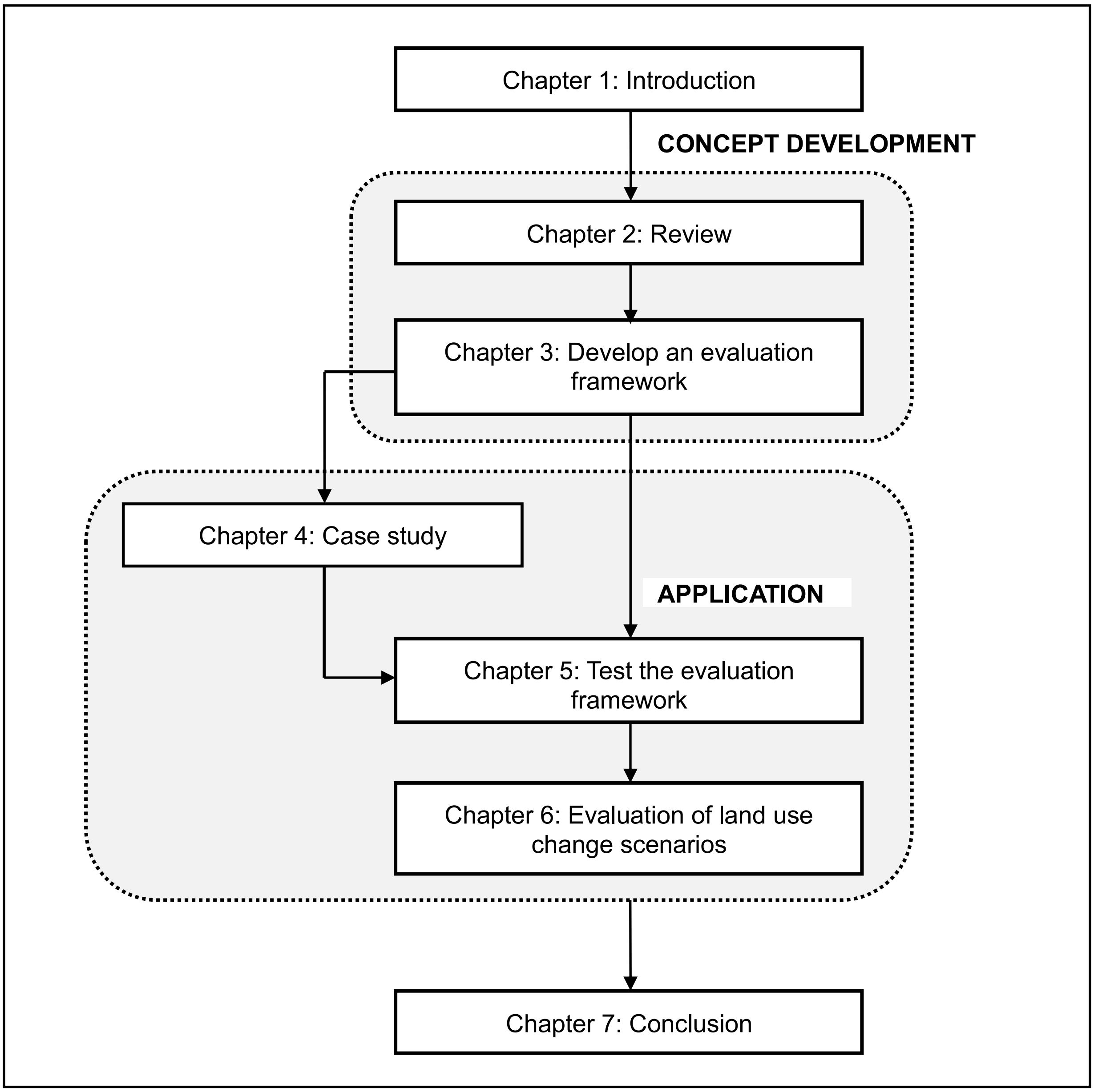


Figure 1.2 Thesis structure.

Chapter 2: Land use and environmental pressures resulting from current and future bioenergy crop expansion: A review [3]

2.1 Introduction

Recent energy and climate policies, particularly in the developed world, have increased demand for bioenergy (especially biofuels) as an alternative to fossil fuels, which has led to both direct and indirect land use changes (dLUC and iLUC) and an array of environmental and socio-economic concerns. The increased demand for biofuels has been linked to the 'global food crisis' and sparked the food versus fuel debate (Mitchell 2008; Matondi, Havnevik & Beyene 2011), which revolves around the current use of major food crops for biofuels feedstock (de Vries et al. 2010). Environmental concerns have also included deforestation and increased greenhouse gas (GHG) emissions, soil and water degradation, and biodiversity loss (Reijnders 2009). Further, social polarisation (between large land holders and smallholder/landless farmers), displacement of communities, and the disregard for local land rights have been reported in developing countries (Worldwatch Institute 2007). The production of dedicated bioenergy crops will most likely continue to place significant demands on land resources worldwide, even though high-yield plant species, such as non-food oil crops (e.g. Pongamia [*Milletta pinnata*], Jatropha [*Jatropha curcas*]) and lignocellulosic crops (e.g. switchgrass [*Panicum virgatum*], and Short Rotation Coppice [SRC]) plantations will be introduced in the medium- to long-term. Thus a comprehensive understanding of the land use dynamics of bioenergy crop production and how and to what extent the bioenergy-driven land uses affect environmental sustainability are essential for the development of sustainable bioenergy and land use policies.

The purpose of this chapter is to analyse the patterns and dynamics of bioenergy-driven land use change. The review focuses on four regions as the most prominent locations in which these patterns and changes occur: Brazil; Indonesia and Malaysia; the United States of America (U.S.A.); and the European Union (EU). Opportunities for land use options and policy instruments that can reduce the impact of future bioenergy crop expansion are also

[3] This chapter is based on a published paper: Miyake, S, Renouf, M, Peterson, A., McAlpine, C, & Smith, C, 2012, 'Land use and environmental pressures resulting from current and future bioenergy crop expansion: a review', *Journal of Rural Studies*, vol. 28, no. 4, pp.650-658. The paper is attached in Appendix.

identified at the end of this chapter.

2.2 Materials and methods

2.2.1 Definition of terms

Research which addressed both land cover changes and land use changes was reviewed. Land cover is defined by the attributes of the land surface and its composition, including soil, water, vegetation and topography. Land use represents the purposes for which humans use the land cover (Lambin, Geist & Rindfuss 2006) and it involves changes to the attributes of the land surface (Reijnders & Huijbregts 2009). Important mechanisms for land use change associated with increased production of agricultural crops are: cropland expansion; agricultural intensification; and displacement of other crops or activities (Kløverpris et al. 2008), particularly the conversion of an existing land cover type to another type, such as forest to bioenergy crops.

Bioenergy crop production can involve direct and indirect land use changes. Direct land use change (LUC) is defined as the change in land use on the site used for bioenergy crop production (Berndes, Bird & Cowie 2010). Indirect land use change (iLUC) is the unintended effects that occur elsewhere as a consequence of the displacement of existing crops or other land uses (Berndes, Bird & Cowie 2010; Fargione et al. 2008; Searchinger et al. 2008). Land use displacement often occurs in one or several countries outside the original country due to the impacts of international trade and globalised markets (IEA Bioenergy 2009; Mitchell 2008; Nepstad et al. 2008; Searchinger et al. 2008; Tyner et al. 2010).

Land use change from a particular crop is not necessarily driven by bioenergy demand where its by-products are also in high demand for multiple purposes (e.g. oil palm). However, all land use changes associated with major bioenergy crops that are also cultivated for food, feed and other purposes (see the next section for name of crops) are referred to as bioenergy-driven land use changes in this review.

2.2.2 Review methodology and scope

This review focused on two information sources: relevant research identified from the ISI Web of Science database; and reports, documents and materials (grey literature) prepared by government agencies and consultants. The latter information was included because research programs addressing the land use change effects of biofuels frequently are initiated or funded by government agencies as part of international and national climate change strategies. Both peer-reviewed papers and the grey literature were included in the review when they met the following criteria: they were published and written in English; and they documented changes in agricultural land use and/or land cover, either directly or indirectly as a result of existing and proposed future bioenergy crop production.

The database search was undertaken globally using a combination of keywords: ‘deforestation’, ‘land-use change’ or ‘agricultural intensification’; and ‘bioenergy’, ’biofuel’, or the name of major bioenergy crops that are cultivated for food, feed and other purposes, such as ‘corn’ or ‘maize’, ‘soybean’, ‘oil palm’, ‘sugarcane’, ‘wheat’ and ‘cassava’. This review also included land use changes to/from grassland (including pasture, native grassland, rangeland and savanna based on land use categories defined by the IPCC [2006]) and agricultural land uses that were not in current or future production (e.g. set-aside and agricultural marginal land), since bioenergy crop production has started occurring and/or has been proposed on such lands in many parts of the world.

In total, 57 studies addressing a wide range of topics from both developed and developing regions were identified for review, of which 30 reported past and current land use changes and of which 31 documented projected land use changes (Appendix 2 [Table A2.1]). Key data relating to land use change were recorded including the type of land use change (e.g. deforestation/cropland expansion and displacement of other crops/indirect land use change), location, time-scale, crop type, previous land use before conversion, causes, and environmental consequences. Then, the key data and information contained in each source were synthesised for further analysis for four geographic regions where bioenergy driven land use changes were reported with the highest number of times: Brazil; Indonesia and Malaysia; U.S.A.; and EU. Bioenergy crop type and land use prior to bioenergy crop conversion in the literature were used to describe the bioenergy-driven land use change pathways within the four regions covered in this review (Figure 2.1). Information on land use change drivers and time-scales were also recorded and used for this analysis.

2.3 Results

2.3.1 Brazil

Brazil has successfully established the world's second largest bioenergy production base (24.5 billion litres in 2008) and largest export market (5.1 billion litres in 2008) for ethanol fuel under its sugarcane ethanol fuel programme (Martinelli & Filoso 2008; UNICA 2008a, 2008b; RFA 2010). Sugarcane is one of the cheapest, most efficient and productive crops for ethanol production, with favourable energy balances and GHG emission potential (von Blottnitz & Curran 2007; Renouf, Wegener & Nielsen 2008). The area of harvested sugarcane in Brazil increased from 1.4 to 8.6 million ha between 1961 and 2009 (FAOSTAT 2011), with production concentrated in the south-central region (UNICA 2008b). The total planted area increased by 53 percent (%) between 2004 and 2009 due to the increased demand for fuel ethanol (FAOSTAT 2011). Under the Brazilian Biodiesel Program (PNPB), Brazil also has a legislated biodiesel target. Soybean oil is currently the major biodiesel feedstock (Pousa, Santos & Suarez 2007). The area of soybean production increased from 0.2 million ha in 1961 to 21.7 million ha in 2009 (FAOSTAT 2011). This has been driven by high global prices for soybean since 1990 (Fearnside 2001b), although only 7 % of total soybean production was used to produce biodiesel in 2010 (The Soybean and Corn Advisor 2010).

In Brazil, soybean production, cattle ranching, and more recently high, global demand for sugarcane ethanol, are the major drivers of the conversion of native forests and savannas to agriculture (Fearnside 2005; Morton et al. 2006; Brannstrom et al. 2008; Sawyer 2008; McAlpine et al. 2009; Barona et al. 2010). The expansion of soybean and cattle pasture has resulted in large-scale deforestation of the Amazon rainforest (700,000km^2 deforested) (Fearnside 2001b 2005; Morton et al. 2006; Sawyer 2008; Barona et al. 2010; Gibbs et al. 2010) and the Cerrado savanna (800,000-1,600,000km^2 deforested) (Fearnside 2001b; Mueller 2003; Brannstrom et al. 2008; Sawyer 2008; Fearnside et al. 2009) (Figure 2.1 [a]).

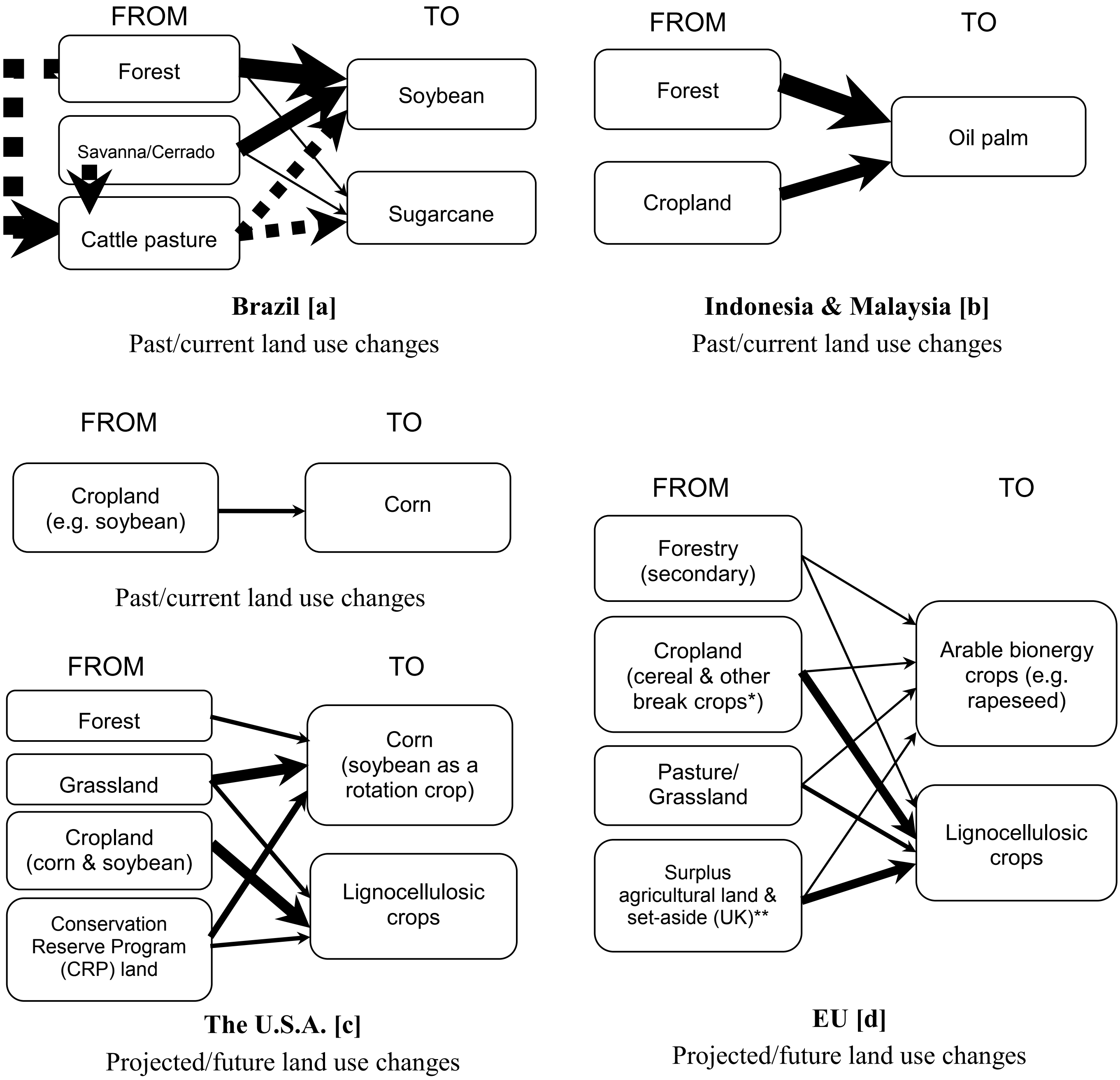


Figure 2.1 Bioenergy-driven land use change pathways in four geographic regions. Arrow width is proportional to the number of documentation of land use changes in the reviewed literature, which included both direct and indirect land use changes.

* Other break crops include linseed, lupins, drypeas and soybeans (Bauen et al. 2010).
**Set-aside (UK): The set-aside measure under the EU's Common Agricultural Policy (CAP) was abolished in 2008 after the global food price crisis, but the UK revived the concept as a voluntary approach in 2009 due to the strong benefits for soil and water quality, and wildlife conservation (Campaign for the Farmed Environment 2010)

The geographic expansion of sugarcane is less than that for soybean production, because the Amazon and most of the Cerrado are not suited to sugarcane production (Goldemberg, Coelho & Guardabassi 2008; Sparovek et al. 2009). However, many studies argued that recent expansion of sugarcane production areas and part of soybean production areas has occurred through the conversion of abandoned or degraded lands previously cleared for cattle pasture in the southern states,[4] and this has resulted in indirect deforestation, pushing displaced cattle ranching further into the frontier regions (Figure 2.1 [a]) (Goldemberg, Coelho & Guardabassi 2008; Nepstad et al. 2008; Sawyer 2008; Barona et al. 2010; Lapola et al. 2010; Loarie et al. 2011). High international market prices for bioenergy crops have resulted in a rapid rise in agricultural land prices in the southern states (de Nie, Sayer & McCormick 2009), which will continue to push cattle ranchers into frontier regions as they seek larger and cheaper tracts of land for grazing (Nepstad et al. 2008; Sawyer 2008; McAlpine et al. 2009; Barona et al. 2010). A simulation based on Brazil's biofuels targets for 2020 estimates that sugarcane ethanol and soybean biodiesel will be responsible for 41 % and 59 % of indirect deforestation in Brazil respectively, the main mechanism for deforestation being displacement of cattle ranching by sugarcane in the south-eastern states and by soybean in the south-western states (Lapola et al. 2010).

Brazil has a favourable climate for agricultural crop production, abundant land and water resources, low labour costs, and favourable government policies (Naylor et al. 2007; Martinelli & Filoso 2008; Sawyer 2008). Hence, the scarcity of suitable land for crop expansion in other countries has increased pressure to expand bioenergy crop production in Brazil (Nepstad et al. 2008). Brazil will continue to meet a large proportion of the future global demand for bioenergy, especially demand coming from the U.S.A. (Searchinger et al. 2008; Tyner et al. 2010) and from the EU (Banse et al. 2008; Al-Riffai, Dimaranan & Laborde 2010; Blanco Fonseca et al. 2010; Hiederer et al. 2010; Laborde 2011).

The Brazilian government has identified agricultural expansion and large-scale deforestation as a major challenge, and deforestation rates within Brazil as a result have slowed in recent years (INPE 2012). However, the amended Forest Law [Law no. 4771/65] is regarded as being ineffective in restricting further clearing (Sparovek et al. 2010) due to inadequate legal enforcement and conflicting policy responses between environmental and other agencies that

[4] The south-eastern states of Brazil refer to those in São Paulo, Minas Gerais, and Paraná, while the south-western states refer to those in Mato Grosso, Mato Grosso do Sul, and Goiás.

pursue economic development, agricultural interests, and land reform (McAlpine et al. 2009; Pacheco 2009). Ambitious goals of the Brazilian government and the bioenergy industry to expand the production and export of biofuels still remain. A substantial land area has been identified for future expansion of sugarcane (64.7 million ha) and oil palm (more than 30 million ha) under the Brazilian government's Agroecological Zoning program (Martin 2011).[5]

2.3.2 Indonesia and Malaysia

Palm oil is a valued ingredient in a number of food and cosmetic products. This has resulted in the rapid expansion of palm oil plantations in tropical developing regions, especially Southeast Asia. Palm oil has been the cheapest source of vegetable oil on the global market, and its higher yields and more favourable GHG and energy balance, compared to temperate oilseed crops, make it economically attractive as a biodiesel feedstock (Thoenes 2006; Naylor et al. 2007; Worldwatch Institute 2007). Indonesia and Malaysia are the largest producers and exporters of palm oil, accounting for 84 % of the world palm oil production in 2008 (16.9 and 17.7 million tonnes, respectively; FAOSTAT 2011). Both countries provide ideal agro-climatic conditions for oil palm production, low establishment and running costs for plantations, and hence high profitability (Nantha & Tisdell 2009). The harvested area for oil palm has increased steadily in Malaysia since the 1970s (Abdullah & Hezri 2008), while an exponential increase has occurred in Indonesia since the 1990s (FAOSTAT 2011). Between 2000 and 2007, the harvested area of palm oil in the region increased by 63 % (125 % alone in Indonesia) from 5.1 to 8.3 million ha, which correlates with a doubling of Europe's palm oil imports (FAOSTAT 2011). In addition to their internal biodiesel programs, both countries have announced the allocation of six million tonnes of palm oil for export, to meet the global demand for biodiesel, mainly from EU countries, the U.S.A. and other Asian countries (Biopact 2006; Thoenes 2006; Hoh 2010).

Consistent with an earlier study by Koh and Wilcove (2008), the review identified that more than half the recent oil palm plantation expansion in Indonesia and Malaysia occurred at the expense of forests, with the remainder displacing existing cropland (e.g. rubber plantations in Malaysia, which were originally converted from natural rainforests before the 1970s)

[5] The Sugarcane Agroecological Zoning prohibits sugarcane cultivation in the Amazon, in native ecosystems, and in areas with high conservation values, while oil palm zoning focuses on the recovery of degraded land within the Amazon basin and aims to provide socio-economic benefit to smallholder farmers (Martin 2011).

(Abdullah & Nakagoshi 2007) (Figure 2.1[b]). Thus palm oil plantations have been a major driver of deforestation in the region, with resulting increases in carbon emissions, habitat loss, and biodiversity decline including the endangered orang-utan (*Pongo abelii* and *P. pygmaeus*) (Koh & Wilcove 2008; Danielsen et al. 2009; Nantha & Tisdell 2009; Koh et al. 2011), and land disputes including the lack of clarity around land ownership and the displacement of indigenous people (Naylor et al. 2007).

The high economic return for oil palm has attracted large private sector investment, but also regional government support through economic development policies (Thoenes 2006; Abdullah & Hezri 2008; Nantha & Tisdell 2009), which are frequently prioritised over environmental policies (Abdullah and Hezri 2008). This was demonstrated by the granting of large-scale development permits for the conversion of primary rainforests to oil palm plantations in Indonesia until recently (Nantha & Tisdell, 2009). Future projections of land use change in the region indicate that oil palm plantation was unlikely to expand further into existing cropland due to its decreasing availability, and instead would mainly occur through the conversion of native forests (Bauen et al. 2010). However, international pressures may limit this trend. For example, the Indonesian government entered a partnership with the Reducing Emissions from Deforestation and Forest Degradation (REDD) program in May 2010, with an immediate two-year moratorium to stop issuing new permits for clearing primary forest and peatland (REDD 2010; REDD in Indonesia 2010).

2.3.3 The United States of America (U.S.A)

Top-down biofuel policies and mandates under the National Renewable Fuel Standard (RFS) created under the Energy Policy Act of 2005 (U.S. EPA 2009) were viewed as the primary cause of recent bioenergy-driven land use changes in the U.S.A. (Tyner et al. 2010). From the substantial increase in corn-based ethanol production since the mid-2000s, subsidies and programs under the RFS led to the U.S.A. overtaking Brazil as the largest fuel ethanol producer in the world (34 billion litres in 2008) (RFA 2010). The area of corn harvested increased from 28.6 to 35 million ha between 2006 and 2007 alone (FAOSTAT 2011).

The rapid corn-based ethanol expansion has led to a global debate about food security (Mitchell, 2008), the relative energy and GHG benefits of corn-based ethanol (Pimentel & Patzek 2005; Hammerschlag 2006; Miller, Landis & Theis 2007; von Blottnitz & Curran

2007), and its iLUC impacts (Searchinger et al. 2008; Tyner et al. 2010). The state of California was the first to respond by enacting a Low-Carbon Fuel Standard in 2007 (LCFS) (State of Calfornia Office of the Governor 2007). Its eligibility criteria include life-cycle GHG emissions from iLUC outside the country (Calfornia Air Resource Board 2012). At the national level, the RFS is being revised (RFS2), under the Energy Independence and Security Act (EISA) of 2007, to expand the total biofuels mandate to 136 billion litres (36 billion gallons) by 2022, incorporating advanced biofuels,[6] cellulosic biofuel, and biodiesel. It also requires new GHG accounting methods taking into account the iLUC emissions (U.S. EPA 2010).

The land use change pathways for bioenergy crops in the U.S.A. have mostly involved increased corn production in the Corn Belt region (USDA 2010), replacing existing soybean cropland (Mitchell 2008; Schilling et al. 2008) (Figure 2.1 [c]). Around 22 million ha of cropland will be available for bioenergy crop production by 2050, although this may be insufficient to meet demand under the current national targets (Perlack et al. 2005). The total area under corn production is predicted to reach around 38 million ha by 2008-2016 (Tokgoz et al. 2007), replacing soybean cropland and lands under the Conservation Reserve Program (CRP)[7] (Secchi et al. 2011). Future production under the RFS2 targets will also be met through cellulosic biofuel from lignocellulosic crops (i.e. switchgrass and/or miscanthus) (Schilling et al. 2008; Ng et al. 2010; Ugarte et al. 2010; Vanloocke, Bernacchi & Twine 2010; Le, Kumar & Drewry 2011), possibly grown on CRP lands (Figure 1 [c]) (Schilling et al. 2008; Payne 2010; Love & Nejadhashemi 2011; Wu & Liu 2012). The use of CRP lands for large-scale lignocellulosic crop production has been proposed in the medium- to long-term (Graham, Downing & Walsh 1996; Walsh et al. 2003; Naylor et al. 2007; Payne 2010; Hartman et al. 2011). However, this proposal has been controversial in terms of carbon emissions (Pineiro et al. 2009), natural resource management, and wildlife conservation (Roberts, Male & Toombs 2007; Payne 2010).

More importantly, the U.S.A.'s biofuel program has, and will continue to cause land use change outside the country. iLUC studies suggested that expansion of the U.S.A's corn

[6] 'Advanced fuel' refers to renewable fuel, other than ethanol derived from corn starch that has at least 50 percent less than baseline lifecycle GHG emissions (The Energy Independence and Security Act of 2007).
[7] Conservation Reserve Program (CRP) is a voluntary set-aside program established by the U.S. Department of Agriculture to remove highly erodible and environmentally sensitive land from agricultural production.

ethanol production could trigger large-scale conversion of native forest and grasslands to bioenergy crops worldwide, especially in Brazil, as a result of the displacement of existing crops, such as soybeans (Nepstad et al. 2008; Searchinger et al. 2008; Tyner et al. 2010).

2.3.4 European Union

The EU first introduced biofuel targets in 2003 (COM 2003/30/EC) (European Commission 2003), under which various policy instruments have been used to increase bioenergy use, including fuel tax exemptions, mandates, import tariffs, and financial support for industry development (Blanco Fonseca et al. 2010). As a consequence, fuel ethanol produced from grains and sugar beet increased sevenfold between 2004 and 2009 to 3.9 million litres per annum (ePure 2010), and biodiesel production from rapeseed increased sixfold between 2003 and 2009 to 7.9 billion litres per annum (EBB 2010). The EU is a world leader in biodiesel production, with 65 % of the global biodiesel output in 2009 (Biodiesel Magazine 2010). There has been strong public support for the more costly, domestically produced feedstocks (Thoenes 2006). As a result, the harvested area of rapeseed in the EU 27[8] increased by 53 % from 4.2 to 6.5 million ha between 2002 and 2009 (FAOSTAT 2011), and diversion of domestically produced rapeseed oil from food to biodiesel occurred in line with the increased palm oil import from Southeast Asia (Krautgartner et al. 2011; Thoenes 2006). Biodiesel use accounted for nearly two-thirds of the total EU 27 produced rapeseed oil in 2011 (Krautgartner et al. 2011).

In 2009, the EU replaced existing bioenergy targets with the Renewable Energy Directive (RED) (2009/28), which sets targets of 20 % renewable energy overall and 10 % renewable transport energy by 2020 (European Commission 2009). The RED introduces environmental sustainability criteria for production processes and a minimum rate of direct GHG emission savings for biofuels consumed in the EU, including GHG emissions from both dLUC and iLUC within and outside of the EU (European Commission 2009). This has resulted in extensive research efforts for quantifying GHG emissions from bioenergy-driven land use changes, using life cycle assessment (LCA) and the iLUC factor approach (Fritsche, Hennenberg & Hünecke 2010; Fritsche et al. 2010).

[8] The EU 15 refers to EU member states before the enlargement in 2004: Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxembourg, the Netherlands, Portugal, Spain, Sweden and the United Kingdom. The EU 27 includes EU15, and Bulgaria, the Czech Republic, Estonia, Cyprus, Latvia, Lithuania, Hungary, Malta, Poland, Romania, Slovenia, and Slovakia.

To date, bioenergy-driven land use change within the EU has been limited. However, the reviewed literature indicated future bioenergy demand will influence land use in the EU (Banse et al. 2011) (Figure 1 [d]). In EU 15,[6] the demand for cropland and pasture for food production was expected to decrease, and then 'surplus' agricultural land would become available for future bioenergy crop production (Rounsevell et al. 2006; Rounsevell & Reay 2009; Fischer et al. 2010). A major focus is on bioenergy production from lignocellulosic crops in the mid- to longer term (Powlson, Riche & Shield 2005; Bellamy et al. 2009; Rowe, Street & Taylor 2009), with an increased future allocation of set-aside areas[9] for large-scale production of lignocellulosic crops in the UK (Powlson, Riche & Shield 2005; Rowe, Street & Taylor 2009; Rowe et al. 2011) and in Europe (Rowe, Street & Taylor 2009; Fiorese & Guariso 2010).

Since land is limited in the EU, growing biofuel demand has started to impact on land resources outside the region. Besides the increasing vegetable oil imports, European companies have claimed over 5 million ha of land in the 'South', namely South America, Southeast Asia, and Africa for biofuel production (Borras, McMichael & Scoones 2010; Matondi, Havnevik & Beyene 2011). Countries such as Spain have started importing soybean-based biodiesel from Argentina (Biodiesel Magazine 2010), and this has raised environmental and social sustainability concerns (Grau, Gasparri & Aide 2008; Panichelli, Dauriat & Gnansounou 2009; Tomei et al. 2010). To meet the biofuel target, EU countries will depend more on imported feedstock and processed biofuels, especially fuel ethanol, from countries where agricultural expansion is possible—Brazil, Argentina, Ukraine and other Commonwealth of Independent States (CIS) countries, the U.S.A. and Canada (Banse, van Meijl & Woltjer 2008; Al-Riffai, Dimaranan & Laborde 2010; Blanco Fonseca et al. 2010; Hiederer et al. 2010; Laborde 2011). This may involve further conversion of primary forests, savannas, and grasslands to bioenergy crops in these countries. For example, in Brazil, 58 % of cropland extension is projected to occur on savanna grassland and 15 % on primary forest by 2020, due to the implementation of the EU biofuels mandate (Al-Riffai et al. 2010).

[9] Until 2008, set-aside was required under the EU's Common Agricultural Policy (CAP) to regulate food production, develop non-food crops, mitigate environmental impacts and support small farmers. The EU abolished the scheme in 2008, but the UK revived the concept as a voluntary approach in 2009 in expectation of the strong environmental benefits.

2.3.5 Synthesis of land use change pathways

The bioenergy-driven land use changes of the four regions are synthesised into four pathways (Figure 2.2).

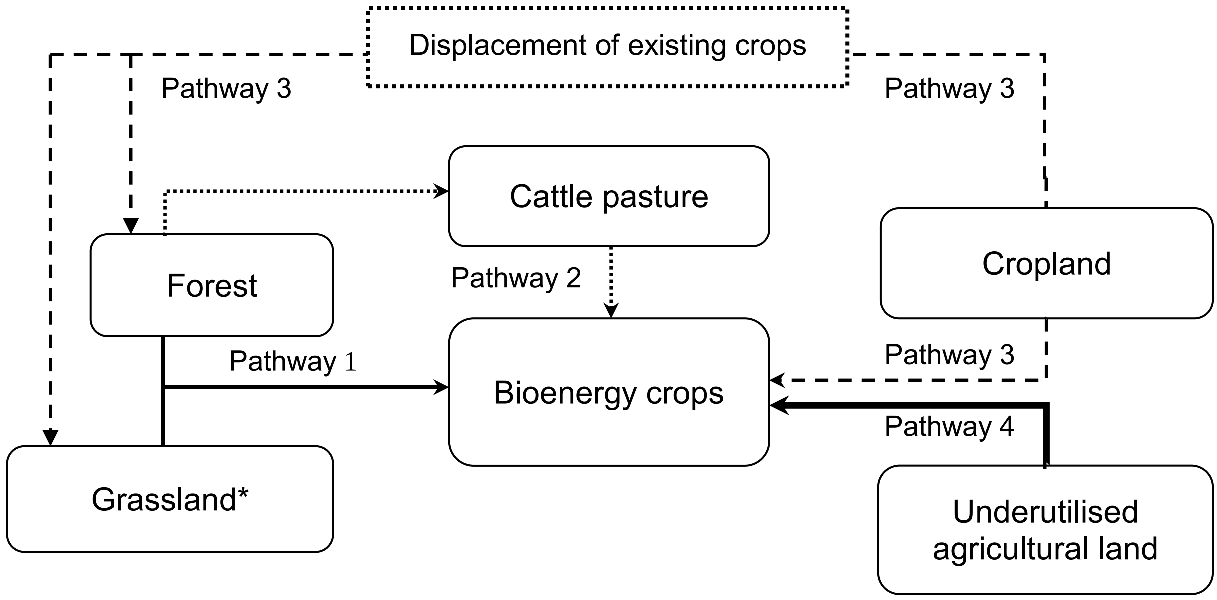


Figure 2.2 General pathways for bioenergy-driven land use change across all regions.
* Grassland includes natural grassland, rangeland and savanna.

Pathway 1 involves direct clearing of native forests, savannas, and grasslands to make way for bioenergy crop expansion. This pathway was the most common land use change pathway for developing regions, such as South America, Southeast Asia, and Africa. Pathway 2 involves conversion of cattle pasture resulting indirectly from bioenergy crop expansion, as described for Brazil. The displacement of cattle ranching may lead to indirect deforestation in other locations (IEA Bioenergy 2009). Pathway 3 represents the conversion of existing cropland to bioenergy crop production, and was primarily documented in the U.S.A. and the EU. This pathway could also trigger iLUCs in other locations through the displacement of existing crops. There is also a risk of indirect deforestation due to the displacement of existing agricultural lands in developing regions with abundant land resources, ideal agro-climatic conditions, and strong development pressures, such as Brazil.

Pathway 4 involves the conversion of marginal, degraded, or abandoned agricultural land to bioenergy crop production, especially for non-food and lignocellulosic crop production.

These agricultural lands are not in production for more than a certain period of time[2], or not suitable for food production for reasons, such as poor soil fertility, and unsuitable topographic and climate conditions (referred to here as ‘underutilised agricultural lands’), which include lands under CRP in the U.S.A. and set-aside areas in Europe. The availability and potential use of these agricultural lands have been increasingly recognised as having potential for future bioenergy production in order to minimise various land use change impacts (Campbell et al. 2008; Fargione et al. 2008; Field, Campbell & Lobell 2008; Wiegmann, Hennenberg & Fritsche 2008). However, the sustainability of the use of these lands for bioenergy crop production is uncertain and requires further research (Wicke 2011).

2.3.6 Environmental and socio-economic consequences of bioenergy-driven land use changes

The majority of the reviewed literature addressed the environmental consequences of bioenergy-driven land use changes (Table 2.1), although the level of impact differed among the various cases. These consequences depended on various factors, such as the land use before the conversion, crop type/feedstock, geographical location, soil, climate and agricultural management practices. Of these factors, the reviewed studies agreed that the land use change pathways (i.e. combination of the previous land use and crop type/feedstock) were the key factors determining the environmental outcomes of bioenergy production. For example, Pathway 1 was widely considered to involve more significant environmental consequences than Pathway 4 (Fargione et al. 2008; van Dam et al. 2009; Ackom, Mabee & Saddler 2010). This was because the land use before conversion substantially affected the amount of carbon dioxide (CO_2) and methane (CH_4) released from the soils and biomass (Fargione et al. 2008; Gibbs et al. 2008; Searchinger et al. 2008; Kim, Kim & Dale 2009; van Dam et al. 2009) and the biodiversity and ecosystems (Koh & Wilcove 2008). The clearing of native vegetation can involve negative consequences for multiple natural systems including air, water, soil and biodiversity, and can impact on their provision of ecosystem services. These consequences stem from the multiple functions of primary forests, which differ significantly from cleared land, including cropland, pasture and even forest plantations (Reijnders & Huijbregts 2009).

Similarly, the bioenergy crop type/feedstock associated with both land and cropping management practices (e.g. ground cover, tillage, harvesting and stubble left intact, fertiliser application) was another key factor in determining the environmental outcome of bioenergy crop production, as previously identified by a large number of life cycle assessments (LCAs)[10] (von Blottnitz & Curran 2007). Crop type can determine the energy yield, the land areas required for cropping, energy balance and GHG emissions, amount of water and agro-chemical inputs required, impacts on soil, and the amount of carbon sequestered by the crop (EEA 2007; de Vries et al. 2010). In general, the conversion to perennial grasses and short rotation tree crops was understood to result in lower environmental impacts than conventional row crops, such as reduced soil erosion/sediment (Dimitriou et al. 2009; Love & Nejadhashemi 2011; Wu & Liu 2012), enhanced or rehabilitated soil and water quality (Bryan, Ward & Hobbs 2008; Schilling et al. 2008; Rowe, Street & Taylor 2009; Ng et al. 2010; Love & Nejadhashemi 2011) and biodiversity in the agricultural landscape (Bellamy et al. 2009; Rowe, Street & Taylor 2009; Rowe et al. 2011). Sustainable cropping practices, such as reduced and no-till methods, choice of crop rotation, and fertiliser application can significantly reduce the potential environmental impacts, such as the amount of carbon released from the soil (Kim, Kim & Dale 2009) and the negative impacts on soil and water quality from bioenergy-driven land use changes (EEA 2007; Ugarte et al. 2010; Langpap & Wu 2011; Secchi et al. 2011).

On the contrary, a limited number of the reviewed studies documented the socio-economic implications of bioenergy-driven land use changes. However, this review indicated that the land use changes resulting from increased bioenergy demand could negatively impact on local communities in developing regions, particularly minority and indigenous people, in situations where formalised land tenure and land rights were often not in place (de Nie, Sayer & McCormick 2009; van der Horst & Vermeylen 2011). Land disputes between large landholders and small farmers due to conflicting and ambiguous land tenure law were identified in oil palm plantations in Indonesia (Naylor et al. 2007) and also in the Brazilian Amazon (Fearnside 2001a).

[10] Life cycle assessment (LCA) was employed in a large number of studies as a methodology to compare the environmental outcomes of bioenergy products with fossil fuels (e.g. von Blottniz and Curran, 2007). The approach is based on the assumption that environmental impacts vary according to the particular feedstock and the production process. However, many of the past LCAs have not addressed land use change issues (de Vries et al, 2010; Foley et al., 2005; Petersen et al., 2008).

Table 2.1 Synthesis of the environmental consequences of bioenergy-driven land use changes.

	Environmental consequences	References
Climate (global)	release of soil and biomass carbon to the atmosphere (e.g. deforestation, peat soils)	Bauen et al. (2010) ; Fearnside (2005); Fearnside et al. (2009); Liska & Perrin (2009); Martinelli & Filoso (2008) ; Morton et al. (2006) ; Naylor et al. (2007) ; Nepstad et al. (2008) ; Osher, Matson & Amundson (2003) ; Panichelli et al. (2009) ; Roberts et al. (2007) ; Sawyer (2008) ; Searchinger et al. (2008); Tomei et al. (2010); Tyner et al. (2009) ; Van Dam et al. (2009)
	GHG (e.g. N_2O, CO_2, CH_4) emissions from fuel and fertiliser consumption	
Climate (regional)	potential change in regional climate (temperature, precipitation and wind patterns) due to changes in vegetation cover, surface runoff and albedo	Fearnside (2005) ; Loarie et al. (2011) ; Nepstad et al. (2008) ; Sampaio et al. (2007); Twine et al. (2004);
Air (quality)	air pollution (e.g. smoke, ash and toxic pollutants such as particles) from burning forests for clearing and cane burning	Goldemberg et al. (2008); Martinelli & Filoso (2008); Sawyer (2008)
	aerial and terrestrial spraying of agrochemicals	Sawyer (2008)
Water (quality)	degraded water quality through increase of nitrogen, phosphorous and sediment loads	Grau et al. (2008); Martinelli & Filoso (2008); Mueller (2003) ; Sawyer (2008); Schilling et al. (2008) ; Tomei et al. (2010)
	improved water quality through decrease in nitrate and/or sediment loads (lignocellulosic crops compared to the conventional crops)	Ng et al. (2010); Schilling et al. (2008); Wu and Liu (2012)
	destruction of instream and coastal aquatic biodiversity	Martinelli & Filoso (2008)
Water (quantity)	changes in hydrological regimes/balance (e.g. surface runoff, water yield, chance of floods)	Goldemberg et al. (2008); Le et al. (2011); Mueller (2003) ; Roberts et al. (2007); Schilling et al. (2008); Twine et al. (2004); Ugarte et al. (2010); Vanloocke et al. (2010)
	increased water consumption for irrigation	Roberts et al. (2007); Sawyer (2008)
Soil	soil erosion and landslides	Abdullah & Nakagoshi (2007); Abdullah & Hezri (2008); Martinelli & Filoso (2008) ; Mueller (2003) ; Sawyer (2008)
	soil quality degradation (e.g. contamination, compaction, and loss of soil carbon and nutrient stocks)	Fearnside (2005); Grau et al. (2008); Martinelli & Filoso (2008); Mueller (2003) ; Roberts et al. (2007) ; Sawyer (2008); Tomei et al. (2010)
Biodiversity	loss of native biodiversity through habitat loss, fragmentation and degradation	Abdullah & Hezri (2008); Fearnside (2005); Fearnside et al. (2009); Koh & Wilcove (2008); Martinelli & Filoso (2008); Morton et al. (2006); Mueller (2003) ; Nantha & Tisdell (2009); Naylor et al. (2007); Nepstad et al. (2008); Roberts et al. (2007); Wilcove & Koh (2010)
	introduction of exotic plant and animal species	
	benefits for native biodiversity (lignocellulosic crops compared to the conventional crops)	Bellamy et al. (2009); Rowe et al. (2009); Rowe et al. (2011)
Other impacts	visual impact (e.g. change in ascetic landscape)	Fischer et al. (2010); Rowe et al. (2009)

The growth in biofuel production in recent years was discussed as the greatest threat to global food security, and the cause of the sharp increase in the price of internationally traded food crops (the so-called 'global food crisis') in 2007-08 (Mitchell 2008; van der Horst & Vermeylen 2011). However, recent economic analyses have disagreed with this view and argued that the global food crisis was actually a consequence of a number of causes (e.g. global supply and demand trends, oil price and speculation) and not the increased production of biofuels alone (Ajanovic 2011; Tyner 2013). In any case, the increase in the food commodity price not only impacted negatively on people's lives in terms of food accessibility, but also resulted in higher land prices and possible displacements. In the southern states of Brazil, cattle ranching took advantage of the increased land prices, selling the land to soybean or sugarcane farmers and sought cheaper land in the frontier regions (Fearnside 2001b; Nepstad et al. 2008; Sawyer 2008; de Nie, Sayer & McCormick 2009; McAlpine et al. 2009). Such displacement not only caused indirect deforestation, but its social implications were potentially significant (Van der Horst & Vermeylen 2011). The increased land prices were also thought to accelerate a shift in land tenure from smallholder farming to large-scale commercial plantations (de Nie et al. 2009), as small farmers were thought to be disadvantaged due to financial constraints. For example, the production of major conventional bioenergy crops, such as sugarcane, soybean and oil palm have been undertaken by large landholders, rather than small farmers, due to the requirements for large-scale mechanisation and the resultant economies of scale (Sawyer 2008). In the sugarcane and soybean industries in Brazil, a concentration of profits on large landholdings and the exploitation of the landless poor under low wage, unsafe and unhealthy conditions (e.g. seasonal cane cutting workers) were reported (Martinelli & Filoso 2008; Sawyer 2008), and projected bioenergy-driven land use changes were reported as being likely to contribute to further social polarisation. However, further research is still required in this area.

2.4. Discussion

The results from this review should be considered in the wider context of global and regional land use and bioenergy policies. The following opportunities were identified for more effective policy development in relation to bioenergy crop production, and its expansion.

Opportunity 1: Give high priority to no and/or less land-using bioenergy feedstock.

Bioenergy produced from waste and residues has substantial environmental advantages over bioenergy produced from dedicated energy crops (Ackom et al. 2010). It can largely avoid the sustainability issues associated with bioenergy-driven land use changes such as: competition for land with food production; the release of carbon from the soil and biomass; impacts on water, soil, and biodiversity (Hill et al. 2006; Campbell et al. 2008; Field, Campbell & Lobell 2008; Wiegmann, Hennenberg & Fritsche 2008); and more importantly, the risk of iLUC (Fargione et al. 2008). The consensus view in the literature was that bioenergy policy should continuously give priority to the use of such feedstocks that require no and/or less land resources and their associated technologies, such as lignicellulosic conversion technologies and biofuels from high-yield algae (Fritsche, Sims & Monti 2010; Yeh & Witcover 2010). Costs are still a challenge for commercialisation of lignocellulosic conversion technologies (Sims et al. 2008). Algae is a promising feedstock due to its much higher yield and lower land demands compared to terrestrial crops. However, a significant technical breakthrough is still required to achieve its commercialisation (Singh & Olsen 2011). Despite these limitations, bioenergy production from no and/or less land-using feedstocks must continue to be given priority over dedicated bioenergy crops in the long term. This can be achieved via various policy instruments such as financial support for research and development of related industries and supply chains.

Opportunity 2: Develop sustainable land use options for bioenergy crop production.

A significant portion of future bioenergy demand can be met in the long term by the previously mentioned no and/or less land-using feedstocks and other renewable energy technologies. However, land is still required for dedicated bioenergy crops to meet the short- to mid-term demand for bioenergy. This review emphasised that careful consideration must be given to the nature of the land use change pathways to ensure that their effects are minimised. Sustainable land use options for bioenergy crop production may involve two solutions: agricultural land use intensification; and the use of underutilised agricultural land.

Intensification of production on existing agricultural lands is certainly a solution that will help minimise further agriculture expansion of bioenergy crops. It can be achieved through the introduction of high-yield, land-efficient crops, improvements to the productivity of existing crops through the application of appropriate agricultural management practices, maximum use of by-products and co-products, and the introduction of multiple crop rotations (Wicke et al. 2012). For example, increasing grazing density has been proposed in Brazil to minimise deforestation associated with agricultural expansion and the associated carbon emissions (Lapola et al. 2010).

A second solution is the use of underutilised agricultural lands for non-food and lignocellulosic bioenergy crops (Pathway 4 in Figure 2.2). However, the sustainability of the use of these lands is controversial and uncertain for several reasons. Firstly, the availability and the potential of these lands may be much smaller than initially estimated (Fritsche, Sims & Monti 2010). Secondly, the environmental impacts from the use of these lands still requires further research, as they often require significant inputs of water and nutrients to maintain productivity (Robertson et al. 2008; Fritsche, Sims & Monti 2010; Wicke 2011), and may have high conservation and biodiversity values, particularly if abandoned for long periods (Bowen et al. 2007; Robertson et al. 2008; Reijnders & Huijbregts 2009). The socio-economic and social outcomes are also questionable. In India, Africa, and other developing regions, marginal land is an important part of the livelihood of smallholder farmers and the rural poor (Rajagopal 2007; Matondi, Havnevik & Beyene 2011; van der Horst & Vermeylen 2011). For example, livestock production on these lands is important for the rural economy in Africa (Matondi et al. 2011). Thus the socio-economic consequences of their use must be carefully evaluated. The sustainability of underutilised agricultural land for bioenergy production is an important emerging area of research (Wicke et al. 2012), the results of which will assist policy makers in understanding the potential impacts of its use.

Opportunity 3: Develop agreed international policy mechanisms and instruments for sustainable land use options for bioenergy crop production.

A sustainable land use policy for bioenergy production must be implemented through effective land use planning intervention. Bioenergy-driven land use changes often have been described as capitalist relationships between 'North' and 'South' in the political and social science literature (e.g. Borras et al. 2010). This review confirmed that in developed countries, land use planning was mostly well regulated, and there had been much less evidence of large-scale conversion of natural vegetation to bioenergy crops. There were also land constraints in many developed countries (e.g. EU), and this resulted in large-scale bioenergy crop production and investments in countries where agricultural expansion was still possible. The challenges were more acute in developing countries, where political, institutional, and enforcement capabilities are limited, land use legislation and planning can be ineffective, and economic development and private interests often take priority over environmental and sustainable land use policies.

Effective strategies for avoiding the negative effects of large-scale bioenergy crop expansion are required, including economic mechanisms and institutional improvements through international political action and cooperation, especially in countries where there is a lack of clear environmental and land use policies, a legal framework, and enforcement capability at the national level. Existing international climate policy, such as the emission accounting system under the Kyoto Protocol (IPCC 2006) has encouraged developed countries to import bioenergy products from developing countries, triggering the conversion of native vegetation to bioenergy crop production in developing countries (Schubert et al. 2010). However, emerging climate policies have introduced economic mechanisms, such as REDD under the UNFCCC. REDD aims to provide incentives to protect forests with high biodiversity value and high carbon stock, and is expected to influence future land use policy and planning worldwide. The mechanism is still being developed, and nonetheless, REDD has attracted various concerns and criticisms, including the imposition of long-term constraints on land use in certain areas, because it may affect local communities and cause displacement of deforestation to areas where REDD schemes are not active (Ghazoul et al. 2010). Its effectiveness also has been questioned because of its market-oriented nature (Nantha & Tisdell 2009) and dependence on various conditions such as additionality, leakage or permanence (Gawel & Ludwig 2011). However, the REDD mechanism is generally regarded as a positive step towards minimising the negative environmental consequences from future bioenergy-driven land use changes in developing countries (Nepstad et al. 2008; Gibbs et al. 2010).

Opportunity 4: Strengthen sustainability requirements and certification schemes.

Rapid developments are occurring in international markets, requiring agricultural producers to comply with sustainability requirements and certification criteria in order to participate in international commodity markets. The EU's RED has adopted certification criteria for biofuels, which include a prohibition on the use of those biofuels produced from biomass grown on land converted from forests, wetlands, or other high-carbon stock areas (e.g.

peatland), and highly biodiverse areas (European Union 2010). Thus biofuels used in the EU have to comply with certification criteria including iLUC. In the U.S.A., there are standards for biofuel sustainability in both the public and private sectors (e.g. Council for Sustainable Biomass Production [CSBP]), and this trend is likely to expand into other international markets. There are also international initiatives towards sustainable crop production across various stakeholders and their voluntary certification schemes, such as the Roundtable on Sustainable Palm Oil (RSPO), the Round Table on Responsible Soy (RTRS), Bonsucro (Better Sugarcane Initiatives), the Roundtable on Sustainable Biofuels (RSB), and the Global Bioenergy Partnership (GBEP). As the application of sustainability certification criteria is in its infancy, there are still significant uncertainties surrounding its effectiveness (van Dam, Junginger & Faaij 2010). The main challenges of certification criteria relate to weak application in emerging markets, implementation time and cost, inconsistency in the definition of terms (e.g. the distinction between primary and secondary forests, biodiversity-rich areas), diversification between initiatives in methodologies, parameters and default values, and limited attention on the quantification of possible biodiversity impacts (van Dam, Junginger & Faaij 2010; Wilcove & Koh 2010), and the uncertainty surrounding their ability to ensure compliance in producing countries (Tomei et al. 2010). However, certification schemes can be certainly one of the useful tools to stimulate sustainable land use on a local and regional level (van Dam, Junginger & Faaij 2010), and they have the potential to influence not only future land use policy but also its implementation and enforcement in bioenergy crop producing countries in coming decades. The environmental and social impacts of these certification schemes need to be comprehensively evaluated in the coming years.

2.5 Summary

This review and analysis revealed the differences in patterns and dynamics of bioenergy-driven land use changes on four regions. To date, the increased bioenergy demand on a global scale has directly and indirectly caused the conversion of native vegetation in parts of South America and Southeast Asia, and this has resulted in major environmental and socio-economic consequences. These pressures will spread into other regions and impact more severely on these ‘land- and resource-abundant’ countries such as Brazil over the next few decades. Until recently, there has been limited capacity for implementing sustainable land use policies and more effective planning frameworks in countries in these developing regions, due to political and institutional constraints. Consequently, trends in international commodity markets have been the major driver in these land use changes in most countries. Stronger emphasis should be placed on economic mechanisms to help develop future land use policy and implementation strategies for sustainable bioenergy crop production, with a focus on a more considered and sustainable choice of lands for future crop production. To achieve this, further research is needed to identify more suitable land for bioenergy crop production, especially the sustainability of the use of underutilised agricultural land for future bioenergy crop production. Opportunities for more sustainable outcomes are available through the development of international climate change policy (e.g. REDD) and certification criteria for sustainable bioenergy products (e.g. EU RED). However, zero or low land use change risk bioenergy feedstock (e.g. wastes and residues) and the associated technologies must be given high priority to minimise bioenergy driven land use change and its impacts in the long term.

Chapter 3: A framework for evaluating the environmental consequences of bioenergy-driven land use changes at a regional scale

3.1. Introduction

Bioenergy-driven land use changes can impact negatively or positively on the environment at a range of scales. The review of research reporting bioenergy-driven land use changes (Chapter 2) confirmed that the level of impact depended on a number of factors, particularly the land use prior to conversion, the level of native vegetation loss, crop selection and agricultural management practices. The environmental aspects that were affected included the following (Table 2.1 in Chapter 2):

- global climate change due to greenhouse gas (GHG) emissions;
- regional climate due to changes in vegetation cover, surface runoff and albedo;
- air quality from biomass burning;
- water quality and quantity;
- soil quality and erosion; and
- biodiversity.

This chapter develops a framework for evaluating regional scale environmental consequences associated with bioenergy-driven land use changes. The framework will address a minimum number of indicators that are potentially applicable to various regions. This research focused on the regional scale environmental impacts due to land use changes, which, until recently, have received less attention in the literature. It addressed the identified need for a spatially explicit and integrated framework to evaluate regional-scale environmental impacts of land use change to enable more environmentally sustainable land use decisions for future bioenergy crop production.

Graham et al. (1996) proposed the first framework to quantitatively evaluate the environmental impacts of large-scale switchgrass production in Tennessee, the U.S.A. However, this evaluation framework only captured water quality degradation. More recently, studies establishing environmental indicators and an assessment framework for bioenergy (especially biofuel) sustainability have emerged in the U.S.A. (Zhang et al. 2010; McBride et al. 2011; Efroymson et al. 2013) and Europe (van Dam et al. 2009; Langeveld et al. 2012). This has been in response to increased requirements for environmental sustainability in bioenergy production. For example, Langeveld et al. (2012) quantified the effects of Short Rotation Coppice (SRC) scenarios on soil quality, water and biodiversity (using 17 indicators) in the European context, and used spider charts to present the results. However, the method used was based predominantly on expert opinions/judgements and observations from the extensive field trials by the Swedish and German Governments (ERA-NET Bioenergy 2010). In reality, the same level of expertise is unlikely to be available for most bioenergy projects, and more importantly environmental sustainability needs to be predicted prior to crop production. Therefore, a framework/method is required to predict environmental impacts using a minimum number of commonly available datasets.

A spatially explicit modelling approach has strong advantages for the analysis of spatial data and mapping outputs. Zhang et al. (2010) developed a framework to explore trade-offs between bioenergy production and multiple environmental variables, such as biomass yield, GHG emissions, erosion, nitrogen (N) and phosphorous (P) losses. However, they noted that addressing biodiversity concerns was a challenge, and this was not included in their analysis. They assessed 54 scenarios of various biofuel crops (alfalfa, corn, soybean, miscanthus, native prairie, hybrid popular and switchgrass) and rotations, and concluded that no single spatial configuration could simultaneously optimize all the objectives (Zhang et al. 2010). More recently, van der Hilst et al. (2012a) presented an integrative framework to quantify the potential environmental impacts of miscanthus and sugar beet production scenarios in the Netherlands. By using GIS and spatial modellings, the results were presented as maps with scores as the indices, thus enabling areas with high risks to be indicated clearly. The framework included eight environmental impact areas (GHG emissions, soil quantity and quality, water quantity and quality, and biodiversity) and various methods to quantify the land use change impacts. The study attempted to assess the impacts of land use change scenarios for policy and decision making concerning future bioenergy crop production on a single region in the European context. Thus the framework involved a large amount of detailed spatial data (e.g. soil characteristics, climate, current land use, yield, crop management, fertiliser and manure inputs etc.), and included values and parameters for a model (for the nitrogen [N] and phosphorous [P] balance calculation) and other methods (for other impacts). Such high quality intensively research data, values and parameters are unlikely to be available for most global regions, including regions which currently are experiencing high land use and environmental stress from bioenergy crop expansion. In these cases urgent land use planning solutions are required (Chapter 2). Quantifying biodiversity impacts for universal application was also a shortcoming in this framework.

In this context, this research also developed a spatially explicit framework for evaluating the environmental effects of bionergy-driven land use changes, but it was designed to enable broader application to a range of geographical regions. In general, a framework requiring large amount of inputs, data and resources may have higher precision, but it limits future application to regions with less data availability and intensity, especially developing countries. This trade-off needed to be taken into consideration when designing the framework. This chapter outlines a framework that requires minimal inputs, data, resources and expertise, to generate information on a limited number of indicators, but which can provide reasonable predictive capacity. The framework serves as a foundation for decision making on bioenergy and land use policies and planning at the regional level, particularly for regions where data and resources are limited. For this reason, the evaluation framework in this research needed to focus strongly on key regional-scale environmental issues, especially biodiversity, in order to close an important gap in the current knowledge. In addition, it was envisaged this evaluation framework be flexible to accommodate additional indicators and tools to allow more 'integrative' evaluations, in instances where land use patterns are more complex and/or more data and resources are available.

3.2 Methods

In this Chapter, the evaluation framework is developed in four steps. The discussion starts with conceptualisation of the environmental consequences associated with bioenergy-driven land use change, identified from the review of global literature (2.3.6). Firstly, conceptual diagrams were developed to understand key environmental issues at all temporal and spatial scales, and to understand the cause and effect relationships. The next step focused on key issues that the evaluation framework should take into account at the local and regional scales. This process was followed by selection of key indicators to quantify the issues. The indicators were selected based on the literature review, the conceptual diagrams, and expert opinions. Lastly, the tools/models/methods to generate these indicators were selected using a set of criteria. These key steps are presented in this chapter.

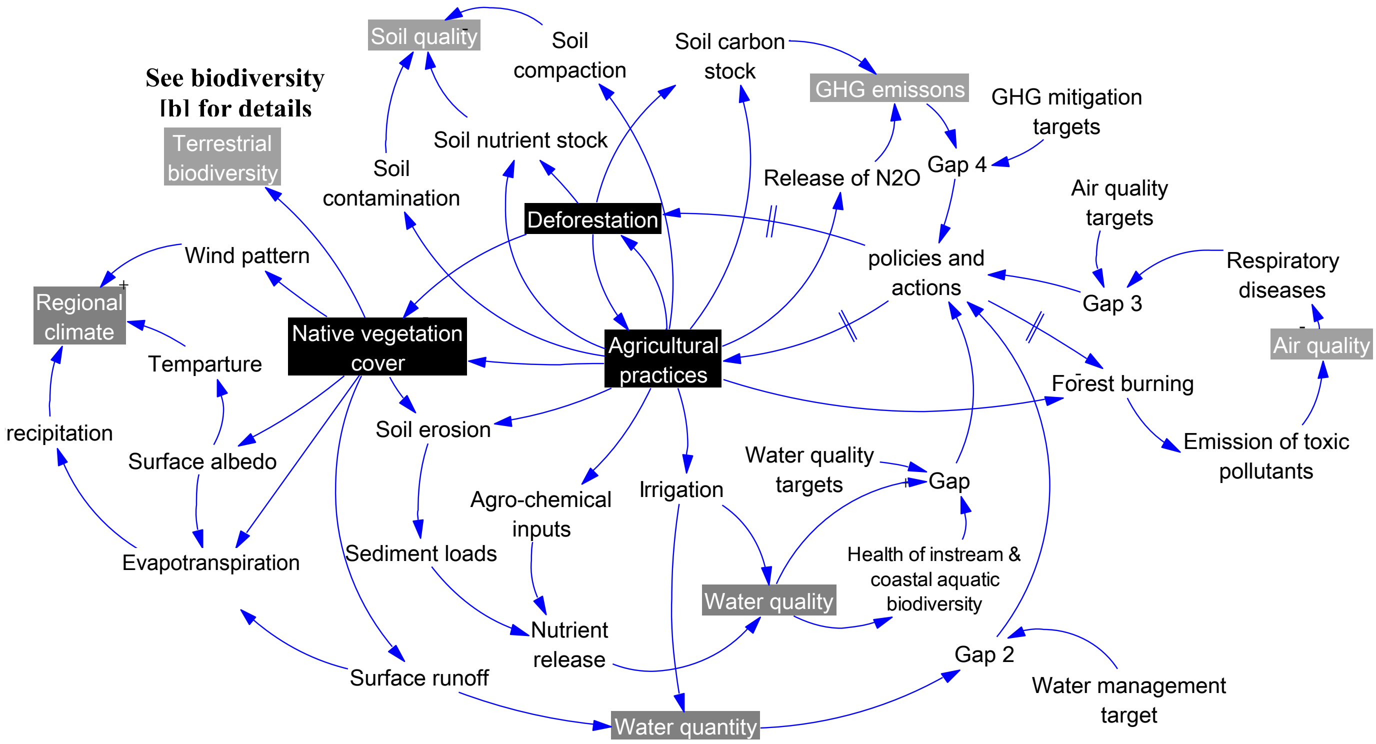
3.3 Regional scale environmental consequences

3.3.1 Conceptual diagrams

Based on a synthesis of global literature on land use change dynamics associated with bioenergy crops (Table 2.1) conceptual diagrams were developed to understand key environmental aspects and issues associated with bioenergy-driven land use changes, and their influencing factors. The diagrams illustrate the nature and direction of the relationships (e.g. cause and effect) that influence particular issues (e.g. biodiversity loss or water quality decline), and enable an enhanced understanding of the entire system. The diagrams illustrate the land use change pathways from forest/native vegetation to bioenergy crop production, because these processes are well documented in the literature and encompass the widest range of environmental issues on various spatial scales from global to site scales (Figure 3.1 [a] for air, water, soil and climate, and [b] for terrestrial biodiversity).

Interrelations between factors were also derived from the literature, describing the various environmental consequences of bioenergy-driven land use change. The resulting conceptual diagrams show that 'native vegetation cover' and 'agricultural practice' are the key factors associated with bioenergy-driven land use that directly influence environmental outcomes, because they can influence multiple environmental aspects, namely air/climate, water, soil and biodiversity. For example, change in native vegetation cover for agricultural activities (i.e. land clearing) increases the rate of the surface erosion (Sioli 1985; Fearnside 2005; Sawyer 2008), and surface runoff (Bosch & Hewlett 1982; Twine, Kucharik & Foley 2004), changes the surface albedo, temperature and evapotranspiration (Sampaio et al. 2007; Loarie et al. 2011), and then causes habitat loss and fragmentation for fauna and flora (Fahrig 2003; Abdullah & Hezri 2008). Agriculture practices also contribute to various environmental effects, including GHG emission from nitrogen (N) fertiliser through the emission of nitrous oxide (N_2O) (Crutzen et al. 2007; Hill 2007; Kim & Dale 2008), release of soil carbon from tillage practices (Tolbert et al. 2002; Kim, Kim & Dale 2009), air pollution from biomass burning (Martinelli & Filoso 2008; Sawyer 2008; Tsao et al. 2012), water consumption (Sawyer 2008; Tomei et al. 2010; Ugarte et al. 2010), soil erosion and sedimentation (Martinelli & Filoso 2008; Wu & Liu 2012), degradation of soil quality (Tomei et al. 2010) and water quality (Martinelli & Filoso 2008; Schilling et al. 2008; Ng et al. 2010) through loss of agrochemical inputs, and biodiversity loss due to conversion of native ecosystems to monoculture landscapes (Koh & Wilcove 2008; Sawyer 2008; Wilcove & Koh 2010).

[a] air, water, soil and climate



[b] biodiversity

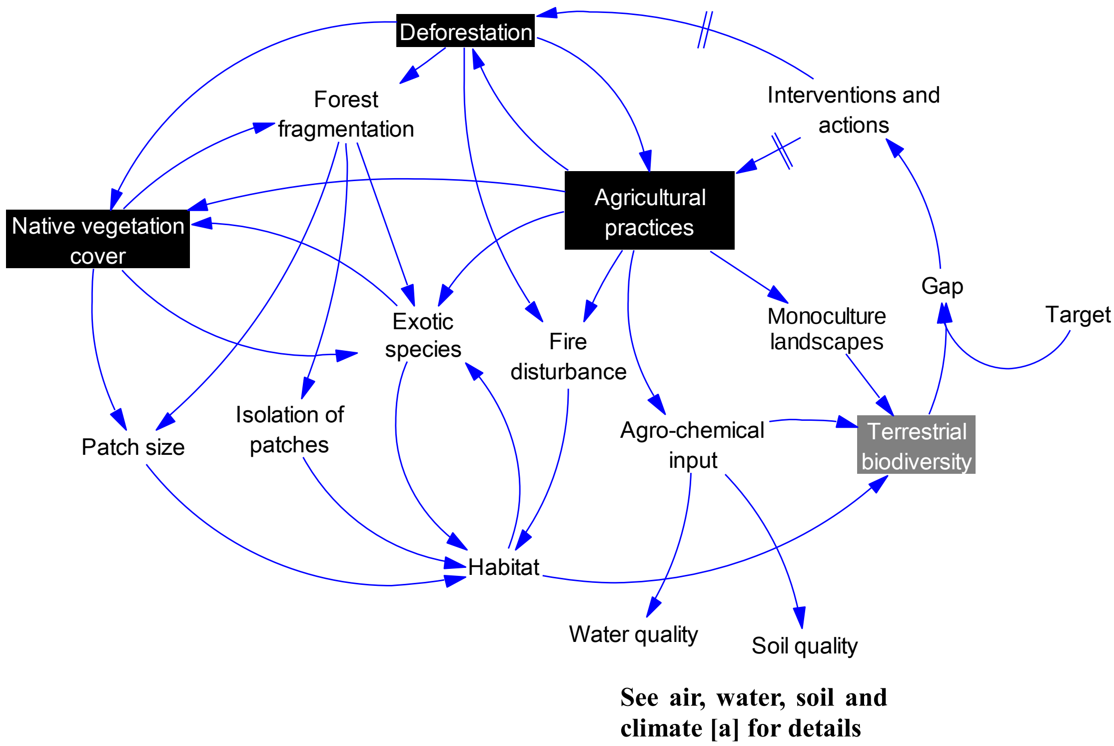


Figure 3.1 Conceptual diagrams illustrating cause and effect relationships and the consequences of deforestation from bioenergy crop production. 'Gaps' were identified where policy interventions and/or actions were required in the literature.

3.3.2 Environmental consequences at the regional scale

The main environmental consequences of bioenergy-driven land use changes were identified from the literature review (Table 2.1) and the conceptual diagrams (Figure 3.1). To address the identified gap in knowledge, the focus of the evaluation framework is on regional and local scale environmental issues (1.1.1 in Chapter 1). Major global trends in environmental and land use policy and decision-making have placed greater reliance on decentralised arrangements at the local and regional scales (Lane, Taylor & Robinson 2009). This framework was designed to provide an evaluation methodology for land use change and environmental impacts to support policy and/or decision makers at the regional level for future bioenergy land use options.

The following were identified as key issues at the regional-scale in relation to air/climate, water, soil and biodiversity (Table 2.1):

- air pollutant emissions (e.g. trace gas and particles) from biomass burning;
- changes to local and regional climate from broad-scale deforestation through change in temperature, surface albedo, evapotranspiration and also precipitation;
- change in water quality due to agro-chemical inputs and sediment, and change in conditions for the receiving-water ecosystem as a consequence;
- change in hydrological regimes (e.g. surface runoff) from land use change and agricultural/forestry activities;
- soil erosion from land use change and agricultural/forestry activities; and
- change in native vegetation cover, structure and quality, and habitats of particular species.

While all of the above issues are important, the evaluation framework in this research focused on the issues of water quantity and quality (and soil erosion), and terrestrial biodiversity, the impacts of which are commonly experienced regardless of geographical region. In contrast, the issues in relation to air pollution from biomass burning, and regional climate change were reported to be more area-specific in the context of bioenergy driven land use changes. For example, the emission of air pollutants was mostly related to burning vegetation to establish oil palm plantations in the Southeast Asia, particularly in Indonesia (Varkkey 2012, 2013), or the burning of sugarcane before harvesting in Brazil (Goldemberg, Coelho & Guardabassi 2008; Martinelli & Filoso 2008; Tsao et al 2012). For the same reason, impacts on local and regional climates were thought to be beyond the scope of this evaluation framework. Although this is an important and complex research area with extensive literature, to date most of the attention has focused on the Amazon rainforest where large-scale deforestation is currently of concern at both global and regional levels (Oyama & Nobre 2004; Costa et al. 2007; Sampaio et al. 2007; McAlpine et al. 2009; Hirota, Oyama & Nobre 2011). Nevertheless, all these studies revealed a strong relationship between regional climate change and vegetation cover change, since the latter directly affects radiation, energy (e.g. albedo, heat flux, temperature), and water balance (e.g. evapotranspiration [ET] rates, runoff volume) at the surface (Oyama & Nobre 2004; Sampaio et al. 2007). Thus indicators for vegetation cover change, which also are linked closely with water quality and quantity and biodiversity, could be used to indicate the potential risks associated with regional climate change in the regions where the phenomenon is strongly experienced as a result of land use changes. This is because the change in vegetation cover affects surface albedo, wind patterns and evapotranspiration and results in changes in temperature and precipitation patterns (Figure 3.1).

The detailed selection process in relation to the relevant environmental sustainability indicators is discussed in the following sections. Indicators for area-specific issues could be included as options for future application of the framework depending on local concerns and purposes for measurement. However, they were beyond the scope of this research.

3.4 Evaluation framework

The development of the evaluation framework involved two processes: selection of appropriate indicators for the identified environmental consequences, and selection of appropriate tools/models for evaluating the indicators.

3.4.1 Environmental sustainability indicators

The key regional scale environmental consequences (3.3.2) were translated into relevant environmental sustainability indicators for incorporation into the evaluation framework. Efroymson et al. (2013), who synthesised a number of past studies, stated that indicators can be used to assess, communicate and compare the status of the environmental outcome, sometimes with respect to a target; to monitor trends; to provide early warning signals of changes; or to provide evidence concerning causes of observations. For this framework, the indicators were used to assess, communicate and compare the status of the environmental sustainability quality of various land use scenarios. The selection of indicators was based on the following criteria:

- relevant to regional scale environmental consequences;
- commonly used and effectively representing the impact of each environmental aspect; and
- providing adequate historical data and information and accessible at appropriate spatial and temporal scales.

Based on these criteria, indicators for water quantity and quality (and soil erosion) and for terrestrial biodiversity were determined as described below.

3.4.1.1. Soil erosion, water quantity and quality

Runoff volume, and sediment (from soil erosion), nitrogen and phosphorous loads in streams have been the most common indicators used to date to evaluate impacts on water quantity and quality due to bioenergy feedstock production (Schilling et al. 2008; Ng et al. 2010; Love & Nejadhashemi 2011; Efroymson et al. 2013). Past hydrological studies have affirmed that land use changes from native vegetation to other human-induced land uses increase: the overland flow of storm runoff; suspended sediment; and nutrient exports in surface water (Williams & Melack 1997; Brodie & Mitchell 2005). Thus these were taken into account when selecting indicators for this framework. Such hydrologic alteration and changes in sediment and nutrients in surface water have led to significant impacts on both stream (Allan 2004) and receiving-water ecosystems (e.g. river reaches, lakes, wetlands, mangroves and coral reefs) worldwide.

Runoff volume is a common indicator of land use change effects on water yield. Land use change can alter vegetation cover and structure, and surface hydrological process (e.g. evapotranspiration [ET], soil evaporation) and groundwater levels (Schilling et al. 2008; Le, Kumar & Drewry 2011). The increased runoff from agricultural land use per se is not a critical issue (Schilling et al. 2008), unless there is a major increase, such as that resulting from large-scale deforestation (e.g. the Amazon forests). However, increased runoff also has the potential to increase the risks of flooding, soil erosion, and the pollutant delivery through nitrogen and phosphorous exports to adjacent water bodies (Schilling et al. 2008).

Sediment load is an important indicator of both soil erosion and water quality, as erosion is the source of 99% of the total sediment loads in waterways around the world (Nearing, Norton & Zhang 2001). There are higher rates of soil erosion on land cleared for agriculture if this is combined with poor land management, and topographical and climate factors (Abdullah & Hezri 2008; Sawyer 2008). Thus land use change for agricultural activities could result in a significant increase in sediment discharge in freshwater, estuarine and coastal marine waters (Brodie & Mitchell 2005). Seagrass meadows and coral reefs are particularly susceptible to increased levels of sediment and nutrient discharge in water from terrestrial sources (Short & Wyllie-Echeverria 1996; Fabricius 2005). Nutrients, such as nitrogen and phosphorous, are also transported in sediment to downstream waters. They are essential elements in natural resources, but again human-induced land use change, such as pasture, cropping and urban uses, has led to excessive amounts of these nutrients which have resulted in eutrophication of water systems worldwide (Brodie & Mitchell 2005). Therefore, these nutrient loads are key indicators for water quality. In particular, it is commonly understood that excessive inputs of nitrogen and phosphorous fertilisers on agricultural land are one of the most serious and widespread sources of water pollution in many countries since the 1960s (Tilman et al. 2001), particularly in the catchments in which agricultural land uses are dominant (Puckett 1995; Carpenter et al. 1998; Sharpley 2002).

In summary, run-off volume (million litres [ML] per year) was selected as an indicator for change in water quantity. Total nitrogen and phosphorous (TN and TP) (tonnes per year) were selected as indicators for water quality, and total suspended solids (TSS, sedimentation load) (kilotonnes per year) was selected as an indicator for both soil erosion and water quality.

3.4.1.2 Terrestrial biodiversity

This evaluation framework focused on the direct effects of land use changes on biodiversity. To quantify the impacts, two different approaches were undertaken at regional scale: analysis of the spatial pattern and configuration (structure and composition) of native vegetation; and calculation of total biodiversity conservation value of the region. For the first approach, two different input maps were used to understand landscape processes associated with different native vegetation communities; and native vegetation groups as defined by their conservation and/or biodiversity status (e.g. endangered). The latter was important particularly to help understand the impacts on rare and threatened species and ecosystems in the region.

Spatial pattern of native vegetation and landscape process analysis

Habitat loss from clearing is the primary cause of terrestrial biodiversity decline worldwide (Tilman et al. 1994; Fahrig 2001; Ewers & Didham 2006), and this is followed by habitat fragmentation (Fahrig 1997, 2003; Ewers & Didham 2006). Intact native vegetation, in the form of primary forests, woodlands, savannah, grassland and natural wetlands, plays a critical role in multiple ecosystem services. Native vegetation has high biodiversity conservation value and provides habitats for rare and threatened species of flora and fauna. Its destruction or modification, as a result of human-induced land use change, has led to significant habitat loss and fragmentation across the world. Thus, analysis of the spatial pattern of native vegetation cover over different spatial and temporal scales enables examination of these landscape change processes and the extent of suitable habitat for these species of special concern which depend on a functioning landscape.

Habitat loss can be measured by the change in the total area of native vegetation remaining in a region (Fahrig 2003). This is one of the simplest and most widely understood indicators available. Habitat fragmentation can be measured by using a combination of various landscape metrics. Although the definition of landscape fragmentation varies, this includes a combination of the following effects: decrease in amount of habitat; increase in the number of habitat patches; decrease in the size of habitat patches; and increase in the isolation of patches (Fahrig 2003). Therefore, three indicators were selected for the evaluation framework: total area of native vegetation remaining; patch size; and the number of patches. The total area of different types of native vegetation is a critical indicator to assess the degree of habitat loss, while all indicators are important for both habitat loss and habitat fragmentation. These indicators can be expressed in different landscape metrics depending on the tools to be used for the analysis.

Habitat quality can be assessed using the same indicators, but requires different input maps based on the conservation and/or biodiversity status of different ecosystems/communities (e.g. endangered). Thus two different vegetation maps classified by native vegetation communities and by native vegetation groups as defined by their conservation status are required to generate information for the selected indicators.

Biodiversity conservation value of human-induced land uses

Biodiversity conservation value of different agricultural lands must be taken into consideration when evaluating the impacts of bioenergy-driven land use changes. To date, studies in landscape or spatial ecology have investigated the effects of vegetation clearing and fragmentation, and those of intensity of human activities (e.g. grazing, logging) on terrestrial fauna (especially bird species) around the world. Studies have emerged in recent years to explore the potential role of bioenergy crops for biodiversity conservation. All studies assumed that human-induced land use change impacted on biodiversity to various degrees, depending on the type of land use combined with a number of factors relevant to individual sites. These may include climate, vegetation types, site location and surroundings (e.g. next to road), intensity of activities, local species in the region, their habitat preferences and behaviour characteristics, and so on.

Vegetation clearing for cropping, grazing and urban development results in significant changes in the species diversity and composition. More specifically, the landscape ecology literature (Martin & Catterall 2001; Martin et al. 2006; Collard, Le Brocque & Zammit 2009; Eyre et al. 2009) has commonly found that species depending on native vegetation and/or sensitive to structural and micro-climatic changes are in decline in response to habitat loss and fragmentation, while 'generalist' and 'open or developed land' species have increased in abundance corresponding with the expansion of cleared lands. The generalist species include both native and exotic, and are generally of low conservation status due to their abundance. For this reason, the biodiversity conservation value of cleared land is rated much less than that of intact native vegetation (Green & Catterall 1998). Nevertheless, as stated previously, the value significantly varies depending on the land uses and other factors. For example, cropping land was documented to generally support very few species due to its intensive and homogeneous nature (Martin & Catterall 2001; van Rooij 2008; Alkemade et al. 2009). In comparison, urban areas such as suburban residential areas could support higher abundance and richness of bird species, containing generalist and/or open or developed land species (Martin & Catterall 2001). In relation to grazing area and forest plantation, Eyre et al. (2009) found in their bird survey that biodiversity conservation value was closely linked with the intensity of grazing or logging activity and the land management practices, which affected species compositions (diversity and abundance) in the landscape. Studies conducted in southern Queensland, Australia, documented a clear relationship between the intensity of grazing and logging activities and the abundance of a small number of larger bodied generalist forager bird species (e.g. noisy miner, rainbow lorikeet, torresian crow, Australian magpie, grey butcherbirds, which are common in Australian urban landscapes). These larger birds displaced the habitats of a wide range of small passerine species of higher conservation status (Kanowski, Catterall & Wardell-Johnson 2005; Martin et al. 2006; Eyre et al. 2009). Smith and Agnew (2002) also suggested that mature plantation forests such as dry eucalypt plantations in southeast Queensland could provide habitats for microbats and arboreal mammal species (e.g. gliders). On the other hand, Kanowski et al. (Kanowski, Catterall & Wardell-Johnson 2005; Kanowski et al. 2006) argued that eucalypt plantations in southeast Queensland had no intrinsic biodiversity value in terms of rainforest biodiversity (e.g. bird and reptile species), and land uses such as modified pasture played a limited role in supporting rainforest-dependent species, particularly reptiles. In general, homogeneity and monoculture tends to reduce biodiversity. Overall, these studies indicated that human-induced land use change impacted on the original species compositions in the native vegetation, but also human-modified landscapes could retain some biodiversity conservation values for certain species as long as they were managed carefully and the disturbance was minor.

Biodiversity conservation value for bioenergy crop production is also of increasing interest in recent bioenergy studies, including in Europe and the U.S.A. The literature review (Chapter 2) identified that the land use before conversion to bioenergy production, and the crop selection and associated management practices are key factors in determining environmental sustainability outcomes including the conservation of biodiversity. In this context, the conversion to lignocellulosic crops such as Short Rotation Coppice (SRC) in Europe and native perennial plants such as switchgrass [*Panicum virgatum L*] in the U.S.A. has attracted high attention in relation to restoring biodiversity conservation values. Despite a wider view that the biodiversity benefits of monoculture or homogeneous characteristics of bioenergy plantations are limited, several studies reported that perennial crops (e.g. miscanthus [*Miscanthus x giganteus*]) (Semere & Slater 2007; Bellamy et al. 2009), SRC in Europe (Rowe, Street & Taylor 2009; Rowe et al. 2011; Langeveld et al. 2012), and native perennial crops in the U.S.A. (Dale et al. 2010; Robertson et al. 2011) could provide higher biodiversity conservation values than bioenergy production from conventional annual crops. Nevertheless, crop/forest management practices (e.g. harvesting method/rotations, tillage frequency, biomass residue removals) at the field/plantation could considerably reduce biodiversity conservation values (Dale et al. 2010; Hartman et al. 2011; Robertson et al. 2011). For example, intensive biomass residue removal at SRC plantations could significantly impact on many types of species that depend on it as important habitat (Bouget, Lassauce & Jonsell 2012; Lattimore et al. 2013).

Although the impact of bioenergy production on biodiversity conservation value is highly complex, involving a number of factors, the reviewed European bioenergy studies attempted to incorporate the biodiversity conservation values for different land use classes into their spatial framework. Van Rooij (2008), Dornburg et al. (2008) and van der Hilst et al. (2012a) employed the mean species abundance (MSA) value, which was originally used to quantify biodiversity loss in areas that were disturbed by human activities, relative to their abundance in primary vegetation (Alkemade et al. 2009). The values represented the average response of the total set of species in each land use class, and were used to calculate the impact of land use on the remaining biodiversity for a prolonged period (Alkemade et al. 2009). Despite limitations due to over-simplification and the subjective nature of the values, the concept of MSA values facilitates understanding of change in biodiversity associated with land use changes and provides a good overview. Hence a similar concept was adopted in the evaluation framework as an additional indicator for biodiversity, since the literature suggested that biodiversity conservation values outside the native vegetation area (including bioenergy production) were not negligible.

3.4.2 Models and/or tools

The next step was the identification of suitable models and tools for assessing the selected indicators. More specifically, a spatial hydrological model and a spatial landscape pattern tool were required to generate numbers for all indicators in the evaluation framework. Spatial hydrological models can quantify the run-off volumes, sediment and nutrient loads by using spatial data including land use datasets. Spatial landscape pattern tools can quantify the landscape processes (i.e. native vegetation loss and fragmentation) that impact on terrestrial biodiversity in the region. The outputs of the spatial landscape pattern tools are landscape metrics, such as total area of patches (e.g. different native vegetation groups), patch size, and number of patches in the landscape, which help to explain the landscape processes before and after the land use changes.

The most important issue relating to the model and tool selection was that they must have been tested and verified in a wide range of contexts (Efroymson et al. 2013). Thus the following sections review the available models/tools to identify those that were most suitable to assess the above environmental sustainability indicators.

3.4.2.1 Spatial hydrological models

When selecting the hydrological model for the evaluation, several factors were taken into account including: spatial and temporal capability; appropriate level of model complexity; level of data/empirical parameter requirement; and applicability to a wide range of catchments. Firstly, the model had to have strength in a spatial analysis context; and be capable of part/full integration into GIS, to allow users to take advantage of its capabilities in spatial data acquisition, storing, retrieval, processing, manipulating and visualising (Malczewski 2004). The second and third criteria were highly linked because a high quality model output generally needs sufficient calibrated data/parameters and a high level of expertise for development and calibration (eWater CRC 2011). Thus the model needed to present a flexible structure that enables users to decide the appropriate level of model complexity and predictive performance depending on data availability. This is because the model had to be adaptable to different situations and catchments for future applications, including a region with limited data and technical availability.

An extensive review of the literature identified representative spatial hydrological models that addressed the above requirements. They were designed to evaluate and project land use change effects on water quantity and quality at a catchment scale (Table 3.2). The Soil and Water Assessment Tool (SWAT), which was developed by Dr. Jeff Arnold for the U.S. Department of Agriculture (USDA) Agricultural Research Service (Arnold et al. 1998), is the most widely applied model. SWAT is a physically based, long-term, continuous, catchment scale simulation model with daily time steps, and full integration with ArcGIS (e.g. ArcSWAT was developed as an ArcGIS extension). By the mid-2000s, over 250 peer-reviewed published articles had reported applications using this model and had reviewed its components (Gassman et al. 2007). Its validation and scenario application have been reported worldwide particularly in U.S. catchments, for a wide range of catchment scales and environmental conditions. To date, a few applications to Australian catchments have been reported, including the Barr Creek catchment (Githui, Selle & Thayalakumaran 2012) and the Woady Yaloak River catchment (Watson et al. 2005) in Victoria, and the Liverpool Plains in New South Wales (NSW) (Sun & Cornish 2005, 2006). The model continues to improve, based on this extensive testing and application (SWAT Development Team 2013).

One of the strong advantages of SWAT is its predictive performance. SWAT has been reported by many studies as a useful tool in predicting the environmental implications of agricultural land use change reasonably well. The popularity of this model was indicated by applications to U.S. catchments in recent bioenergy-driven land use change studies (Love & Nejadhashemi 2011b; Ng et al. 2010; Schilling et al. 2008; Wu & Liu 2012). However, the downside is its complexity (U.S. Environmental Protection Agency 1997). SWAT requires a significant amount of data and empirical parameters for development and calibration (Borah & Bera 2004), many of which are not available or limited outside the U.S.A. (Zhang, Srinivasan & van Liew 2008). Complex models generally involve more processes and require a higher level of expertise. This is an obstacle to the timely and cost-effective use of these models. Sufficient high quality observed data is needed for model calibration, but data paucity is often experienced in many catchments in terms of climate, flow and water quality data, including the Australian catchments. Using a complex model does not necessarily contribute to better prediction, unless sufficient calibration data are available (eWater CRC 2011).

HSPF is the U.S. EPA’s hydrological model within the BASINS modelling framework, responsible for simulating a comprehensive range of hydrologic and water-quality processes. Im et al. (2007) evaluated its simulation components as being more accurate, but also more complex, than SWAT. This means that it requires extensive calibration and a high level of expertise to apply (U.S. Environmental Protection Agency 1997; Yang & Wang 2010). The model has the capability of simulating sub-surface water quality processes in both pervious and impervious lands, and thus it is more suitable for the simulation of urbanised catchments (Yang & Wang 2010).

Table3.1 Summary of representative spatial hydrological models to evaluate land use change impacts on water quality and soil erosion.

Model	Reference	Scale & application	Pros & cons	Main input data	Main output data
SWAT (Soil and Water Assessment Tool)	Arnold et al. (1998)	• catchment • a large number of applications especially in the U.S. catchments, including Love & Nejadhashemi, (2011b)	• Reliable predictive performance • Complex model, requires a significant amount of data and parameters	• land use • soil series • climate • agricultural practice (e.g. livestock numbers, tillage, fertilisation, cropping data, yield)	• water yield • nutrient yields • sediment yield • pesticide fate and transport • evapotranspiration (ET) • plant growth etc.
HSPF (Hydrological Simulation Program)	US EPA (n.d.)	• catchment • introduced in 1996 by US EPA, for use by regional, state, and local agencies in performing catchment and water quality- based studies	• High level of accuracy • High model complexity, requires a high level of expertise for model application, and limited application outside the U.S.A.	• land use • hydrography data • land management • hydrologic data • water quality data	• runoff volume • sediment, nutrients, pesticides, and toxic chemicals • climate change scenarios (generated as input of HSPF)
Source (previously known as Source Catchments)	eWater CRC (2010)	• catchment • released in 2012 based on the upgrade of the previous version, Source Catchment.	• Flexible structure • Limited number of applications to catchments outside Australia	• land use/land cover/vegetation/soil • DEM (topography) • climate data	• runoff volume • pollutant loads

In comparison, *Source* (previously known as *Source Catchments*) was developed to provide a balance between model complexity, data availability, and predictive performance, by providing a flexible structure and a range of algorithms to perform specific hydrological tasks (eWater CRC 2011). eWater Cooperative Research Centre (eWater CRC), Australia, launched *Source* in May 2012 to help catchment managers and researchers investigate a wide range of hydrological issues (eWater CRC 2011). It has a spatial analysis feature linking with GIS, allows the selection of the degree of model complexity, and can be applied at a range of spatial and temporal scales. The strong advantages of *Source* compared with other equivalent spatial models are: less complexity and more flexible structure. This allows users to build a simple model in the *Source* platform that requires the minimum number of parameters where few data and parameters are available. As this evaluation framework is targeted for future application to any geographical location in the world, these qualities are essential. The model has been applied to a number of important hydrological projects for Australian catchments. These include the modelling and evaluation of water quality impacts on the Great Barrier Reef (GBR) under the Paddock to Reef program, which is part of the Reef Water Quality Protection Plan 2009 (Reef Plan) (Queensland Government 2009a), and the simulation of the comprehensive system of the River Murray to improve water management, and distribution and delivery decisions by the Murray-Darling Basin Authority (MDBA) (eWater CRC 2012b). The model's application outside Australia is also in progress, as evidenced by its recent applications to projects in Singapore and Lake Tai basin (the border of Jiangsu and Zhejiang provinces) in China (eWater CRC 2012a). As this model equally meets the selection criteria, *Source* was selected to quantify runoff volume, sediment and nutrient loads from various land use scenarios for this evaluation framework.

3.4.2.2 Spatial pattern analysis tool

The selection criteria for the spatial pattern analysis are simpler than those for the spatial hydrological model and were selected based mainly on spatial and temporal capability.

To begin with, the review of past studies identified four main tools (Table 3.2). Among those tools, *Fragstats* has been the most widely used among professionals since 1995, and it has been upgraded regularly due to its popularity (as of July 2012, version 4). It is a free software program, developed by Dr. Kevin McGarigal, to analyse the physical characteristics of landscape structure, such as habitat loss and habitat fragmentation (McGarigal & Marks

1995). *Fragstats* quantifies a wide variety of landscape metrics that represent the spatial characteristics of patches, classes of patches, and/or landscape mosaics (McGarigal & Marks 1995). Patch Analyst is a modified version of *Fragstats* and was developed as an extension to ArcGIS. Therefore, *Patch Analyst* is easier to operate in the ArcGIS environment, although it provides a limited number of metrics. Another issue with *PatchAnalyst* is that the current version (version 4.0 that accommodates ArcGIS10) calculates spatial statistics on only polygon (vector) files. That may be problematic when calculating edge metrics, as it includes all the edge, including the boundary edge. MSPA-GUIDOS has been developed recently based on mathematical morphology to overcome the statistical weakness of *Fragstats*, such as the landscape metrics for connectivity (Soille & Vogt 2009). However, its application is limited to binary maps (i.e. habitat and non-habitat) so that it requires an over-simplification of the landscape on input maps (Vogt 2010). Thus it was not suitable for analysis of several classes (i.e. vegetation communities, and vegetation in different conservation statuses) in this evaluation framework.

The Global Biodiversity Model framework (GLOBIO3) was developed recently to assess the consequences of different land use scenarios for biodiversity (Alkemade et al. 2009). A few applications of this model in different geographical locations (e.g. Thailand, Zambia) were reported in European studies (van Rooij 2008; Trisurat, Alkemade & Verburg 2010). This model calculates the mean species abundance (MSA) relative to the species abundance in undisturbed vegetation. It is based on the cause-effect relationships, and presents changes in abundance of local species (Alkemade et al. 2009). Because MSA itself does not represent the selected indicators for this evaluation framework, GLOBIO3 was not suitable for this environmental evaluation framework.

As a result of these considerations, *Fragstats* 4.0 was selected to calculate the selected indicators. The tool has a number of strong advantages, such as its versatility, applicability, simplicity and effectiveness in terms of calculating the selected indicators. It requires a land-cover map or vegetation map as the only input for the tool, and this dataset is readily available for most areas in the world. In addition, it is capable of generating a wide range of metrics, including Total Class Area (CA), Largest Patch Index (LPI), and number of patches (NP). CA is sum of the areas (ha) of all patches belonging to a given class (e.g. native vegetation community) so that the change in CA for native vegetation classes between different years indicates the change in the amount of habitats for species that depend on these vegetation classes. LPI refers to the percentage (%) of the total landscape that is made up by the largest patch that represents the patch size, and is expressed between 0 and 100 by *Fragstats*. NP is the total number of patches for an individual class on the map (UMass Landscape Ecology Lab n.d.). A decrease in the CA and LPI of a native vegetation class commonly indicates habitat loss. However, habitat fragmentation requires interpretation of NP and these metrics. For example, a decrease in CA and LPI and an increase in NP also indicate habitat fragmentation. In general, information on NP alone does not have any interpretive value because it has no information about area, distribution or shape of the fragments (McGarigal and Marks 1995). For this reason this index was calculated together with CA and LPI to enable enhanced interpretation of the data.

Table 3.2 Summary of representative spatial pattern analysis tool to evaluate land use change impacts on biodiversity.

Model	Reference	Scale	Application, pros & cons	Input data	Output data
GLOBIO3	Alkemade (2009)	various (global and regional)	• Developed in 2009 based on the previous version. A few applications in different locations by European studies. • Its output (MSA) quantifies biodiversity loss on areas disturbed by human activities. • Lack of capability of evaluating the selected indicators.	• land use/intensity • infrastructure development • fragmentation • atmospheric nitrogen deposition • climate change	• remaining mean species abundance (MSA) of original species
MSPA-GUIDOS (Morphological Spatial Pattern Analysis)	Soille & Vogt (2009)	various (global, continental, regional and local)	• Developed to improve the statistical reliability and the existing index for connectivity of *Fragstats*. • Strength in landscape metrics, especially connectivity. • Very limited application, and limited outcomes.	• binary land use and land cover map (i.e. habitat/ non-habitat)	• segmentation of the input maps into MSPA classes (i.e. core, islet, edge, loop, bridge and branch)
Patch Analyst	McGarigal & Marks (1995)	various (regional, local and on-site)	• A modified version of *Fragstats*. Extension of the ArcView GIS for quantifying landscape structure. • Easy to operate in the ArcGIS environment. • Limited number of metrics compared to *Fragstats*.	• land use or land cover map (vector)	• values for various landscape metrics (e.g. patch size)
Fragstats	McGarigal & Marks (1995) McGarigal et al. (2002)	various (regional, local and on-site)	• Released in 1995. The most widely used program to compute a wide variety of landscape metrics for categorical map patterns. • A wide range of metrics to present spatial patterns and configuration.	• land use or land cover map (raster)	• values for various patch, class and landscape metrics (e.g. patch size)

3.4.2.3 Biodiversity conservation value

In addition to the spatial pattern analysis of native vegetation area by *Fragstats*, biodiversity conservation value for the entire region was quantified in this evaluation framework. This is because the spatial pattern analysis by *Fragstats* did not cover change in biodiversity conservation value on lands modified by human uses and activities.

Although GLOBIO3 was not selected as a tool for the evaluation framework, its concept of mean species abundance (MSA) is valuable and is easily applied to any geographical region. MSA is an index to represent the average response of the total set of original species relative to their abundance in native vegetation (Alkemade et al. 2009). Although MSA may be limited due to issues of oversimplification (Alkemade et al. 2009; van der Hilst et al. 2012), the values have been adopted by European bioenergy researchers (van Rooij 2008; Dornburg et al. 2010; van der Hilst et al. 2012) to quantify the biodiversity conservation values on different land classes and impacts of bioenergy-driven land use changes on biodiversity (Table 3.3).

Table 3.3 Generic MSA values for various land use classes (Source: van Rooij 2008)

Land use class	Generic MSA value
Primary forest	1
Light used primary forest	0.7
Secondary forests	0.5
Forest plantations	0.2
Agro forestry	0.5
Extensive agriculture	0.3
Perennials & biofuels	0.2
Intensive agriculture	0.1
Native grass & shrub lands	1
Livestock grazing	0.7
Human made pastures	0.1
Natural bare, rock and snow	1
Built up areas	0.05
Natural inland water/artificial water/ river/stream	-

The values must be modified to ensure effective regional application. This entails extensive literature research and the incorporation of expert opinions, and must be linked to land use classes on the land use maps. In this evaluation framework, these values were employed as weighted values to calculate the total biodiversity conservation value of the entire region as 'actual habitat amount'. It was calculated by using CA (obtained by *Fragstats*) and the following equation:

Actual habitat amount of the area (ha) = $\sum_{i=1} BCV_{LUi} \times CA_i$

Where BCV LU*i* is the biodiversity conservation value of land use *i* represented by a number between zero and one (with one being a biodiversity value the same as native vegetation and zero being no biodiversity value), and CA*i* is the land area of land class *i*.

The actual habitat amount gained from the above procedure was compared between different land use change scenarios to understand the change. The detailed method for this calculation and application is presented in Chapter 5 (5.2.8).

3.5 Summary

The aim of this chapter was to develop a framework for evaluating regional scale environmental consequences associated with bioenergy-driven land use changes. The framework reported on a minimum number of indicators that allow for universal application. Key regional environmental sustainability aspects and issues associated with bioenergy-driven land use changes were identified using conceptual diagrams (Figure 3.1[a] and [b]). The environmental sustainability issues on a regional-scale were identified as air/climate, soil erosion, water (quantity and quality), and biodiversity. However, water quantity and quality (and soil erosion), and terrestrial biodiversity were selected as the focus areas of the evaluation framework since the other issues were regarded as rather area-specific in the literature.

Next, the development processes of the evaluation framework were explained in this chapter together with justifications for the selection of indicators and models/tools. Firstly, the key regional environmental sustainability issues were directly translated into relevant indicators (Figure 3.2). Runoff volume, sediment (total suspended solids [TSS]), nitrogen (total nitrogen [TN]) and phosphorous (total phosphorous [TP]) loads were selected for the water quantity and quality indicators, since they are commonly used when assessing the effects of human-induced land use change on receiving waters worldwide. TSS also plays an important role as an indicator of soil erosion. To quantify these indicators, *Source* was identified as the most suitable spatial hydrological model due to its flexible platform and potential to equally satisfy the selection criteria. In addition, four indicators for biodiversity were identified, namely, total area (Class Area [CA]), size and number of patches (Largest Patch Index [LPI] and number of patches [NP]) of native vegetation, and biodiversity conservation value. The first three were identified as the most important metrics/indicators to describe the process of habitat loss and habitat fragmentation. To quantify the three indicators, *Fragstats* (version 4.0) was selected as the best tool because of its strong versatility, capability, simplicity and applicability to various spatial and temporal scales. In addition, the total biodiversity conservation value was included as the last indicator for biodiversity. This analysis is needed because the spatial pattern analysis using *Fragstats* does not capture the non-native vegetation areas, which have been modified by human use and disturbed by human activities. The concept was developed to quantify biodiversity loss on areas disturbed by human activities, relative to their abundance in primary vegetation (Alkemade et al. 2009), and in this evaluation framework, the concept was employed as weighted values to calculate the total biodiversity conservation value of the entire region as 'actual habitat amount'.

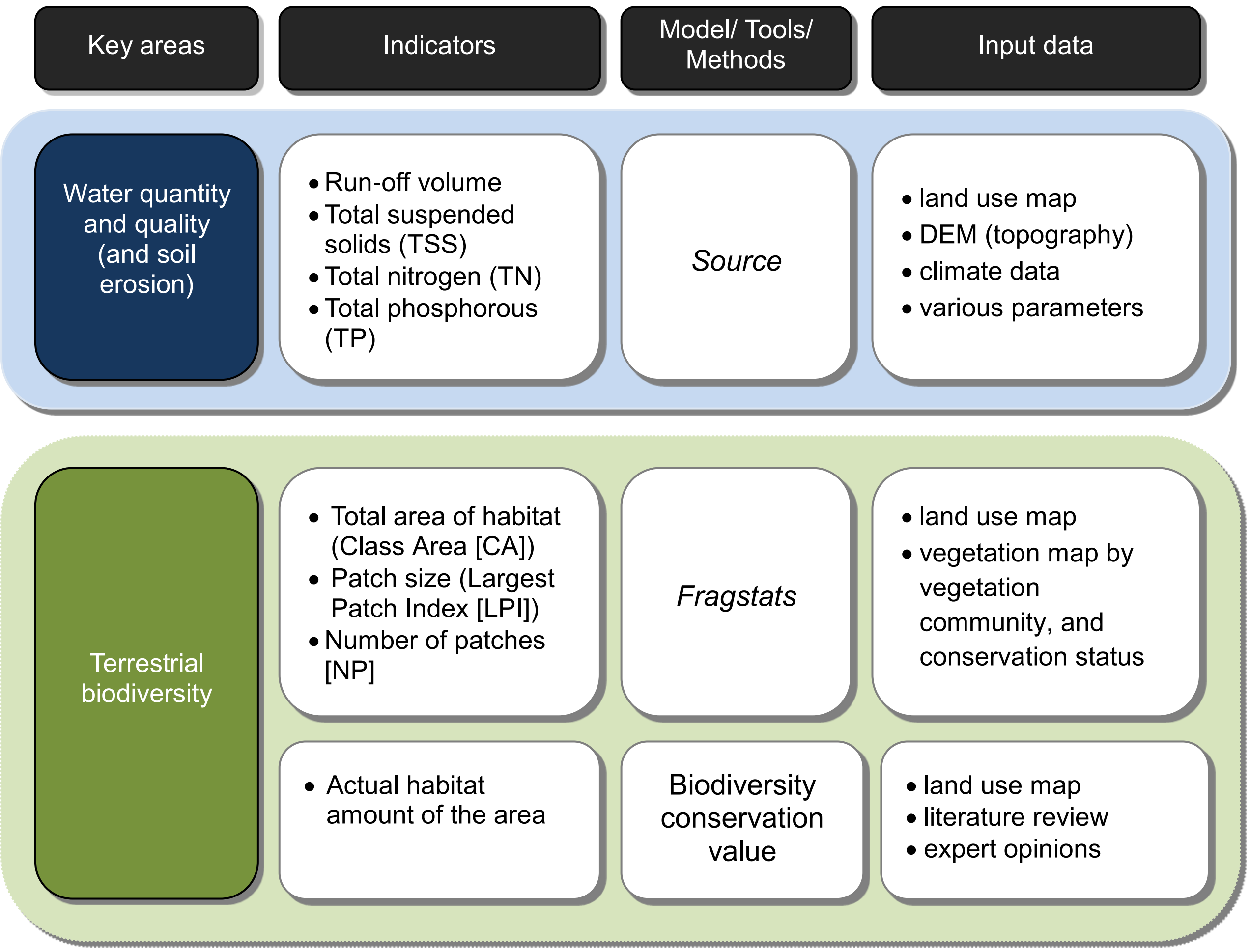


Figure 3.2 A framework for evaluating the environmental consequences of bioenergy-driven land use changes.

The following chapters will discuss how the evaluation framework was applied to various land use scenarios to assess the impacts of bioenergy-driven land use change. Chapter 5 tests the overall effectiveness of the evaluation framework by applying it to retrospective land use change in a case study catchment. In Chapter 6, it was then applied to six bioenergy-driven land use change scenarios to assess their impacts of the land use changes.

Chapter 4: Case study region: the Burnett River catchment, Australia

4.1 Introduction

The Burnett River Catchment, Australia was selected as the case study region, as discussed in the first part of this chapter, to examine the effectiveness of the environmental framework in relation to past land use changes (Objective 3 and Chapter 5) and to evaluate the environmental consequences of several future bioenergy-driven land use change scenarios (Objective 4 and Chapter 6). The aims of this chapter are to explain and justify the selection of the case study region, and to briefly examine the past changes in climate, land cover, vegetation, and land use, which are critical for the analysis of the impacts of the past land use change (Chapter 5). Thus the first part of this chapter describes the selection criteria and processes of the case study region, and the last part mostly describes the background of the Burnett River catchment, itself. Since this catchment is one of the most ecologically diverse (marine, freshwater and terrestrial) regions in Australia, understanding the past and current environmental quality of the catchment, especially in relation to water quality and biodiversity, is necessary to enable better interpretation of the results (Chapter 5) and to address the policy implications of future land use changes in the catchment (Chapter 6). In addition, future climate projections and land capability studies related to agricultural production are essential datasets to develop future land use change scenarios incorporating bioenergy crop production in the catchment (Chapter 6).

4.2. Selection of the case study catchments

Case studies have been employed in a wide range of research areas from social science to medical research, as they allow the investigation of a small number of cases—often just one—in considerable depth. There is always some unit for case study research, such as number of samples and geographical area, and the data are collected and/or analysed in relation to case(s) (Hammersley & Gomm 2000). This methodology is expected to capture the complexity and the uniqueness of a single case or a small number of cases, and thus there is debate in the literature about whether case study research can be used as a basis for generalisation (Tellis 1997; Hammersley & Gomm 2000). Stake (1995) argued that the data generated by case studies would often resonate experientially with a broad cross section of readers, thereby facilitating a greater understanding of the phenomenon. Yin (2009) examined a number of case studies and identified at least four aims of these applications, which included:

- to explain complex causal links in real-life interventions;
- to describe the real-life context in which the intervention has occurred;
- to describe the intervention itself; and
- to explore those situations in which the intervention being evaluated has no clear set of outcomes.

In this research, a case study methodology was employed to explain the complex causal links of land use changes associated with bioenergy crop production, to describe the context in which land use changes occurred, and to explore the situation/effects of land use changes. More specifically, a case study region with a particular individual geographical unit was selected to collect data and test the evaluation framework (Objective 3), and to predict the potential environmental outcomes of several land use change scenarios which included high-yielding bioenergy crop production on underutilised agricultural lands (Objective 4).

Case study selection is very important to maximise what can be learned (Tellis 1997). In this research, the geographical unit for analysis was a catchment, which is an important hydrological, geomorphologic, and ecological spatial unit; for example, water quality and quantity indicators are most easily interpreted at a catchment scale (McBride et al. 2011), and change in the catchment hydrology and water quality could impact significantly on regional biodiversity. For these reasons, Australian regionalisation for the purposes of natural resource management is predominantly river basin-based (e.g. Natural Resource Management [NRM] bodies) (Lane, Taylor & Robinson 2009). Subsequently, selection criteria were established to identify a suitable case study catchment that sufficiently addressed both Objective 3 and Objective 4. The case study catchment had to:

- have experienced land use change and environmental pressures in the past as a result of agriculture expansion;
- have sufficient data available to analyse the region's land use history and the key environmental indicators included in the evaluation framework. This needed to include not only map data but monitoring and field survey data, in relation to the history of clearing, land use changes, water quantity, quality, and biodiversity over a range of years;

- offer a potential production base for a range of bioenergy feedstocks in the future, including areas with more restricted climate and soil conditions for non-food tropical crops/plants, and lignocellulosic crops;
- present a diverse range of land uses that could generate different land use change scenarios (except for urban or intensive land uses which generally prohibit conversion to crop production); and
- include land which met the definition of “underutilised agricultural land”.

The last three criteria were important to enable the generation of land use change scenarios incorporating high-yielding and non-food bioenergy crops that are resilient in marginal conditions (Objective 4 or Chapter 6). As a result, eight catchments from the Great Barrier Reef (GBR) region in Queensland, Australia were initially selected as candidate case study regions. This included the Burdekin River catchment (130,089km^2), the Herbert River catchment (9,850km^2), the Johnstone River catchment (2,326km^2), the Murray River catchment (1,107km^2), the Pioneer River catchment (1,572km^2), the Plane River catchment (2,536 km^2), the Burnett River catchment (33,257 km^2) and the Mary River catchment (9,433 km^2) (Figure 4.1).

There were a number of reasons that made these GRB catchments suitable for this application. Firstly, all catchments contained good examples of past agricultural land use and activities that threatened the high environmental values of the region. They are adjacent to the GBR lagoon, which is the world’s largest coral reef ecosystem extending over 2,000 kilometres and covering 348,000 km^2, and which has been listed as a World Heritage area since 1981. The reef contains a wide range of valuable marine ecosystems, and the water quality in the reef lagoon has declined due to the impact of vegetation clearing and agricultural activities and this had seriously threatened the reef’s marine ecosystems (Gilbert & Brodie 2001; McKergow et al. 2005; Haynes et al. 2007; Queensland Department of the Premier and Cabinet 2009b). Therefore, compared to other parts in Australia, there was a large body of available data from federal and state government research and monitoring programs (Queensland Department of the Premier and Cabinet 2009b) in relation to water, soil and biodiversity.

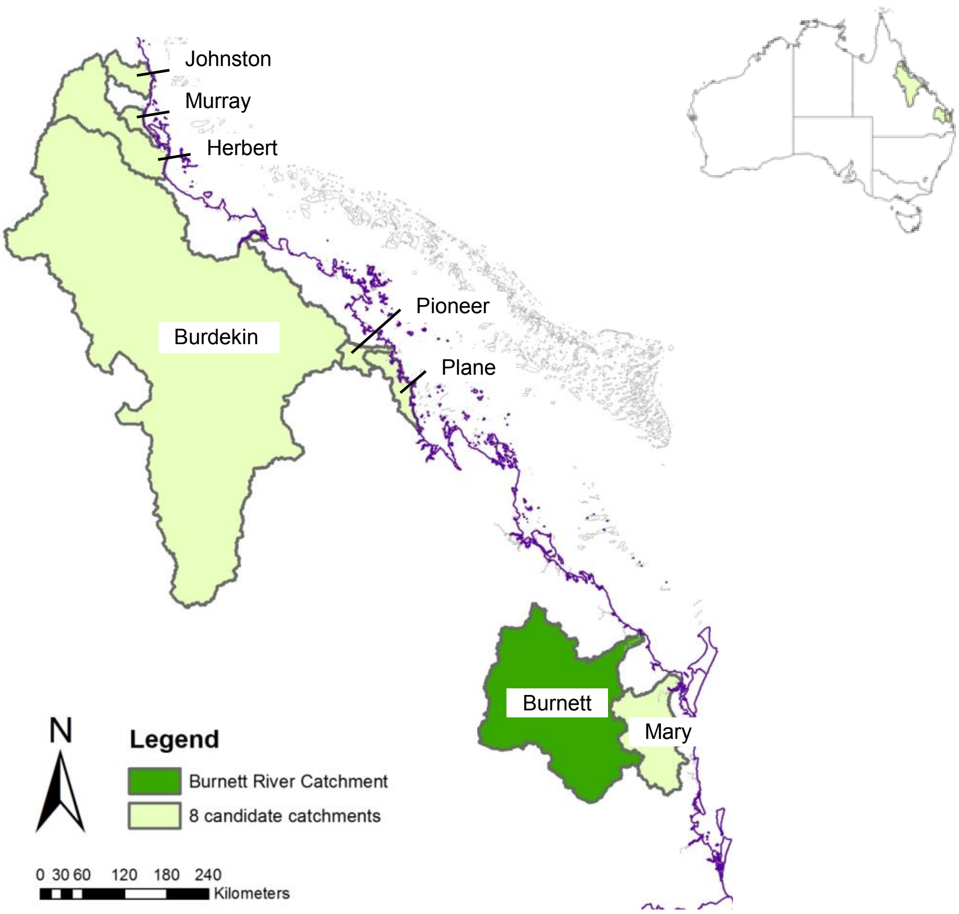


Figure 4.1 Location of eight candidate catchments in the Great Barrier Reef region, Australia.

In addition, this region could support the production of a range of tropical and subtropical high-yielding bioenergy crops such as sugarcane (*Saccharum officinarum*), agave (*Agave tequilana*) (Chambers & Holtum 2010), Pongamia (*Millettia pinnata*) (Murphy et al. 2012), sweet sorghum (*Sorghum bicolor L.*) (RIRDC 2013), and eucalypt species (Shepherd et al. 2011a,b). Sugarcane has been a key agricultural crop in the coastal areas of these catchments[11] and its residue, bagasse, has been one of the key renewable energy sources in Australia (Clean Energy Council 2008; ABARES 2011b). As part of the selection process, preliminary figures were prepared by overlaying gridded climate data – mean annual rainfall (Bureau of Meteorology 2009), mean maximum temperature in summer, and mean minimum temperature in winter (Bureau of Meteorology 2008) - with the basic climatic requirements of agave (D. Chambers 2011, pers. comm., 15 July), Pongamia (Professor P. Gresshoff 2011,

[11] Queensland produces 95% of the Australia's total sugarcane production (27.3 million tonnes in 2010) (Canegrowers, 2011).

pers. comm., 4 August), and sweet sorghum (Tienwong, Dasananda & Navanugraha 2009). This was undertaken to determine whether these catchments actually have production potential for these high-yielding bioenergy crops. The results indicated that these catchments would offer a potential production base for some of these crops, including Pongamia and sweet sorghum (Figure 4.2).

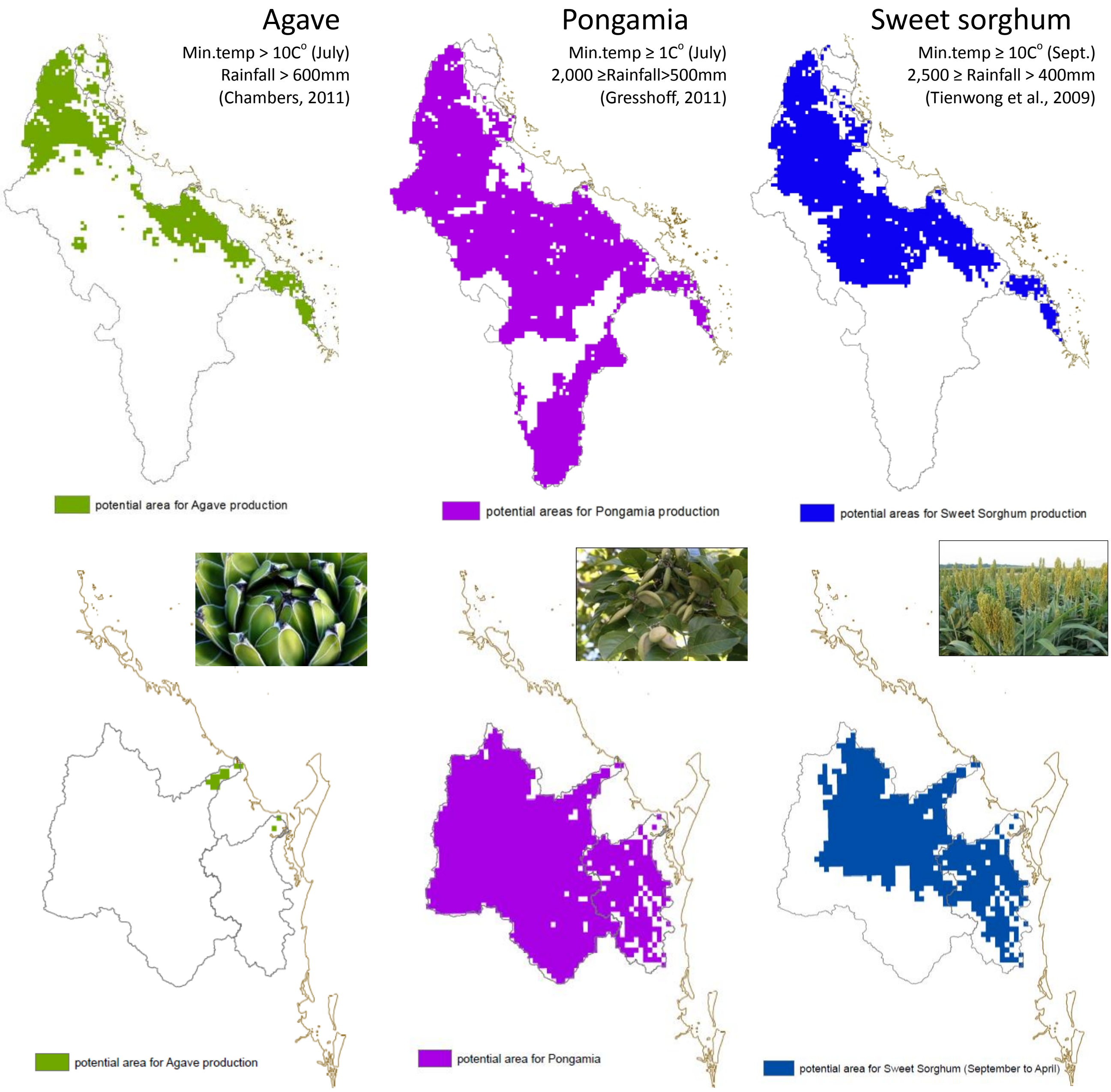


Figure 4.2 Preliminary suitability studies for potential bioenergy crop production, derived from past climate data (annual rainfall, max. temperature in summer, min. temperature in winter) (Source: generated from BOM gridded climate data).

To select the most appropriate sub-region from within the GBR region to use as the case study catchment, information about the eight candidate catchments was collected and analysed in relation to the selection criteria (Table 4.1). An expert panel was convened (16 November, 2011)[12] to evaluate each candidate catchment against the criteria, and to populate a decision matrix to facilitate decision making (Table 4.2). The aim was to select a catchment that could accommodate a wider variety of land use change scenarios in order to test the hypothesis in this research, by meeting the previously-noted case study selection criteria.

It was not straightforward to translate some of the criteria into quantitative values. For example, the criterion of 'having a potential production base for a range crops' was difficult to determine due to a number of factors that were not easily quantified. This criterion was mainly judged from a combination of climatic, geographical, biophysical and agricultural features of these catchments, which enabled the growth of a variety of potential crops and provided opportunity to develop various scenarios for later analysis. The panel members were familiar with these catchments from their own expertise and experience in the field, such as natural resource management, agriculture, ecology, and environmental assessment. Scores were assigned based on these discussions. The panel raised the following points with respect to the key criteria (Table 4.1) during the discussion:

- Accessibility to the case study region should be an important criterion. The panel considered that the long distances to north Queensland catchments would hinder the access to the case study region (e.g. travel costs). Further, north Queensland is more affected by flood and cyclone activity during the wet season (November to April), and this would have made it difficult to visit and conduct fieldwork.
- The Herbert catchment could be a good case study catchment, but there are limitations in terms of land availability and data availability in the upper catchment.
- Coastal catchments with favourable climate conditions for agricultural production (e.g. Plane, Pioneer, Johnston, and Murray catchments) should be eliminated from consideration, as the lands are in high demand for food production, and urban and

[12] The workshop was held on 16 November, 2011 at Room 320 Steele Building, the University of Queensland. The participants included Associate Professor Ann Peterson (natural resource management), Dr. Marguerite Renouf (LCA, sugarcane production), Professor Clive McAlpine (ecology) and Saori Miyake from the School of Geography, Planning and Environmental Management (GPEM), and Dr. Carl Smith (environmental systems and decision) and Dr. Malcolm Wegener (agriculture, agricultural economics) from the School of Agriculture and Food Science.

tourism developments.

The outcome was that the Mary and Burnett River catchments, located in the southernmost end of the GBR lagoon, were selected as the case study catchments for this research. These catchments satisfied most criteria equally well (Table 4.2), capturing a wide range of agricultural land uses and climate types, extending from the coastal strip to inland areas. This diversity provided a strong advantage for the selected catchments as it would allow greater diversity in relation to the selection of the land use change scenarios. More importantly, the availability of various environmental and land use data, and the ease of access to the region were also rated highly by the evaluation panel. As a result of the discussions in this workshop, and taking into account the time limits for data collection and analysis, one catchment, the Burnett River catchment, was selected as the case study region. In addition, the Burnett River catchment has the majority of its land in inland areas where there is less rainfall and a wider range of temperature, and this resulted in the availability of lands defined under 'underutilised agricultural land', which is an important theme in this research.

The coastal catchments (the Pioneer, and Plane catchments) and the Wet Tropic catchments (the Johnstone and Murray catchments) were regarded as not suitable for bioenergy feedstock production, due to more suitable climate conditions for agricultural activities and thus high demand for not only food production, but urban and tourism developments. The use of food production areas for future bioenergy crop production contradicted the basic premise of this research, that future bioenergy feedstock production should avoid replacing lands on which food production was conducted. In this regard, the Herbert and Burdekin River catchments were considered to be good candidates; however they lacked variety in land uses, and had limited availability of some data.

Table 4.1 Key criteria and information considered for final decision on case study catchment.

Criteria	Information collected and analysed
Accessibility to the catchment	• Distance and travel cost from Brisbane • Climate and topographic restrictions (e.g. wet season)
Past land use changes	• analysis of the past land use data, such as QLUMP (1999, 2004 & 2009) and/or Land Use of Australia (version 3 in 2006) • analysis of the past land-cover change maps for those in the 1990s, such as SLATS land-cover change maps (1991 & 1999), which indicate the sugarcane expansion during the period
Variety of land use classes	• analysis of the current land use data • area and distribution of each land use class on land use maps
Availability of environmental quality data	• availability of water quality monitoring data (sediment and nutrient loads and runoff volume) • availability of vegetation maps, such as Regional Ecosystem (RE) maps
Availability of parameters and data from past application of hydrological models	• past application, calibration and validation of the spatial hydrological model (*Source* ver. 3.2.3beta) to the catchments
A potential production base for a range of crops	• a combination of climatic, geographical, biophysical and agricultural features of candidate catchments, which enable them to grow a variety of potential crops and provide opportunity to develop various scenarios for the later analysis (mostly judged by the experience of the panel members) • initial suitability study of potential bioenergy crops (e.g. Agave, Sweet sorghum, Pongamia) in the catchment derived from overlaying various climate data (annual rainfall, max. temperature in summer, and min. temperature in winter) (Figure 4.2).
Availability of agricultural suitability studies and 'underutilised agricultural land' within the catchment	• availability of past agricultural land suitability studies and soil studies that identified agriculturally 'marginal lands' in the catchment
Potential of competition of lands with other land use (e.g. food crop production)	• location of the catchment (e.g. coastal/inland) • analysis of climate data (rainfall, max. temperature in summer and min. temperature in winter) • analysis of the current land use data, and the past land suitability studies and classifications • area of the catchment

Table 4.2 Discussion matrix for selecting the case study region against selection criteria (16 November, 2011).

Catchment	Accessibility of the region (Score:1-5)	Past land use change due to sugarcane expansion (%)	Availability of land use maps (Score: 1-3)*	Availability of environmental quality data (Y/N) water	vegetation	Past application of hydrological modelling (Y/N)	Potential production base for different crops (Score: 1-5)*	Diversity of land use classes (Score: 1-5)*	Availability of land suitability/ capability (Score: 1-5)*	Competition of lands with other land use (Score: 1-5)*
Burnett	5	3%	3	Y	Y	Y	5	3	5	4
Mary	5	18%	3	Y	Y	Y	5	4	4	2
Burdekin	1	6%	3	Y	Y	Y	5	1	1	5
Herbert	1	13%	2 (no 2004 data)	Y	Y	Y	5	3	3	3
Johnstone	1	2%	3	Y	Y	Y	1	5	4	2
Murray	1	65%	3	Y	Y	Y	1	2	1	1
Pioneer	1	1.5%	3	Y	Y	Y	5	3	5	1
Plane	1	14%	3	Y	Y	Y	5	4	5	1

*On these scoring scales, 1 = low, 3 = medium, and 5 = high quality against these criteria.

4.3 The Burnett River catchment

The following sections provide background, current and projected status of the Burnett River catchment in relation to climate, geographic, land cover and land use characteristics.

4.3.1. Area features

The Burnett River catchment covers a total area of 33,257 km^2 with a population of approximately 60,000 (Great Barrier Reef Marine Park Authority 2009). This is one of the southernmost catchments of the GBR lagoon, and includes Bundaberg (population: 97,800 in 2011) (OESR 2012), and the towns of Kingaroy in the South Burnett (population: 13,900 in 2009) (ABS 2010c), Mundubbera (population 2,200 in 2009) (ABS 2010b) and Gayndah (population 1,790 in 2011) (Qpzm 2013) in the Central Burnett, and Monto in the North Burnett (population: 2,500 in 2011) (ABS 2010a). Bundaberg is the largest urban centre in the catchment, and is located in the coastal area (Figure 4.3).

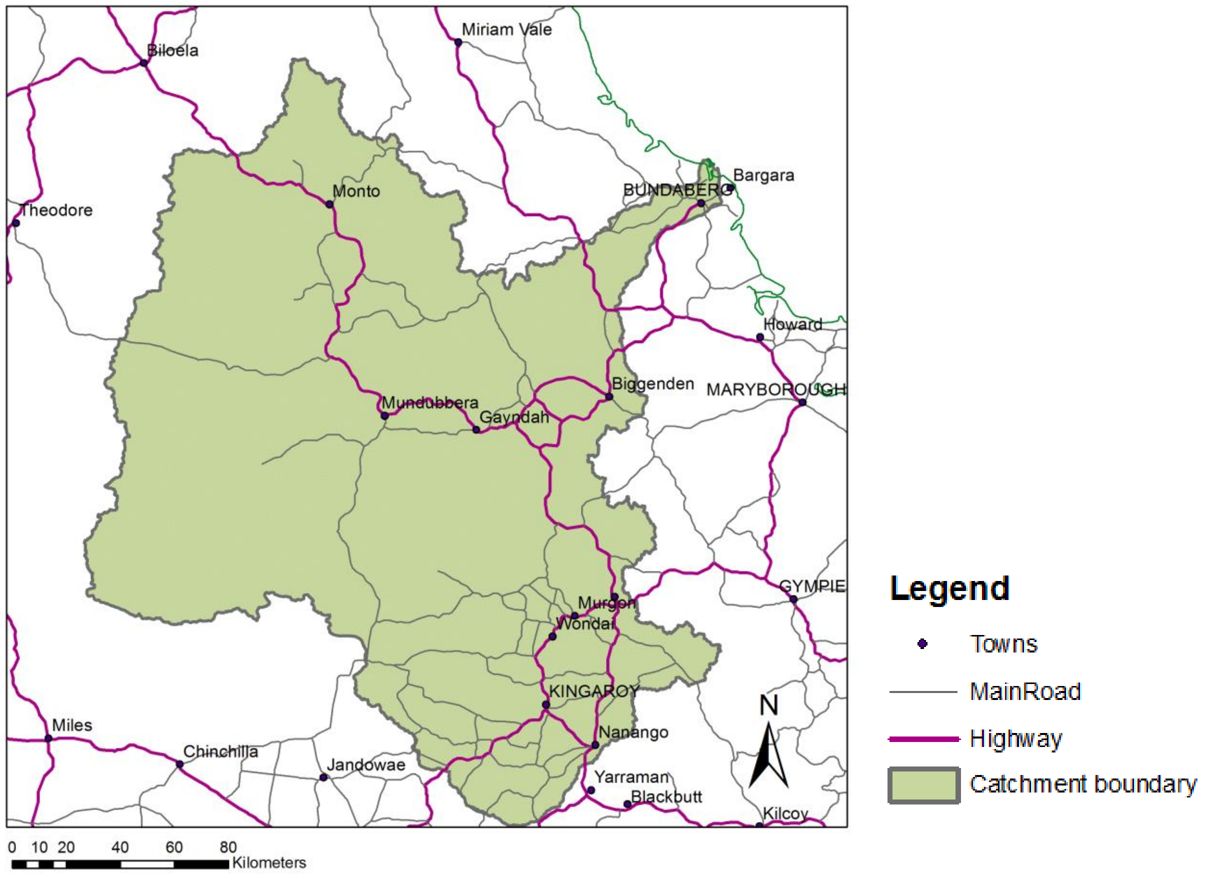


Figure 4.3 The Burnett River catchment, Australia.

4.3.2. Current and future climate

The Burnett River catchment has a sub-tropical climate, with long hot summers and mild winters. Rainfall is concentrated between December and February when cyclones cross the Queensland coast. This can cause major flood events, such as the one that occurred in January 2013 due to intensive rains from ex-tropical cyclone Oswald (Burnett Mary Regional Group 2013). However, rainfall can vary considerably from year to year, and droughts also occur regularly (Vandersee & Kent 1983). The coastal area and the rest of the inland areas of the catchment have slightly different climatic characteristics. The coastal area around Bundaberg is more humid (<1,200mm/year), but the inland areas are mostly semi-arid with rainfall below 800mm per year (Figure 4.4). Bundaberg has a higher rainfall and also a lower range of temperature than inland locations. The mean minimum temperature is above 10^{o}C in July in Bundaberg, but below 5^{o}C in inland areas (Figure 4.4). Frost generally occurs in low lying parts of the inland areas during the winter months, although the average date of the first and the last frost vary depending on other conditions, such as topography (Vandersee & Kent 1983).

Due to the changing climatic conditions expected as a result of climate change, there are predicted to be increases in temperatures, more severe-weather events and higher evaporation rates within the catchment. The climate model projections from the statistical Climgen method (Mitchell & Osborn 2005) were obtained from the CCAFS GCM downscaled Data Portal (CIAT & CCAFS 2012). In this climate change projection, the datasets on annual rainfall, maximum (December) and minimum (July) temperatures in the 2050s were generated using CSIRO Mk3.0 model and IPPC SRES A1B emission scenarios (Figure 4.5). Comparing these projected climate maps for the 2050s with the current climate datasets (1950-2000) obtained from WorldClim (Hijmans, Cameron & Parra 2005), annual rainfall is estimated to reduce within a range of 109-245mm by the 2050s in the Burnett River catchment, while maximum (December) and minimum (July) temperatures are expected to increase within a range of 0.4-5.3^{o}C and 0.7-4.8^{o}C respectively.

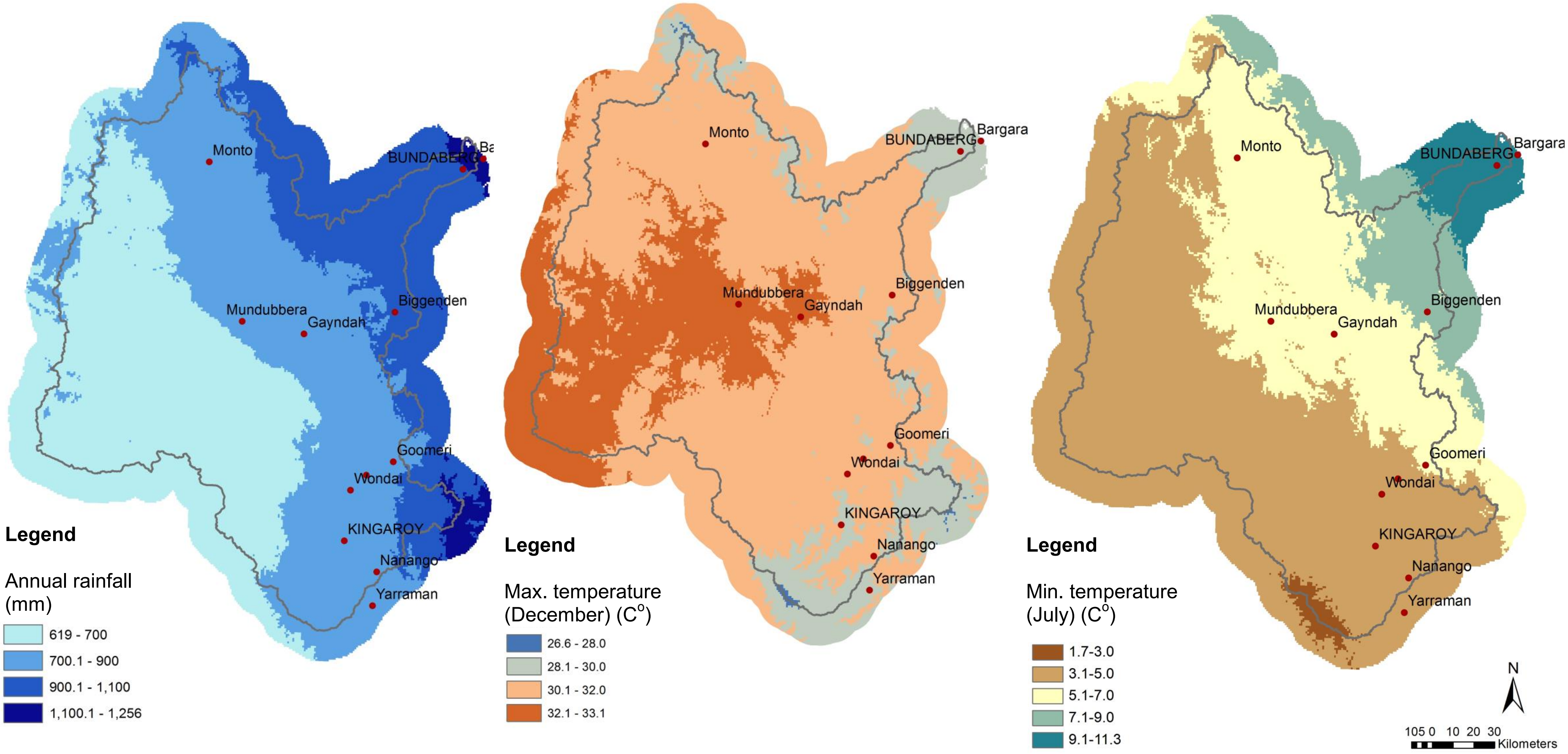


Figure 4.4 Average annual rainfall, maximum (December) and minimum (July) temperatures (1950-2000) across the Burnett River catchment (Source: generated from WorldClim dataset).

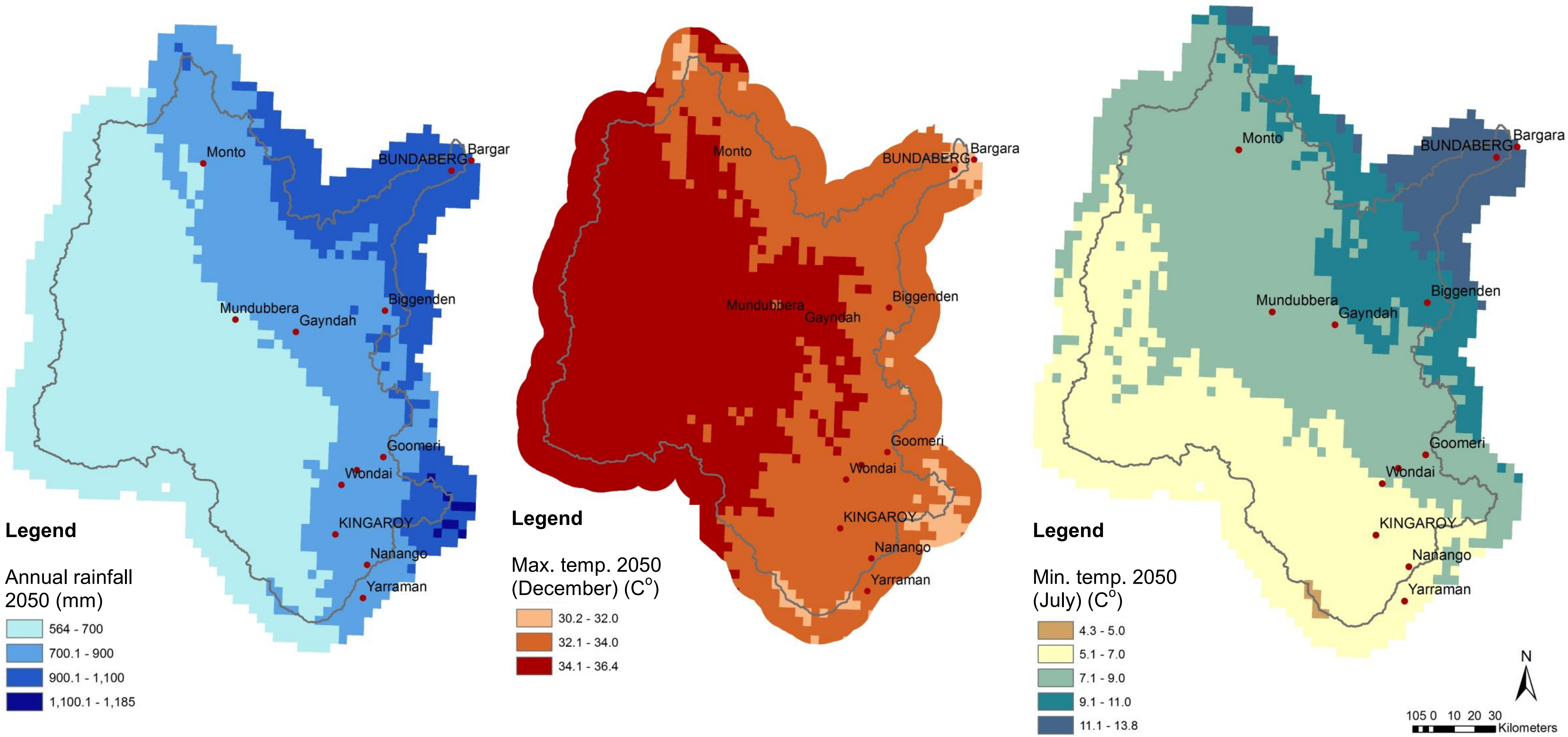


Figure 4.5 Projected annual rainfall, and maximum (December) and minimum (July) temperatures in the 2050s across the Burnett River catchment (Source: generated from the CCAFS GCM downscaled datasets).

4.3.3 Land use change

Brief information is provided in this section on current land uses in the Burnett River catchment. This includes historical development of the region, land cover and native vegetation changes, current land uses at catchment and sub-catchment scales, land capability, and information on water and biodiversity.

4.3.3.1 Historical development of the Burnett region (1840-1999)

The land use and land cover change in the Burnett River catchment is highly related to the history of European settlement since the early 1840s in the South Burnett (Matthews 1997a) and the late 1840s in the Central and Upper Burnett (O'Connor 1948). Although aboriginal people had lived in Australia for many thousands of years prior to European settlement, they changed the landscape to a limited extent due to their traditional way of life. On the other hand, European settlement resulted in extensive and rapid changes in the landscape as a result of land clearing and transformation for settlement, wood production, and pastoral production for wool, dairy and beef products (Matthews 1997a; O'Connor 1948; Seabrook, McAlpine & Fensham 2006). This involved extensive attempts to make the landscape conform to European visions (Bonyhady 2002). In particular, the intensification of pastoral development in the region occurred from the 1950s due to strong political drives for expansion of agricultural exports and infrastructure development, and as a result of technological advances, such as the use of heavy machinery, blade ploughs, and herbicides (Seabrook, McAlpine & Fensham 2006). The peak of broad scale clearing in the Burnett River catchment occurred between the 1960s and 1980s (I. Crosthwaite 2012, pers. comm.) and continued until environmental concerns emerged in the Australian society in the 1990s and the first vegetation management legislation, the *Vegetation Management Act 1999* (amended in 2004, 2009 and 2013) was introduced in Queensland to restrict broad-scale clearing of remnant (structurally intact) native vegetation (Queensland Government 2009b).

From the analysis of the Southern Brigalow region, Seabrook, McAlpine and Fensham (2006) identified five interacting drivers for regional-scale landscape change: (i) population; (ii) economy; (iii) policy; (vi) science/technology; and (v) cultural values. The Burnett region is located adjacent and to the north of the Southern Brigalow region, and featured similar historical development (Matthews 1997a, 1997b; O'Connor 1948). Although the above five drivers acted in synergy throughout the period, economic drivers together with population growth (immigration) and government policy (e.g. land settlement policy) played the most important role throughout the 1840s-1990s in the South Brigalow, because of the export-driven characteristics of the Australian agricultural industry supplying produce to Europe initially and then countries such as Japan and the U.S.A. (Seabrook, McAlpine & Fensham 2006). The importance of biophysical properties, such as soil type, topography, vegetation and climatic conditions, were also identified as the most obvious constraints to agricultural development and regional-scale landscape change (Seabrook, McAlpine & Fensham 2006).

4.3.3.2 Land cover change and vegetation (1999-2009)

From 1995, a vegetation monitoring program using satellite imagry was implemented by the Queensland Government under the Statewide Landcover and Trees Study (SLATS) [13]. By the late 2000s, approximately 71% of the Burnett River catchment had been cleared since European settlement (Great Barrier Reef Marine Park Authority 2009). The 2010 SLATS report (DERM 2010) confirmed a slower rate of remnant vegetation clearing in the Burnett-Mary NRM region[14] in the early 2000s due to the introduction of *Vegetation Management Act 1999*, and the general decrease in vegetation clearing in recent years (Figure 4.6). Nevertheless, over 50,000 ha of remnant woody vegetation in the NRM region was cleared after 1999 (DERM 2010).

[13] Statewide Landcover and Trees Study (SLATS) is a state-wide monitoring program of woody vegetation cover since 1988 under the *Vegetation Management Act* 1999 (amended in 2004 and 2009). The program aims to gather accurate wooded vegetation cover and woody land cover change information using satellite imagery for vegetation management planning and compliance, and for their GHG inventory purposes (DSITIA, 2012c).

[14] The Burnett-Mary Natural Resource Management (NRM) region (56,000 km^2) covers all lands drained by the Mary, Kolan, Burnett, Auburn, Boyne, Elliot, Gregory, Isis and Burrum rivers and their tributaries. It encompasses the World Heritage-listed Great Sandy Straits including Fraser Island, and the southern tip of the Great Barrier Reef Marine Park. The population of the region is estimated at 290 000, the majority concentrated on the coastal fringes. The population is expanding rapidly overall and is estimated to reach 350 000 by 2026. Bundaberg (in the Burnett River catchment), Gympie and Maryborough (in the Mary River catchment) are the major urban centres.

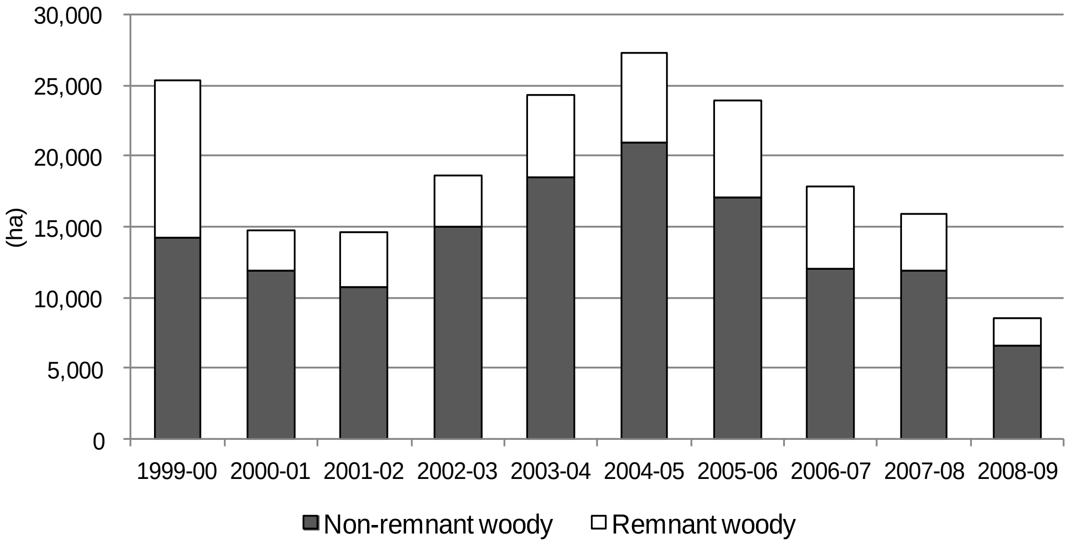


Figure 4.6 Woody vegetation clearing in the BMRG NRM region by remnant status (Source: DERM 2010).

Clearing of woody vegetation occurred mostly on non-remnant vegetation on freehold tenure land at a rate of 8,556 hectares per year (ha/year) in 2008-09. The majority of this woody vegetation in the region was cleared for pastures and plantation forestry, while clearing for cropping significantly reduced after 2001-02 (Figure 4.7) (DERM 2010).

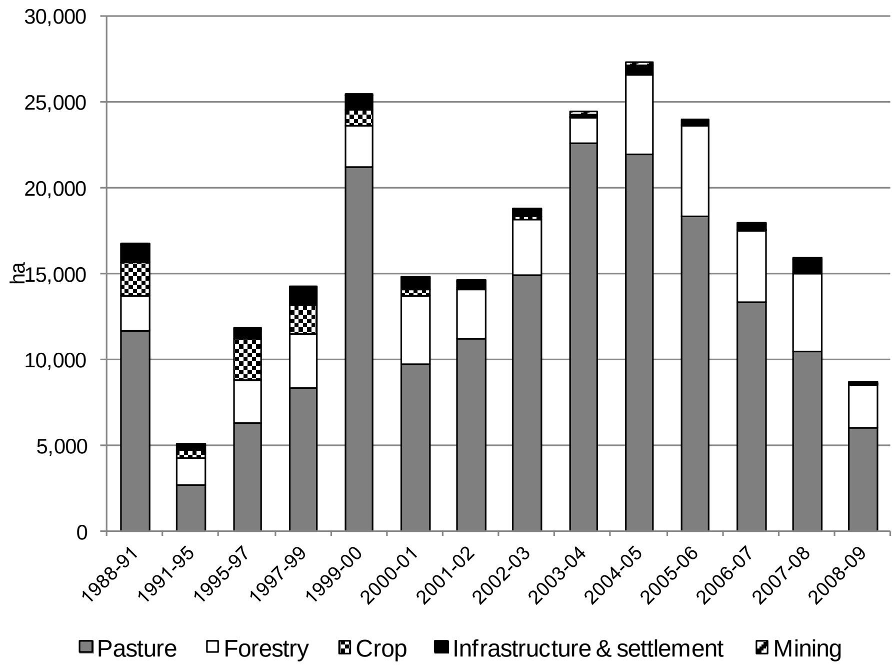


Figure 4.7 Woody vegetation clearing rates in the BMRG NRM region by replacement land cover (Source: DERM 2010).

The loss of wetland and riparian vegetation is a significant issue for the catchment, particularly along the Burnett River, where 45% of the riparian vegetation was lost due mainly to intensive agriculture and altered hydrology, and 52% of the natural shoreline was modified by the mid- 2000s (Moss, Scheltinga & Tilden 2008). Between 2001 and 2005, the loss of wetlands in the BMRG NRM region was 180 ha (0.36 %), while the loss of riparian vegetation was 9,185 ha (1.04%). This was the highest proportion among the GBR regions (Burnett Mary Regional Group n.d.).

4.3.3.3 Current land use on a catchment scale (1991-2005)

There are two land use datasets for the Burnett River catchment. The oldest dates from 1992/93 and was produced by the Australian Government (Land Use of Australia, *version 3*) (1:2,500,000) (ABARES & BRS 2006). Finer scale datasets at the catchment and/or state level (1:50,000) became available after 1999 from the Queensland Land Use Mapping Program (QLUMP) (DSITIA 2012g). Although both datasets map patterns of land use in accordance with the Australian Land Use and Management classification (ALUM), there are differences in the methodologies and the data resolution between the two affects the outputs (ABARES 2011c). Both datasets are generated from available spatial information, satellite imagery and aerial photographs. However, QLUPM conducts field observation for verification and validation (ABARES 2011a), while ABARES employs a modelling approach to incorporate the national agricultural census data into the spatial information (ABARES 2011c). In regard to the Burnett River catchment, the different methodologies resulted in a general overestimate of 'conservation' and 'forestry' classes in the ABARES datasets (1992/93, 2001/02 and 2005/06) compared to QLUMP (1999 and 2009) (Figure 4.8). In particular, 'grazing' land ('livestock grazing' [2.1.0] by ALUM) on QLUMP datasets was often classified into 'conservation' by the ABARES datasets ('other minimal use' [1.3.0][15]) (J. Mewett 2013, pers. comm. [e-mail], 9 August). This is because differentiating these two land classes is difficult and often subjective, especially when the land is rarely used for the specified prime land use (i.e. grazing in this context) (C. Shephard 2011, pers. comm. [e-mail], 8 August).

As a result, both datasets were used in parallel in this research as they complemented each

[15] The definition of 'other minimal use' of ALUM is "land that is largely unused (in the context of the prime use) but may have ancillary uses" (ABARES, 2011b).

other. The most recent QLUMP dataset (2009), released in May 2012, was used to describe the current land use of the Burnett River catchment (Figure 4.9) (DSITIA 2012g). However, the ABARES datasets were used as base maps for the application of the evaluation framework to past land use change (Chapter 5) and future scenarios (Chapter 6). The main reason for this was that the 2009 land use map had not been released when the evaluation framework was applied to past land use changes in the case study catchment (Chapter 5) in 2011. More importantly, the ABARES datasets provide the oldest land use dataset (1992/93) and this allowed for an evaluation of environmental changes in the case study catchment before and after the *Vegetation Management Act 1999*.

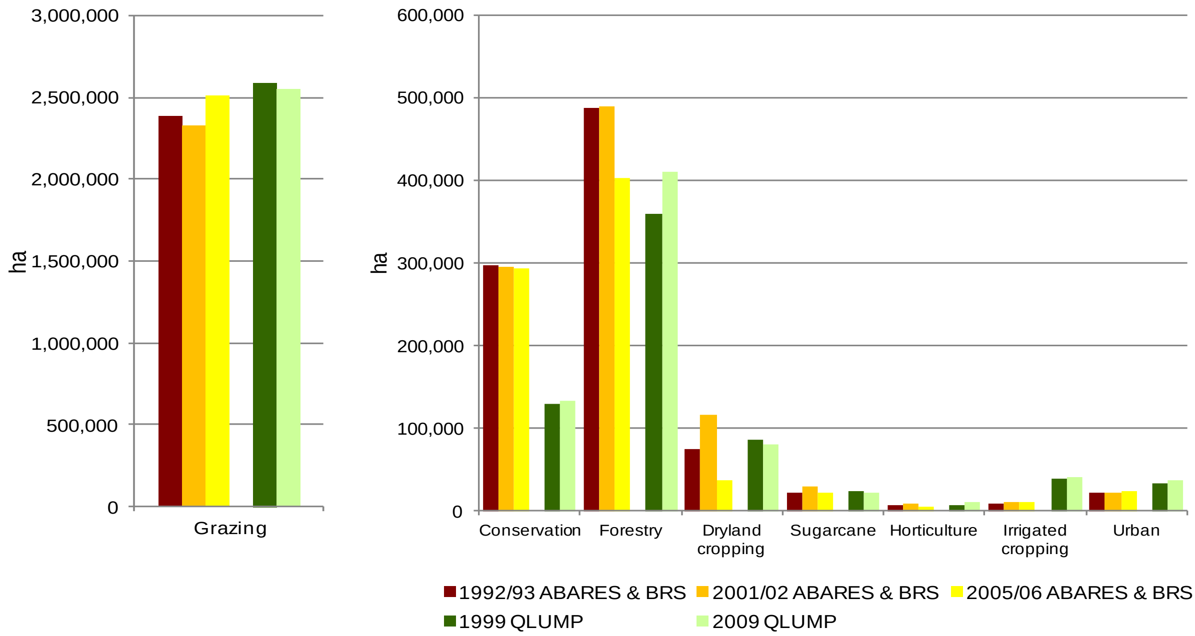


Figure 4.8 Current land use in the Burnett River catchment by comparison of the national scale (ABARES & BRS) and the Queensland (QLUMP) datasets.

* Land use classification by ALUM was consolidated into eight main land classes in the catchment (for the reclassification rule, see Appendix Table A5.1).

The latest 2009 QLUMP dataset indicated that approximately 95% of the land in the catchment was used for agricultural production. Livestock (beef cattle) grazing was the predominant agricultural land use (25,535km^2) accounting for 76.8% of the catchment, followed by cropping (1,220km^2) (3.7 %). Sugarcane (215km^2) (0.6%) and horticulture (102km^2) (0.3%) were the main agricultural land uses in the coastal areas. Grazing in this catchment has been undertaken more on modified pasture, while native pasture has declined. Plantation and production forestry was also significant in the catchment (4,100km^2) (12.3%). However, conservation areas only covered 1,128km^2 or 3.3 % of the total catchment.

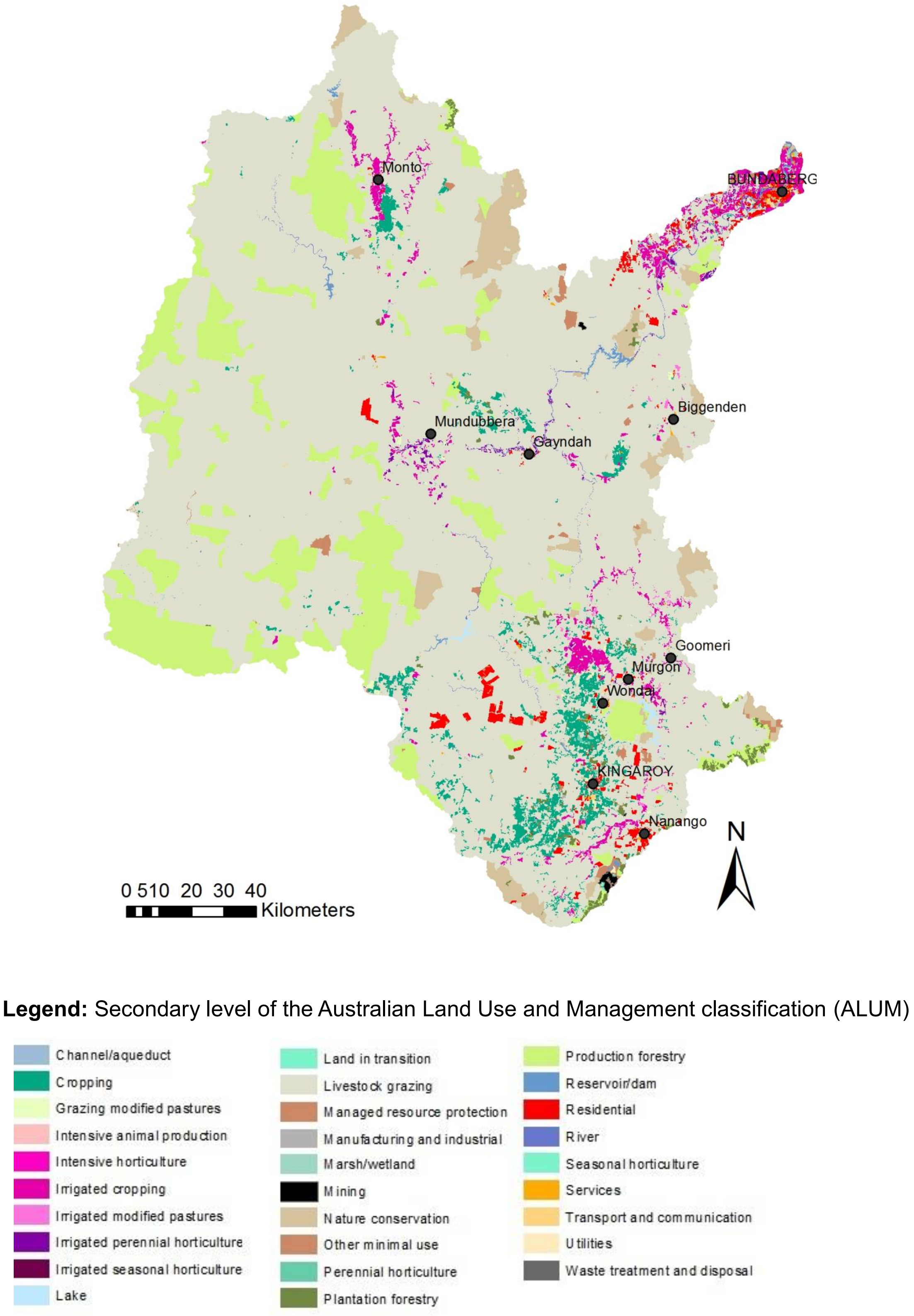


Figure 4.9 2009 Land use in the Burnett River catchment (Source: DSITIA 2012g).

4.3.3.4 Land use on a sub-catchment scale (1991-2005)

The sub-catchments within the Burnett River catchment demonstrated various land use change patterns (Appendix Table A5.5). In particular, the coastal sub-catchment (B1) was distinct from several inland South Burnett sub-catchments (B4, B5, B7 and B8) and others. These differences were presented based on the land use datasets (Figure 4.10. and 4.11) together with the information gained from interviews with local agronomists in Bundaberg and Kingaroy. Interviews were conducted with Mr. Robert Doyle (Burnett Mary Field Officer, Growcom Australia) and Mr. Neil Halpin (Agronomist, Queensland Government) on 17 February, 2012 in Bundaberg, Mr. Ian Crosthwaite (Manager, Agronomy Services, BGA AgriServices) and Mr. Damien O'Sullivan (Senior Extension Officer, Primary Industries & Fisheries, Queensland Government) on 12 and 13 March, 2012 respectively, in Kingaroy (Appendix 3). The main objectives of those interviews were: to obtain regional specific data; and information and to confirm the information retrieved from the datasets.

Coastal area

In the coastal area around Bundaberg, land use is more diverse compared to inland sub-catchments. Grazing accounts for around 80% of the total area (Figure 4.10), although sugarcane production is a traditional industry that occurs in higher rainfall coastal areas within the catchment. The Bundaberg Irrigation Area (36,000ha) was established in 1970 and completed in the 1980s, primarily to respond to the growing water demand from sugar producers along the Kolan and Burnett Rivers. Sugarcane lands have decreased since the late 1990s (Figure 4.10) due to a number of economic, social and environmental challenges caused by greater competition in global sugar markets (Walker, Vella & Kotzman 2005). This also has been driven by the biophysical characteristics of the Queensland coastal regions, which are highly valued for a variety of other land uses, such as hobby farms, horticulture, and urban development driven by industry and tourism activities (Wegerner 1997; Walker, Vella & Kotzman 2005).

In Bundaberg, sugarcane farmers have shifted their focus to other activities, especially horticulture. A local agronomist explained that tree crops—mostly macadamia nuts and avocados—had replaced sugarcane and its fallow lands (about 20% of total sugarcane land) because of their higher gross profit margins (I. Doyle 2012, pers. comm., 17 February). Although the ABARES and BRS datasets did not capture this trend, the 2009 QLMP dataset identified a loss of 1,950 hectare (ha) (9.3%) in 'sugarcane' areas in 1999-2009, and a significant increase in 'horticulture' (1,646 ha, 65.8%) (Figure 4.10). In fact, 1,380 ha of 'sugarcane' land was converted to 'horticulture' use in 1999-2009 (DSITIA 2012g). Moreover, the Queensland Department of Agriculture, Fisheries and Forestry (former Department of Employment, Economic Development and Innovation [DEEDI]) estimated that the total area of horticulture in the Bundaberg Irrigation Area doubled between 1993 (7,000 ha) and 2009 (16,440 ha) (I. Doyle 2012, pers. comm., 17 February).

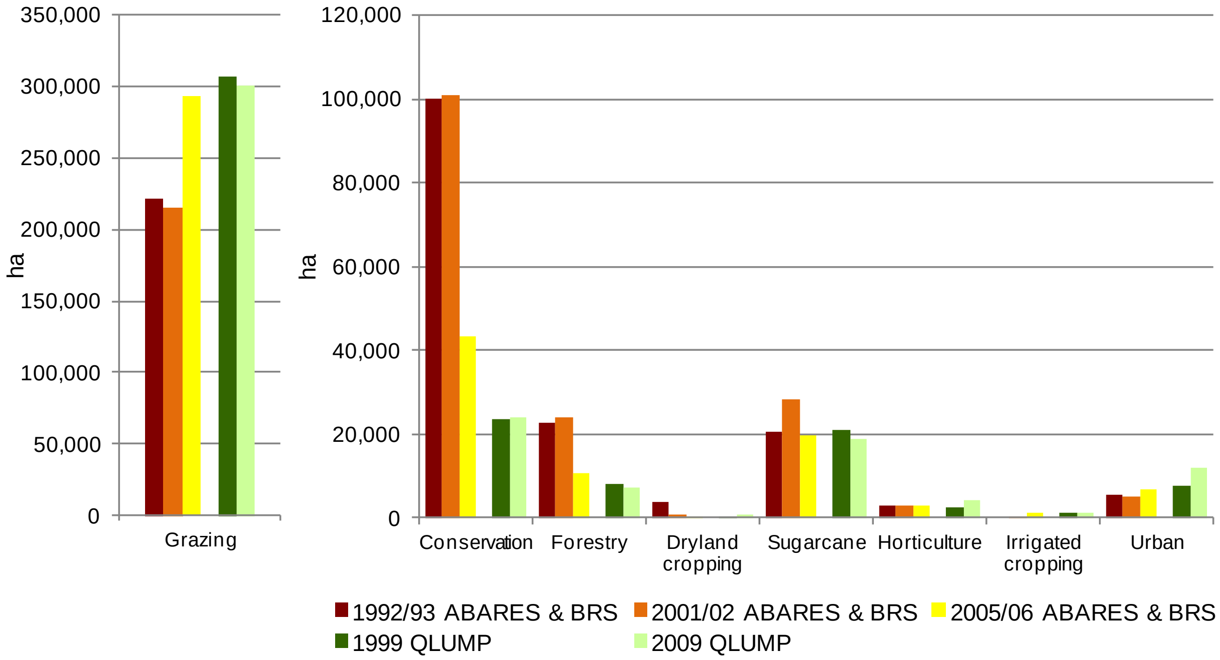


Figure 4.10 Current land use in the coastal sub-catchment B1 by comparison of the national scale (ABARES & BRS) and the Queensland (QLUMP) datasets.

*Land use classification by ALUM was consolidated into eight main land classes in the catchment (for the reclassification rule, see Appendix Table A5.1).

Urbanisation is also a particular phenomenon of the coastal area. The QLUMP dataset showed the rapid expansion of residential developments in the coastal sub-catchment B1 - 43.9% of residential areas in 2009 were newly developed during the past decade. Between 1999 and 2009, around 5,200 ha of land was converted to 'residential' purposes in B1, of which 4,630 ha (88.8%) came from 'grazing' area and around 200 ha (4.1%) from 'sugarcane' (DSITIA 2012g).

Inland area (example of the South Burnett region)

In the South Burnett region around Kingaroy, major clearing for settlements and pastoral production for dairy and grazing (Matthews 1997b) occurred prior to the 1960s, although the dairy industry has almost disappeared over the past two decades (I. Crosthwaite 2012, pers. comm., 12 March). Grazing is the predominant land use in the South Burnett region, accounting for around 80% of the total land use (Figure 4.11). However, areas surrounding Kingaroy also present a diverse agricultural land use pattern, including peanut production and a variety of feed, food and pharmaceutical crops, including cereal, sorghum, corn, mung beans, hay, and alfalfa.

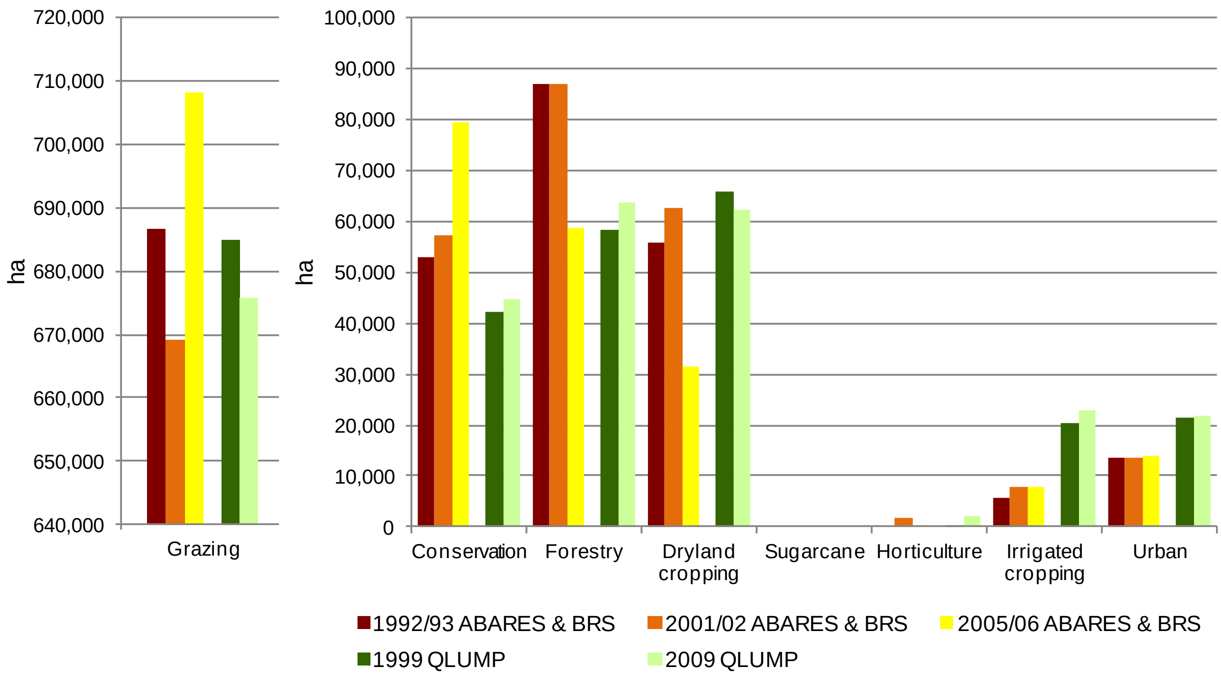


Figure 4.11 Current land use in the South Burnett sub-catchments B3, B4, B7 and B8 by comparison of the national scale (ABARES & BRS) and the Queensland (QLUMP) datasets.

*Land use classification by ALUM was consolidated into eight main land classes in the catchment (for the reclassification rule, see Appendix Table A5.1).

In the South Burnett, plantation and production forestry, and dryland cropping are the two major land uses after grazing. While the two different land use datasets presented opposite trends in land use change in those categories (Figure 4.11), the land use patterns on the ABARES and BRS land use maps (2006) were supported by local agronomists in this case (I.

Crosthwaite 2012, pers. comm., 12 March; D. O'Sullivan 2012, pers. comm., 13 March). Firstly, the dataset illustrated that the dryland cropping area had significantly decreased between 1992/93 and 2005/06, while grazing areas continuously expanded over these years. The abandonment and/or conversion of large areas for dryland cropping to beef cattle grazing over the last few decades was confirmed by local agronomists (Plate 4.1). A combination of various economic and social factors may have contributed to this change, such as agricultural population decline in the region, high risk of poor economic returns from cropping compared to cattle grazing, diseconomies of scale in crop production for many farmers in the region, and improved cattle prices in international markets (I. Crosthwaite 2012, pers. comm., 12 March; D. O'Sullivan 2012, pers. comm., 13 March). Moreover, a series of poor seasons in recent years caused by droughts and wet harvest seasons influenced farmers' decisions to leave dryland cropping for other activities (I. Crosthwaite 2012, pers. comm., 20 August; D. O'Sullivan 2012, pers. comm., 22 August).

Plate 4.1 Abandoned cropland converted to cattle grazing (near Kingaroy) (12 March, 2012).

The ABARES and BRS land use dataset also showed a decrease in land under plantation and production forestry after 2001/02 (Figure 4.11). This can be explained partly by Managed Investment Schemes (MIS)[16] and a number of plantation forests newly established in the early 2000s on good agricultural lands (I. Crosthwaite 2012, pers. comm., 12 March; D.

[16] As a response to the timber shortage in Australia, the federal government introduced Management Investment Schemes (MIS) under the Managed Investments Act in 1998. It allowed investors to take a personal income tax deduction for investing in reforestation and agribusiness activities through the scheme. The sector grew quickly and reached a peak in 2006/07. However, many funds management companies faced financial debt and the sector declined very quickly.

O'Sullivan 2012, pers. comm., 13 March). These plantations produced various native and exotic species, such as Princess tree (*Paulownias tomentose*), Blue gum (*Eucalyptus globulus*), Spotted gum (e.g. *Corymbia citriodora subsp. Variegate [CVV]*), and Chinchilla white gum (*Eucalyptus argophloia*). However, many plantations were abandoned due to the financial bankruptcies of several of these managed investment companies (Plate 4.2).

Plate 4.2 Abandoned plantation forest established under Managed Investment Schemes (near Kingaroy) (12 March, 2012).

4.3.4 Land capability

The Queensland Government has conducted regional land resource studies addressing climate, geology, landform, drainage, vegetation and soils since the 1980s. The boundaries of these studies varied depending on the purposes and interests, such as riparian lands along the Gayndah-Perry River (McCarroll & Brough 2000) and the Bundaberg Irrigation Area (Donnollan 1998) within the catchment. The outputs also varied depending on the purposes of studies, such as land suitability, land capability and land resource mapping.

In regard to the inland Burnett region, three studies were conducted to map the land capability classes for agricultural purposes—for South Burnett (Vandersee & Kent 1983), Central Burnett (Kent 2002), and North Burnett (Donnollan & Searle 1999). These studies cover 86.5% of the catchment area, and were used as a basis for identification of 'underutilised agricultural land' in this research (Chapter 6). They were based on the land capability classification of Rosser et al. (1974), which divided lands into eight classes (from

Class I to VIII) based on the degree of limitation for agricultural purposes (Appendix Table A4.1). Class I refers to the land most suited for all agricultural and pastoral uses without limitations, while Class VIII is land with severe limitations and unsuited for either cropping or grazing. Lands under Class I to IV are considered to be good agricultural lands suitable for cropping. On the other hand, Class V to VIII lands have more limitations. The thirteen limitations included water availability, soil depth, narrow moisture range, rockiness, salinity, microrelief, wetness, flooding, water erosion and topography (Donnollan & Searle 1999). A scale was given to each land unit for each limitation based on the degree of severity, and the capability was usually determined by the most severe limitation (Donnollan & Searle 1999). These studies not only addressed the selection of lands for agricultural activities, but also considered the requirements for appropriate land resource management, particularly for lands in Class V and above. In the three studies, no land in these studies was classified as Class I due to the incidence of occasional overflow flooding (Vandersee & Kent 1983; Donnollan & Searle 1999; Kent 2002). Approximately 72% of the land in the study area was classed as not suitable for cropping (i.e. greater than Class V), while only 28 % of land was classed as suitable for cropping and other agricultural use (i.e. Class II to Class IV) (Figure 4.12 and 4.13).

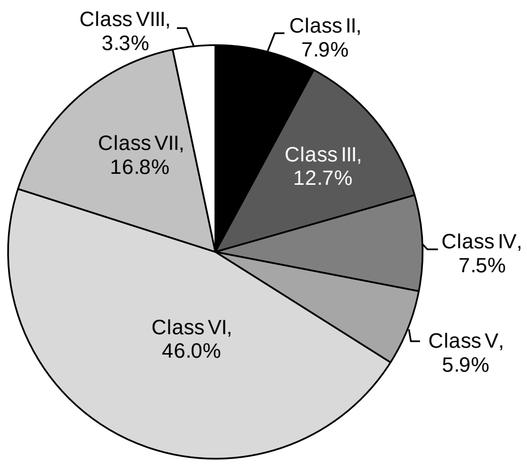


Figure 4.12 Percentage of total area of each land capability class (Source: generated by data from Donnollan and Searle, 1999; Kent, 2002; Vandersee and Kent, 1983).

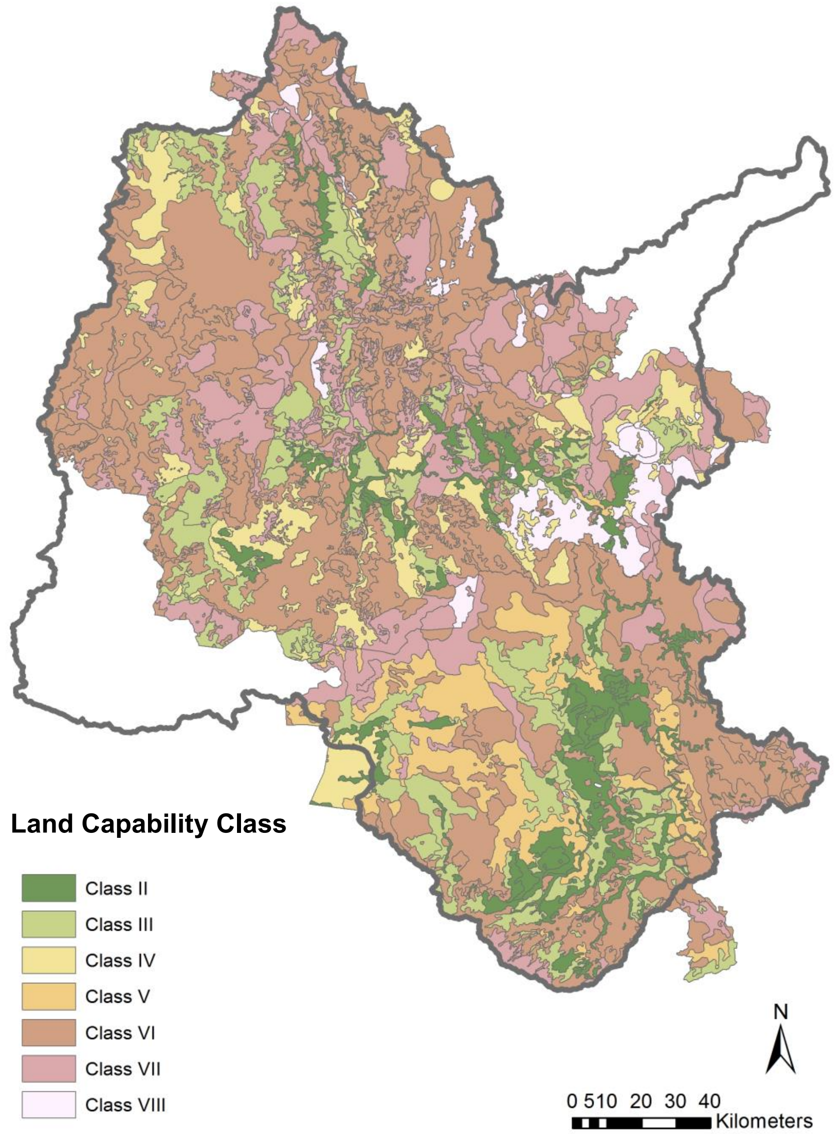


Figure 4.13 Land capability map of the Burnett region (Source: generated from data by Donnollan and Searle 1999; Kent 2002; Vandersee and Kent 1983).

4.3.5 Water quality

The Burnett River catchment is one of the 35 GBR catchments whose waterways directly flow into the GBR lagoon. The degradation of the richest and most diverse natural ecosystems in the lagoon has been linked to increases in land-based runoff of suspended solids, nutrients and pesticides since European settlement (Wachenfeld, Oliver & Morrissey 1998; Brodie & Mitchell 2005; Queensland Department of the Premier and Cabinet 2009b). Kroon et al. (in press) reported that the river loads to the GBR lagoon have increased by 2.9 to 6.8 times for suspended solids, by 2.5 to 4.5 times for total nitrogen, and by 3.9 to 6.4 times for total phosphorous since the pre-European period.

Water quality has also been a challenge in the Burnett River catchment. Although discharges of pollutants to the estuary had reduced greatly since the 1980s, (Moss, Scheltinga & Tilden 2008) estimated that compared to pre-European settlement, sediment, total nitrogen (TN) and total phosphorous (TP) loads in the Burnett River had increased 32 times, 2.5 times and 6.9 times respectively. Compared to the average of the GBR catchments (Kroon et al. in press), this represented a significant increase in the sediment load, with the Burnett River estuary rated as subject to an 'extreme' level of risk of impacts due to human activity, and with 'poor' estuary health (Moss, Scheltinga & Tilden 2008). In particular, the aquatic sediments stressor was rated as 'very poor', and the nutrients stressor as 'poor' (Moss, Scheltinga & Tilden 2008) due mainly to the intensive land uses (e.g. sugarcane, urban areas) adjacent to the estuary. The proportion of cleared land was high, there were only limited areas of undisturbed riparian vegetation, and treated effluent from sewage treatment plants (particularly phosphorous load) in Bundaberg was also a factor.

In response, the Australian and Queensland governments launched initiatives to protect the lagoon. A five year program, Reef Rescue was established in 2008 as a key component of a $200 million initiative of the Australian government's Caring for our Country program. The majority of the funds were allocated to the Water Quality Incentive Grants, which support regional farmers and six NRM groups in the GBR catchments, to increase the adoption of improved land management practices through collaborative partnerships (Queensland Department of the Premier and Cabinet 2009b). The BMRG is the coordinating body in the catchment for Reef Rescue programs that have supported graziers, dairy farmers, sugarcane and horticultural farmers.

The Reef Water Quality Protection Plan (Reef Plan) is a joint Australian and Queensland government initiative. Reef Plan was first introduced in 2003 for the GBR catchments and updated in 2009 through the Reef Rescue package (Queensland Department of the Premier and Cabinet 2009b). In Reef Plan 2009, measurable water quality and land management practice targets were established for the GBR catchments. Land management practice targets are also important components for Reef Plan, while the specific water quality targets stipulate that:

- by 2013 there will be a minimum 50% reduction in nitrogen, phosphorous and pesticide loads at the end of catchments; and

- by 2020 there will be a minimum 20% reduction in total suspended solids (TSS) loads at the end of catchments (Queensland Department of the Premier and Cabinet 2009b).

The Paddock to Reef Integrated Monitoring, Modelling and Reporting Program (Paddock to Reef) was developed as a key action program to measure and report on the progress towards the Reef Plan targets through collaborations between governments, agricultural industries, research organisations, and regional NRM groups (Queensland Department of the Premier and Cabinet 2009a). Paddock to Reef consists of monitoring and modelling activities (Queensland Department of the Premier and Cabinet 2009a), one of which is the GBR Catchment Load Monitoring Program that estimates sediments, nutrient and pesticide loads and yields based on monitoring data. Data from the 2009-10 monitoring (Table 4.3) indicated that the Burnett River catchment was a relatively low contributor of TSS, TN or TP exports compared to larger GBR catchments (e.g. the Fitzroy and Burdekin catchments). In terms of the yields of sediment and nutrients, the catchment also had a lower contribution compared to smaller coastal catchments with high annual rainfall and a high proportion of irrigated cropping land use such as horticulture and sugarcane (e.g. the Pioneer, Johnstone and Plane catchments) (DSITIA 2012e).

Table 4.3 Estimated TSS, TN and TP loads and yields for 2009-2010 in the Burnett River catchment (Source: DSITIA 2012e)

	Estimated total load (tonne)	**Estimated yield (kilogram per km^2)**
TSS	146,732	4,500
TN	1,262	38.4
TP	181	5.5

Note: The data comes from the monitoring sites at Ben Anderson Barrage Head Water of the Burnett River, which covers 99% of the catchment.

As the monitoring program began in 2008-09, there is only a short data record. In response, extensive water quality modelling is undertaken as part of the Paddock to Reef program. Australia has a highly variable climate and there is considerable annual variation in rainfall, and therefore it is critical to understand the long-term average annual pollutant loads for the GBR catchments (D. Waters 2013, pers. comm., 7 February). The eWater CRC *Source Catchments* modelling framework is used to generate the estimated loads of sediment, nutrients and pesticides (Waters & Carroll 2013). The modelling and monitoring efforts go hand in hand, so that the measured data from the GBR Catchment Load Monitoring Program is integrated into the framework to validate the model results. The baseline loads for the Burnett River (2008/09) were established to report the progress towards the Reef Plan water quality targets each year. After a year, it is reported that the Burnett River catchment has been making progress towards the reduction of sediment and nutrients loads (Waters & Carroll 2013) (Table 4.4).

Table 4.4 The baseline loads (2008/09) of TSS, TP and TN, and load reductions after year 1 investment (2009/10) at the Burnett River catchment (Source: D. Waters 2013; pers. com., 1 February; Waters & Carroll 2013).

	TSS (kilo tonne per year)	**TP (tonne per year)**	**TN (tonne per year)**
2008/09 (baseline)	20.19	57.15	297.31
2009/10 (estimate)	19.80	55.10	271.78
Reduction	2.3%	5.7%	11.9%
Progress status	Moderate progress	Good progress	Good progress

The spatial hydrological model, *Source*, was developed for this large scale application to the GBR studies. As this model is also adopted for the application to the case study catchment in this research, the strengths and limitations of this model have been outlined (Chapter 3). One of the limitations of the model's estimates, identified from the Paddock to Reef experience, related to the limited availability of measured data to validate the model's results (D. Waters 2013, pers. comm., 7 February). Nevertheless, the accuracy of the model is expected to be continuously improved in the long term in line with improved availability of measured data over a long period of time under the continuous monitoring program and/or other breakthrough technologies, such as satellite imagery (D. Waters 2013, pers. comm., 7 February).

Lastly, BMRG's Better Catchments initiative funded by the Australian Government's Caring for our Country program aims to improve overall catchment health. A range of projects have been funded, including reducing erosion and promoting groundcover, coastal and estuarine water quality monitoring activities, and biodiversity conservation activities (e.g. pest plant and animal control, improving wetland health, enhancing and/or rehabilitating riparian and terrestrial vegetation, and improving connectivity between high priority riparian and terrestrial areas).

4.3.6 Terrestrial biodiversity

The catchment is located in one of the most ecologically diverse regions in Australia, spanning two biogeographic regions: the Southeast Queensland Bioregion in the east, and the Brigalow Belt Bioregion in the west. In particular, the Brigalow Belt Bioregion is one of 15 recognised National Biodiversity Hotspots in Australia (Australian Government 2003) due to its climate, variety of soils, vegetation and landforms (e.g. mountains, lowlands, coastal). The Burnett region contains the State's highest number of priority species, and such threatened ecological communities include brigalow (*Acacia harpophylla*) and semi-evergreen vine thicket (Wide Bay Burnett Environment and Natural Resources Working Group 2012). Before European settlement, the Brigalow Belt supported vast native vegetation communities dominated by brigalow. However, the broad-scale land clearing schemes initiated in the 1960s resulted in removal of approximately 60% of the original vegetation of the bioregion, predominantly for agriculture (Wilson, Neldner & Accad 2002; Eyre et al. 2009), mostly modified pastures (McAlpine et al. 2009). Much of the retained native vegetation in the catchment is contiguous dry eucalypt forests and woodlands at higher elevation (>300m) (Eyre et al. 2009). The 2005 Regional Ecosystem map (version 6.0b) indicated that dry eucalypt forests/woodlands contain approximately 93 % of the total native vegetation in the Burnett River catchment (Figure 4.14).

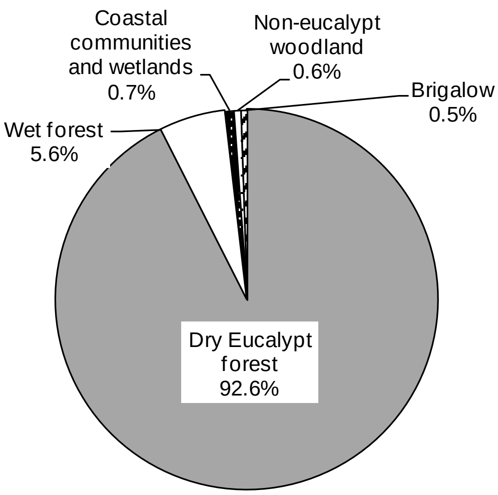


Figure 4.14 Composition of native vegetation communities in the Burnett River catchment 2005 (Source: derived from 2005 Regional Ecosystem map [version 6.0b]).

Despite the significant loss of its original vegetation, the catchment contains important habitats for endangered, rare and threatened species (Table 4.5), and the remaining intact vegetation provides conservation benefits for these species. However, the majority of the eucalypt forests in the Brigalow Belt including the Burnett region are used for cattle grazing and selective timber harvesting, and consequent altered fire regimes have encouraged weed and feral animal invasion (McAlpine et al. 2009), and put at risk small woodland and grassland birds (Australian Government 2003; Martin et al. 2006; Maron & Kennedy 2007; Eyre et al. 2009).

Table 4.5 Main species in Burnett Mary NRM region Queensland, Australia (Source: created from Australian Government 2011; Wide Bay Burnett Environment and Natural Resources Working Group 2012)

Vertebrates	**Common name (Scientific name)**	**EPBC status***
Mammals	Bridled nailtail wallaby (*Onychogalea fraenata*)	Endangered
	Brush-tailed rock-wallaby (*Petrogale penicillata*)	Vulnerable
	Northern quoll (*Dasyurus hallucatus*)	Endangered
Frogs	Wallum-sedge Frog (*Litoria olongburensis*)	Vulnerable
	Giant barred frog (*Mixophyes iteratus*)	Endangered
	Fleay's Barred-frog (*Mixophyes fleayi*)	Endangered
Birds	Eastern Bristlebird (*Dasyornis brachypterus*)	Endangered
	Regent Honeyeater (*Xanthomyza phrygia*)	Endangered
	Red Goshawk (*Erythrotriorchis radiatus*)	Vulnerable
	Black-breasted Button-quail (*Turnix melanogaster*)	Vulnerable
	Glossy Black-Cockatoo (*Calyptorhynchus lathami*)	
	Double-eyed Fig-parrot (*Cyclopsitta diophthalma*)	
	Grey Goshawk (*Accipiter novaehollandiae*)	
Reptiles	Nangur Spiny Skink (*Nangura spinosa*)	Critically endangered
	Yakka Skink (*Egernia rugosa*)	Vulnerable
	Brigalow Scaly-foot (*Paradelma orientalis*)	Vulnerable
	Dunmalls Snake (*Furina dunmalli*)	Vulnerable
	Golden-tailed Gecko (*Strophurus taenicauda*)	
Bats	Semon's Leaf-nosed Bat (Hipposideros semoni)	Endangered
	Grey-headed flying-fox (Pteropus poliocephalus)	Vulnerable
	Greater Long-eared Bat (*Nyctophilus timoriensis*)	
	Eastern Long-eared Bat (*Nyctophilus bifax*)	

* Under *the Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act), 446 threatened species and ecological communities are currently listed using six categories: 'extinct', 'extinct in wild', 'critically endangered', 'vulnerable', and 'conservation dependent'.

In response, several initiatives have been undertaken to improve biodiversity outcomes on a regional scale. Healthy Habitat, implemented by BMRG, is funded through the Australian Government's Caring for our Country program, and focuses on three areas: on-ground investment in terrestrial and coastal biodiversity projects; biodiversity education and awareness programs (terrestrial, marine and freshwater); and monitoring and research activities for priority species. Past on-ground investments have included shorebird conservation (BMRG 2008), feral pig control in the Central Burnett (Burnett Catchment Care Association Inc. 2012a), protection of endangered semi-evergreen vine thicket ecosystems of the Brigalow Belt (North and South) (Burnett Catchment Care Association Inc. 2012c), and revegetation of native vegetation such as Brigalow, Belah (*Casuarina cristata*), and bottletree species (*Brachychiton spp*.) in the North Burnett (Burnett Catchment Care Association Inc. 2012b). Monitoring and research activities have included development of a species and ecosystem monitoring database, and monitoring and protecting six endangered and vulnerable marine turtles such as loggerhead turtle [*Caretta caretta*], and Indo-Pacific humpback dolphins along the coastal areas of the catchment (BMRG 2008).

4.4 Summary

The Burnett River catchment was selected as the case study catchment based on the availability of key data (e.g. land use and vegetation maps, land capability studies, and water quality data); the accessibility of the region; the land use diversity in the catchment; and its potential for the production of various bioenergy crops. The catchment is located at the southernmost end of the GBR lagoon, the world's largest and most diverse coral ecosystem, and spans the Brigalow Belt Bioregion, one of 15 recognised National Biodiversity Hotspots.

The Burnett River catchment is an agricultural catchment, with 71% of the total catchment consisting of cleared lands. The majority of broad-scale land clearing occurred by the 1980s, mostly for settlements and agriculture, but the clearing rate has significantly reduced since 2000 due to the introduction of the *Vegetation Management Act 1999*. The coastal strip around Bundaberg has higher rainfall and a lower temperature range than the inland areas. This has resulted in higher demand for the coastal areas for various human activities—traditionally sugarcane production and more recently urban development. However, cattle grazing is the dominant land use (almost 80%) of the catchment especially in inland areas, replacing dryland cropping.

The catchment is one of the most ecologically diverse (marine, freshwater and terrestrial) regions in Australia. However, the pre-European environmental qualities were significantly degraded by large-scale vegetation clearing and poor land management practices. The past land clearing threatened important habitats for threatened or rare species and ecological communities, and the degraded water quality caused by agricultural runoff from the adjacent catchments has been a major threat to the GBR lagoon. The BMRG has been implementing a range of initiatives with stakeholders (e.g. federal and state governments, relevant industries) through a collaborative partnership to address issues related to declines in water quality and biodiversity.

Chapter 5: Application of the environmental evaluation framework in the Burnett River catchment, Australia

5.1 Introduction

The framework for evaluating the environmental consequences of bioenergy-driven land use changes at the regional scale (Chapter 3) was applied to spatial land use change data in the Burnett River catchment in Queensland, Australia (Chapter 4). The purpose was to test the overall effectiveness of the framework and the appropriateness of the selected indicators, models and tools, before the framework was applied to a range of potential bioenergy-driven land use change scenarios.

The first part of this chapter describes the methods and input data relevant to the application of the environmental framework to the Burnett River catchment. This is followed by an analysis of results obtained from the model simulations and a synthesis of the water and biodiversity effects of the land use changes using radar charts. Lastly, the strengths and limitations of the evaluation framework are discussed against several criteria.

5.2 Methods and input data

5.2.1 Local and regional scales

Catchments are an important spatial unit for hydrological analysis. The evaluation framework (Chapter 3) was applied at two spatial scales—local and regional—to land use change data available for 1992/93, 2001/02 and 2005/06 in the Burnett River catchment. In this research, the regional scale refers to a catchment, while the local scale refers to a sub-catchment. Supplementary sub-catchment scale analysis was undertaken for a closer examination of land use change and its effects at a more detailed scale. Arc GIS 10 was used to map the distribution of estimated results and changes over the period for sub-catchment scale analysis.

For this purpose, the catchment (33,257 km^2) was divided into 18 sub-catchments, with the boundaries determined by the spatial hydrological model, *Source* (ver. 3.2.3 beta), based on stream topography and a 100 metre resolution surface topography Digital Elevation Model (DEM) for the catchment (eWater CRC 2010) (Figure 5.1). *Source* can generate sub-catchment boundaries, stream networks (links) and nodes according to a user-specified minimum drainage area (stream threshold) and flow gauging station positions. The minimum drainage area of 1,000km^2 was used to delineate the sub-catchment boundaries. This enabled the spatial variability in loads due to land use changes scenarios to be demonstrated clearly.

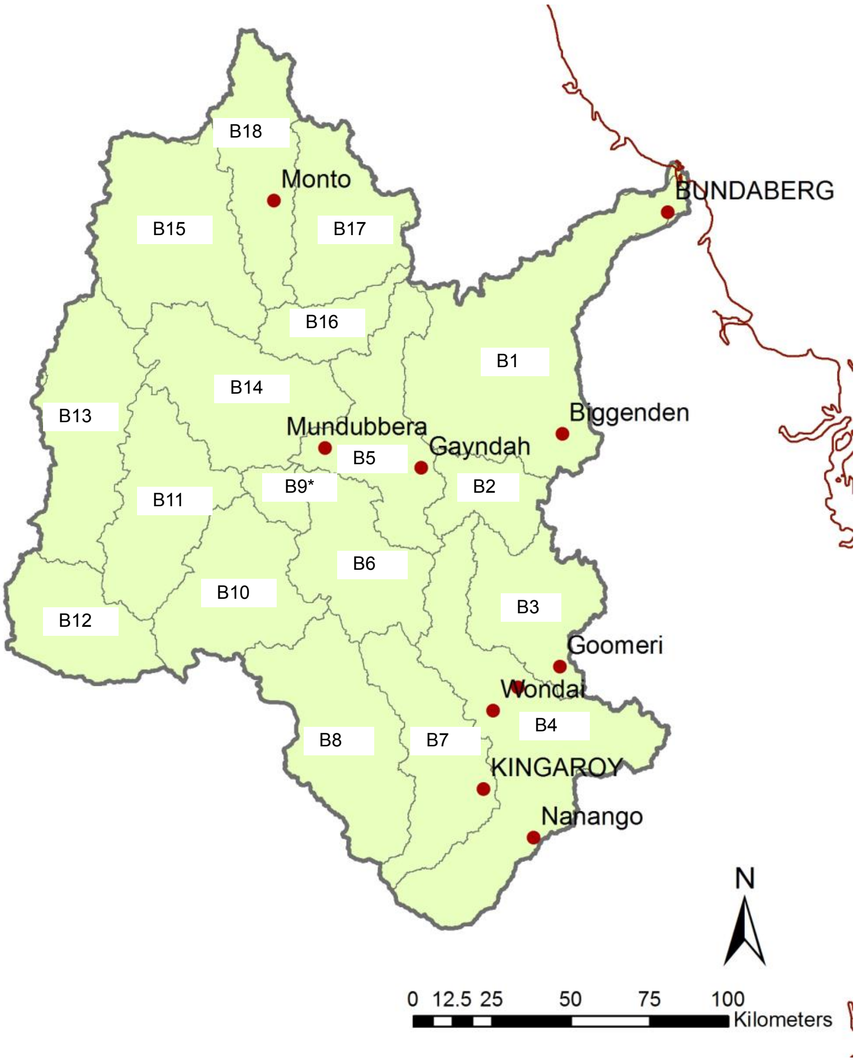


Figure 5.1 Sub-catchments within the Burnett River catchment, Australia.

* B9 is composed of two very small sub-catchments—B9 (1) and B9 (2). For hydrological analysis using *Source*, the two sub-catchments were separately analysed, while for biodiversity analysis using *Fragstats*, they were analysed as one sub-catchment.

5.2.2 Land use maps

The most important input data for the evaluation framework were the land use maps. These were prepared for three time periods, 1992/1993, 2001/2002 and 2005/2006 to enable comparison of the changes over this period. The 1992/93 map is the oldest digital land use map available for this catchment. The maps used were national scale land use maps from *Land Use of Australia, version 3* (ABARES & BRS 2006) and *version 4* (Bureau of Rural Science 2010). This dataset was limited by its coarse resolution (pixel size of 1.1 km^2) and scale (1:2,500,000), and its model-based methodology, which lacked verification and validation through field observation (ABARES 2011a). Finer catchment scale land use maps (1:25,000) generated by the Queensland Government (QLUMP) would have been more appropriate for this application. However, that dataset was available for only two time periods (1999 and 2009). In addition, the 2009 land use map was released in May 2012, after the start of this application in 2011. Thus the national scale land use maps for 1992/93, 2001/02 and 2005/06 were used in this application.

5.2.3 Model development and calibration

A simple catchment model was developed using the *Source* (ver. 3.2.3beta) platform to simulate four water quantity and quality indicators: (i) runoff volume; (ii) total suspended solids (TSS); (iii) total nitrogen (TN); and (iv) total phosphorous (TP). The model was purposely designed to achieve the aim and scope of the evaluation framework. Hydrologists from the Queensland Government provided technical advice in building and implementing the model, and expert judgement in relation to data selection and process, based on their extensive hydrological modelling experience in all Great Barrier Reef (GBR) catchments under the Paddock to Reef Integrated Monitoring, Modelling and Reporting Program (Paddock to Reef) (DSITIA 2012e). Under this program, the hydrologists developed a more advanced model on the *Source* platform, and this enables the simulation of changes related to land use and also a range of land management options – in particular investment in improved agricultural practices (see results for the estimates from the program in Table 4.4, Chapter 4) (Waters & Carroll 2013).

Model calibration and validation are critical processes in hydrological modelling. It is generally an iterative process, which requires substantial amounts of time, expertise and resources for gathering sufficient historic meteorological data and observed data and reviewing parameters to bring predictions closer to observations (eWater CRC 2011). Under the Paddock to Reef program, parameters for two important components of hydrological modelling within the *Source* platform (i.e. rainfall-runoff model and constituent model) were identified by Queensland hydrologists based on careful examination of observed water modelling data (Fentie in press). This advanced model for the Burnet Mary catchments was used as a base calibrated model for the model purposely developed in this evaluation (Fentie in press). This ensured efficiency in terms of the amount of time and resources involved in the general model calibration and validation processes. Nevertheless, adjustments and adaptations were necessary in relation to the calibrated parameters for the catchment model due to the differences in the two models, such as catchment and sub-catchment boundaries. These adaptations were undertaken in consultation with the Queensland hydrologists.

5.2.4 Input data for *Source*

Function Unit (FU) maps

Source required input data about topography and land use (or land management practices), in particular a digital elevation model (DEM), and Function Unit (FU) maps and their classification. FU maps can be generated from information on land use types (e.g. agriculture, urban, forest), soil types (e.g. sandy soils, clay soils) or vegetation types (e.g. forest, pasture, bare soil). For this application, three FU maps for 1992/93, 2001/02 and 2005/06 were generated from land use classification on national land use maps in grids and in the same geographic projection (the Albers projection) (Figure 5.2). The land use classifications in the original land use maps (Australian Land Use and Management Classification (ALUM)) were consolidated to reflect the 11 main land use classes in the catchment: (1) ‘conservation’; (2) ‘forestry’; (3) ‘grazing forested’; (4) ‘grazing open’; (5) ‘dryland cropping’; (6) ‘sugarcane’; (7) ‘horticulture’; (8) ‘irrigated cropping’; (9) ‘urban’; (10) ‘water’ and (11) ‘other’. This reclassification aligned with the land use classification previously used by Fentie (in press) for the Burnett River catchment (Appendix Table A5.1). For the 2001/02 and 2005/06 land use maps, (3) ‘grazing forested’ and (4) ‘grazing open’ land use classes were distinguished using the threshold of 12% Foliage Projective Cover (FPC). FPC was obtained from the Queensland Government’s Statewide Landcover and Trees Study (SLATS) program (DERM 2011; DSITIA 2012b), which presents the level of woody vegetation coverage, obtained from Landsat TM imagery. The 12% FPC definition approximates to 20% crown cover, which has been commonly used as the threshold for vegetation clearing in remote sensing studies

(DNRM 2003). The method for estimating FPC within the SLATS data set changed after 1999. Therefore, to distinguish 'grazing forested' and 'open grazing' for the 1992/93 FU map, FPC 1999 (DSITIA 2012a) and another SLATS product that indicates the location and number of woody vegetation clearing events between 1991 and 1999 were used (DSITIA 2012c).

Rainfall-runoff model, constituent model and parameters

Source enables the user to build a catchment model by selecting the most suitable combination of a rainfall-runoff model and a constituent (pollutant) generation model, depending on data availability and the model complexity required by the users (eWater CRC 2011). For this application, the Simhyd rainfall runoff model and the Event Mean Concentration/Dry Weather Concentration (EMC/DWC) constituent generation model were chosen as the preferred models within *Source*.

Simhyd is one of the most commonly used conceptual rainfall-runoff models available in *Source*. It has been tested extensively across Australian catchments (Chiew & Siriwardena 2005). It requires climate data such as daily rainfall and potential evapotranspiration (PET) data as inputs, and seven parameters for different FU classes (eWater CRC 2010). Daily rainfall and PET data were obtained from SILO climate data, and were converted into grid format for input into *Source*. For this application, daily rainfall and PET data between 1 January 1970 and 31 December 2009 were entered into the model (DSITIA 2012f). The Simhyd parameters (Appendix Table A5.2) for all the Burnett River sub-catchments were adopted from the base calibrated model of the region developed by Fentie (in press). Three sets of Simhyd parameters were applied including one for forested areas, a second for cleared grazing areas and a third for cropped areas. The rationale for grouping functional units into three was based on an assumption that they had three distinct rainfall runoff responses with forest areas being the lowest followed by cleared grazing land and then cropping areas (Fentie in press).

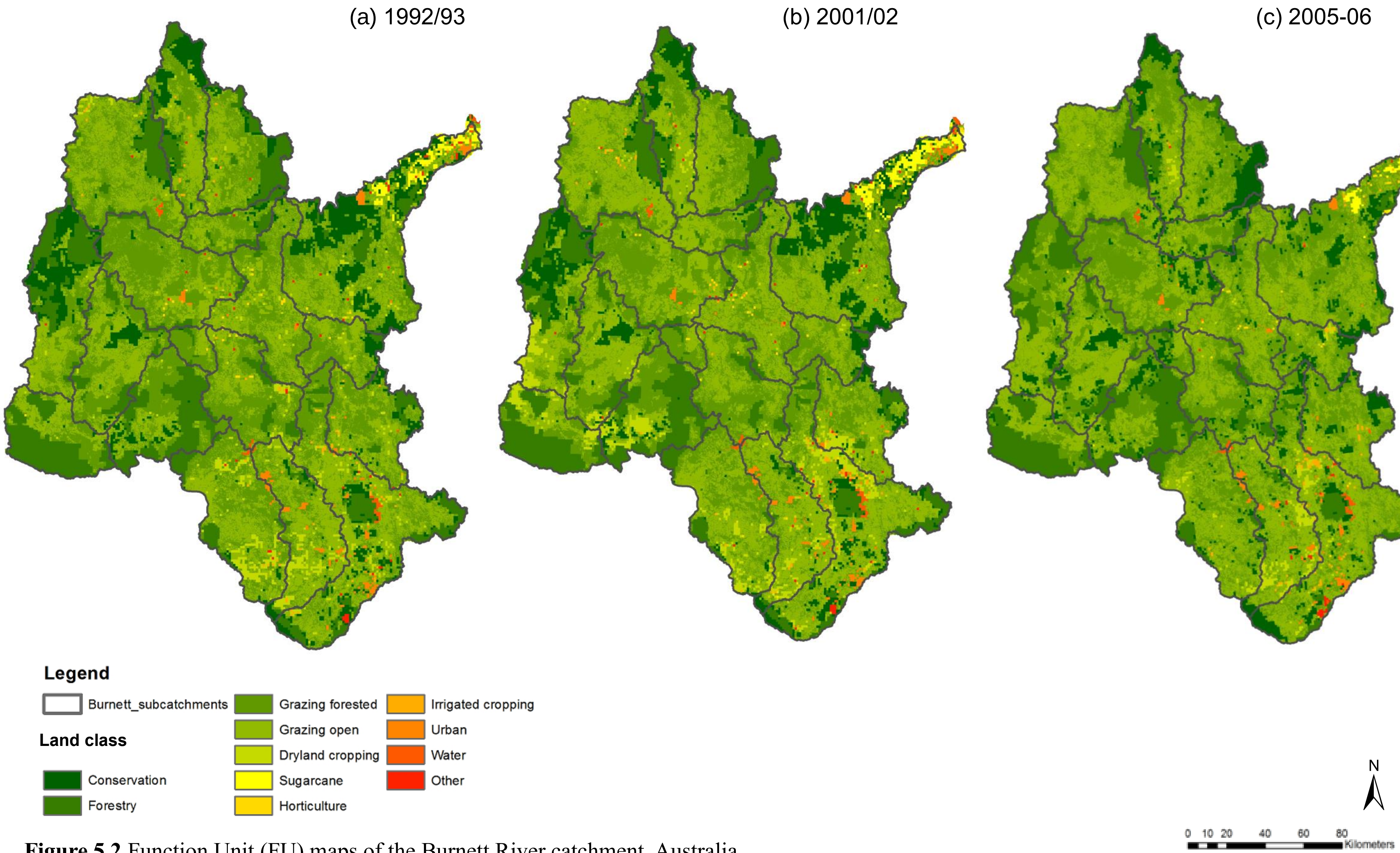


Figure 5.2 Function Unit (FU) maps of the Burnett River catchment, Australia.

The EMC/DWC model was chosen, as it is the simplest and one of the most commonly used constituent generation models in Australia. EMC refers to the average constituent concentration over a storm event, while DWC is the constituent concentration measured during dry weather or baseflow conditions (both expressed in mg/L) (eWater CRC 2010). The model calculates constituent/pollutant load[17] at a time step by applying the EMC value to quick (surface) flow, and the DWC value to slow (base) flow using the following equation (eWater CRC 2010):

Daily pollutant load = quick (surface) flow× EMC + slow (base) flow ×DWC

The model requires EMC/DWC values for each FU. EMC and DWC values vary significantly depending on local conditions, such as soil type, topography, climate and management practices (Chiew & Scanlon 2002), and therefore it is recommended that the values should be derived from a careful analysis of locally observed data, where possible (eWater CRC 2010). The EMC/DWC values for each land use in this application (Appendix Table A.5.3) were derived from modelling results for each FU class as reported in Fentie (in press). Using values, previously developed by Fentie (in press) for the Burnett River catchment, ensured that the average annual runoff and constituent loads generated by *Source* for this application were consistent with the peer reviewed model outputs.

5.2.5 The model simulation period and results

To negate the influence of Australia's highly variable climate on loads, the model was run for a fixed 40 year period from 1 January 1970 to 31 December 2009 for each scenario. This period included both very wet and extremely dry periods and ensured that the model incorporated a broad range of climatic cycles. Hence, climate data for 40 years were prepared for the simulations, and all results were presented as annual average values, with total runoff volume expressed in megalitres (ML) and total loads expressed in tonnes or kilograms. Changes in the annual average values between the periods 1992/93-2001/02, 2001/02-2005/06 and 1992/93-2005/06 were calculated as the percentage change in runoff or load between years.

---

[17] Constituent/pollutant load refers to mass of a particular constituent passing a particular point over a defined period of time (definition by eWater CRC).

5.2.6. Input data for *Fragstats*

Fragstats was employed in the evaluation framework to compute a wide variety of metrics that facilitate understanding of the spatial patterns of vegetation and land-cover. For this evaluation, two types of input maps were prepared for *Fragstats*. First, an analysis was conducted to understand the general changes in the spatial pattern and configuration (structure and composition) of each native vegetation community within the catchment and sub-catchments. This analysis enabled identification of impacts on particular species (e.g. koala) that used particular vegetation communities. Second, the analysis focused on native vegetation with high conservation and/or biodiversity status. In Queensland, including the case study catchment, conservation status of native vegetation and ecosystems is mapped on Regional Ecosystem (RE) maps. This analysis made it possible to identify the changes in native vegetation and/or ecosystems with high conservation and/or biodiversity status (e.g. examining changes in vegetation classified as ‘endangered’ between different time periods).

Changes in native vegetation communities

For the first analysis, these input maps for *Fragstats* were generated to identify vegetation communities related to particular years 1991, 2001 and 2005 (FU/RE maps) (Figure 5.3). They were created as ESRI grids by overlaying RE maps (version 6.0b) (DERM 2009) and the FU maps (Figure 5.2). The RE maps contain comprehensive data about the distribution of “remnant vegetation”[18], ecosystems and their biodiversity status. Datasets were available from 1997. Thus the vegetation map for the early 1990s (in this case 1991) needed to be generated by interpreting the information on SLATS point files, which indicated the location of woody vegetation clearing between 1991 and 97 (DSITIA 2012c). Since no data was available in the SLATS point file dataset for a period 1992-1997, the period 1991-1997 was used to identify the locations and areas of remnant vegetation cleared between the period and to generate the vegetation map for the early 1990s. As a result, the vegetation map reflecting the 1991 status was used as a substitute for 1992/93 land use status for this application.

[18] Remnant vegetation is defined as vegetation that has not been cleared or vegetation that has been cleared but where the dominant canopy has >70% of the height and >50% of the cover relative to the undisturbed height and cover of that stratum and is dominated by species characteristic of the vegetation's undisturbed canopy (Queensland Department of Environment and Heritage Protection, 2013).

The categories of these vegetation maps were consolidated into 17 classes depending on the native vegetation community (when the area was allocated under remnant vegetation on the RE maps) or FU class (when the area was allocated under non-remnant) (Table 5.1). Six vegetation community categories were created by aggregating existing RE map categories (1:5 million Broad Vegetation Groups [BVG]), and the new categories were applied to the remnant vegetation areas on the RE maps (see categories [1] to [6] in Table 5.1). For the remaining areas designated as non-remnant vegetation on the RE maps, 11 land use classes, the same as those used for the FU maps, were applied (see categories [7] to [17] in Table 5.1). It is important to note that as the original land use maps produced at ABARES and BRS are produced at a national scale with a much larger cell size (0.01×0.01 degree, or approximately 1,000×1,000 metres) than the RE maps (vector format), it was necessary to adjust the data for processing. The land use maps were resampled to smaller 25-metre resolution. The RE maps were shapefiles, and therefore the data was converted into grids with 25-metre resolution to capture the details of vegetation patches.

Table 5.1 The classes of FU/RE maps

Value	Classes	Status on RE maps
1	Wet forest	Remnant vegetation
2	Dry eucalypt forest	
3	Brigalow	
4	Non-eucalypt woodland	
5	Grassland	
6	Coastal communities and wetlands	
7	Conservation *	Non-remnant vegetation
8	Forestry (plantation)*	
9	Grazing forested*	
10	Grazing open	
11	Dryland cropping	
12	Sugarcane	
13	Horticulture	
14	Irrigated cropping	
15	Urban	
16	Water	
17	Other	

* Conservation, Forestry and Grazing forested: the areas not allocated under remnant vegetation in RE maps.

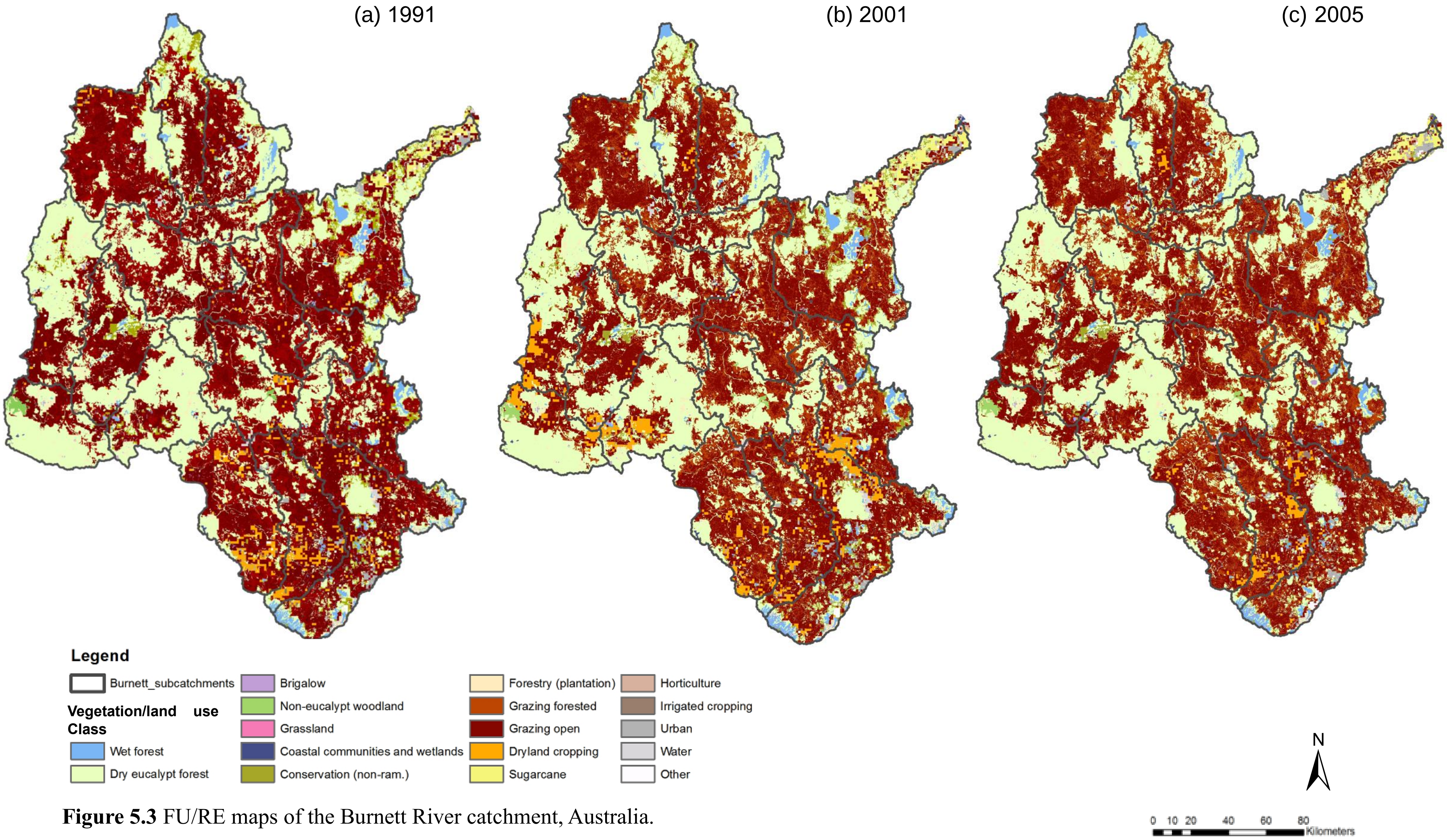


Figure 5.3 FU/RE maps of the Burnett River catchment, Australia.

Changes in vegetation/ecosystems in different conservation/biodiversity status

Queensland's RE datasets contain data on conservation/biodiversity status. Using the 'Biodiversity Status' classes of the dataset, another set of maps for 1991, 2001 and 2005 was generated for the second analysis (Figure 5.4). The Biodiversity Status is determined based on a vegetation condition assessment under the *Vegetation Management Act 1999* (Qld), and is currently the basis for a range of planning and management applications in Queensland. There are three categories of regional ecosystem status defined in the Act: 'endangered', 'of concern', and 'not of concern', and a description of each class is provided in Division 7A (22LA to 22LC) in the *Vegetation Management Amendment Act 2008* (Qld), which replaced the *Vegetation Management Act 1999* (Qld) (Appendix Table A5.4).

Most vegetation patches in the Burnett River catchment were composed of a range of vegetation communities with different Biodiversity Status. However, since *Fragstats* recognises only one class for a patch, input maps for *Fragstats* were prepared to include only one Biodiversity Status for a patch. Thus patches were reclassed into: 'endangered', 'of concern', 'not of concern', or 'other' (non-remnant). This preparation was conducted in accordance with this rule: higher biodiversity status was always prioritised over a lower one. For example, all patches including vegetation classified into 'endangered' were categorised as 'endangered' regardless of the percentage within the patches.

5.2.7 Running Fragstats

After the above procedure, all six maps (Figure 5.3 and 5.4) were input into *Fragstats* (ver. 4.0) for processing. Each metric had a different unit of measurement, e.g. Class Area (CA) was presented in hectares (ha), Largest Patch Index (LPI) in percentages (%) and number of patches (NP) in number. Changes in 1991-2001, 2001-05 and 1991-2005 were expressed as percentages.

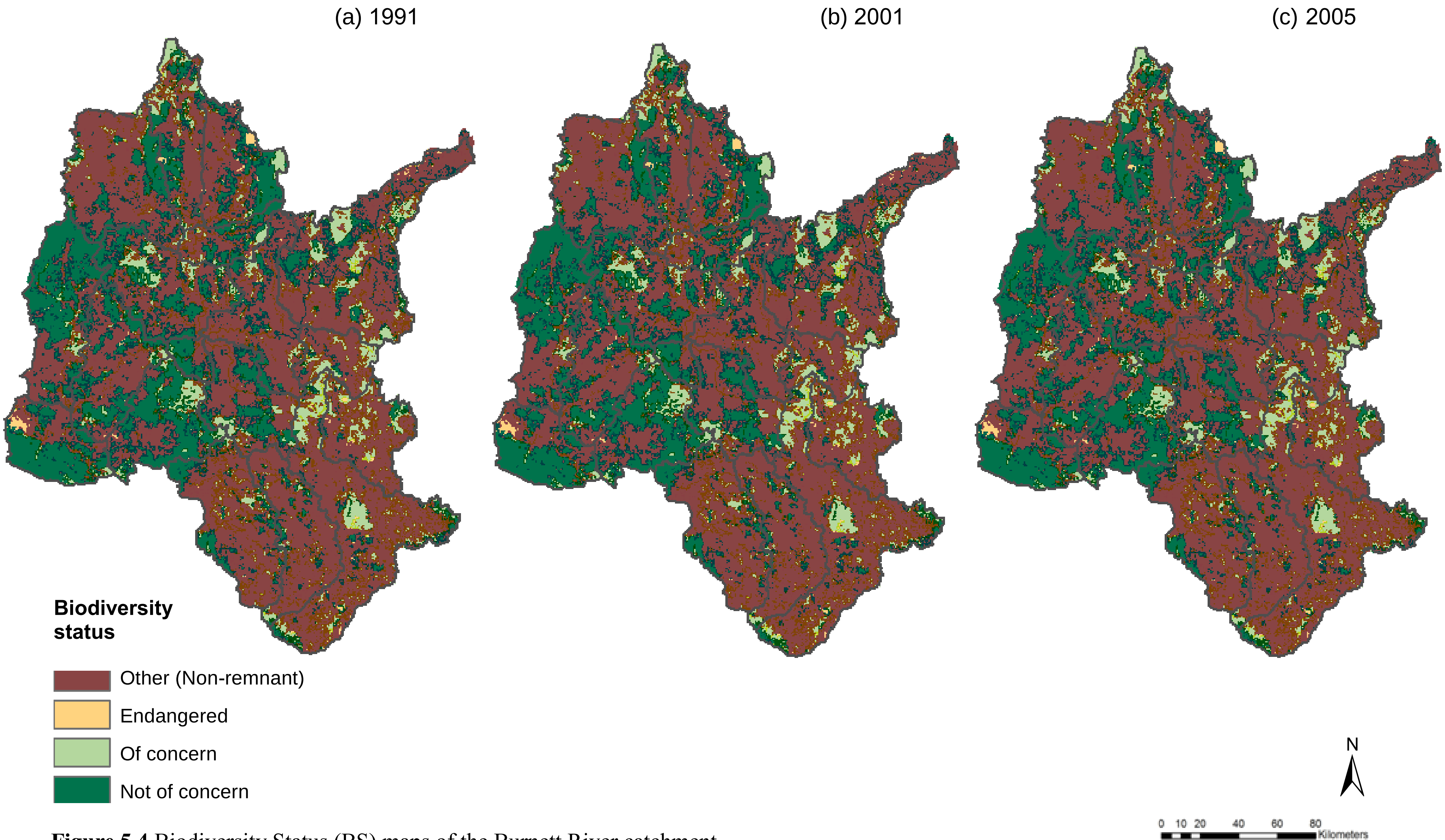


Figure 5.4 Biodiversity Status (BS) maps of the Burnett River catchment.

5.2.8 Biodiversity conservation values and actual habitat value

The biodiversity conservation values of different land uses (e.g. urban, cropping, grazing, forestry) have been studied globally. The mean abundance of original species relative to their abundance in undisturbed vegetation (MSA) was developed as an indicator for the GLOBIO3 model to represent changes in the abundance of local species (Alkemade et al. 2009). This concept of MSA value was adopted in this evaluation framework, with slight modification in relation to the land use category and value to quantify the biodiversity conservation value of the case study region as 'actual habitat amount'. This analysis was needed because the spatial pattern analysis using *Fragstats* did not capture the biodiversity or habitat values for the non-native vegetation areas. The MSA concept has been used by European bioenergy researchers in biodiversity assessments (Dornburg et al. 2008; van Rooij 2008; Alkemade et al. 2009; van der Hilst et al. 2012a) and provides an assessment of the biodiversity values for non-native vegetation areas relative to native vegetation corresponding to each land use class (BCV_{LU}) using the following equation:

$$\text{Actual habitat amount of the area (ha)} = \sum_{i=1}^{n} BCV_{LUi} \times CA_i$$

where BCV_{LUi} is the biodiversity conservation value of land use *i* represented by a number of zero and one (with one being a biodiversity value the same as native vegetation and zero being no biodiversity value), and CA_i is the land area of land class *i*.

For this purpose, land use categories were modified in accordance with the land class used for the FU/RE maps. Due to considerable variation in values between individual studies (Dornburg et al. 2008), the BCV_{LU} of each land class was determined in the Southeast Queensland context from: a review of spatial ecology literature on specific taxa and species native to the region, including mammals, birds and reptiles; and discussions with an expert familiar with the ecology of the region's ecosystems and fauna (C. McAlpine 2014, pers. comm., 20 May). Eyre et al. (2009) confirmed that species composition (diversity and abundance) in the landscape was largely affected by the intensity of human activity - grazing, logging, and land management practices. The ranking of the BCV_{LU} of remnant vegetation reflects the range of disturbance levels that impact on different vegetation communities. For example, the region's wet forests are largely protected, while dry eucalypt forest and 'brigalow' vegetation are subject to higher levels of grazing, logging and fire disturbance which affect their ecological condition and capacity to support native birds, reptiles and mammals (Eyre et al. 2009; C. McAlpine 2014, pers. comm., 20 May). Past studies generally indicated very limited roles for cropping and open grazing lands in supporting species, more specifically birds and reptiles in the southeast and south central Queensland contexts (Martin & Catterall 2001; Kanowski, Catterall & Wardell-Johnson 2005; Kanowski et al. 2006; Martin et al. 2006; Eyre et al. 2009). Eucalypt plantations are common in the case study region, and yet different views were expressed concerning their ecological values. Smith and Agnew (2002) suggested that mature eucalypt plantations could provide habitats for microbats and arboreal mammal species. Kanowski et al. (2006) and Kanowski, Catterall and Wardell-Johnson (2005) on the other hand argued that the eucalypt plantations had no intrinsic habitat value for rainforest biodiversity, including bird and reptile species. The ecological values of eucalypt plantations depend very much on the stand age, with older stands having greater structural diversity and being able to support more bird, reptile and mammal species.

From such information, values were given to all land use categories, including bioenergy crops (Table 5.2) (refer to Chapter 6 for further discussion on values for bioenergy land use). It is important to note that the values were not intended to highlight individual species, but to express the ecosystem intactness of each land category. Also, as the values treat all ecosystem types equally, they are not sensitive to the species richness of different biomes. To overcome challenges in relation to variety in ecosystem conditions for the same land/vegetation categories and also high levels of uncertainty in bioenergy land use, value ranges were provided for all BCV_{LU}s. Based on those values, the actual habitat amount of the area (ha) was gained from the above procedure for 1991, 2001 and 2005 and was compared between different time periods.

Table 5.2 Biodiversity conservation values corresponding to land cover/land use categories (BCV_{LU})

	Categories	Status on RE maps	Southeast QLD (BCV_{LU})**	
			Low	High
1	Wet forest	Remnant vegetation	0.8	1.0
2	Dry eucalypt forest		0.7	1.0
3	Brigalow		0.7	1.0
4	Non-eucalypt woodland		0.7	1.0
5	Grassland		0.65	1.0
6	Coastal communities and wetlands		0.75	1.0
7	Conservation *	Non-remnant vegetation	0.5	0.8
8	Forestry (plantation)*		0.25	0.5
9	Grazing forested*		0.3	0.7
10	Grazing open		0.1	0.2
11	Dryland cropping		0.05	0.15
12	Sugarcane		0.05	0.15
13	Horticulture		0.05	0.15
14	Irrigated cropping		0.05	0.25
15	Urban		0.1	0.25
16	Water		-	-
17	Other		0.05	0.15
18	Bioenergy crop (Pongamia)		0.15	0.3
19	Bioenergy crop (eucalypts)		0.15	0.3

* Conservation, Forestry and Grazing forested: the areas not allocated under remnant vegetation in RE maps.

** Biodiversity conservation values were adopted from generic MSA value by van Rooij (2008) (original values are in Table 3.4).

5.3 Results

A summary of the environmental outcomes of the Burnett River catchment based on three land use maps is presented in 5.4.5. Prior to the section, the detailed results in land use change (5.3.1), water quantity and quality (5.3.2), native vegetation patterns (5.3.3), and biodiversity conservation value and actual habitat (5.3.4) are explained. The analysis was conducted on both catchment and sub-catchment scales. Depending on the case study catchment, there may be significant differences in results between two different scale analyses. In the Burnett River catchment, the sub-catchment scale trends were generally similar to those identified for the whole catchment, thus all results are presented in the same sections.

5.3.1 Change in land use

Using *Source*, the area of each Function Unit (FU) was calculated from the land use maps (Figure 5.2) for the years 1992/93, 2001/02 and 2005/06 in the Burnett River catchment (Table 5.3) and all sub-catchments (Appendix Table A5.5). Overall, no major land use change was identified after 1992/93 on a catchment scale, because most of the vegetation clearing for agriculture had occurred prior to the 1990s. Nevertheless, a number of small catchment scale land use changes were identified. Over all three time periods, 'conservation' areas diminished to make way for other land uses, while total grazing areas (land uses (3) and (4) in Table 5.3) and 'urban' land uses showed a consistent increase. Another notable trend was the significant decline in 'dryland cropping' area which was converted to other types of agriculture, especially grazing after 2001/02 (Chapter 4). This was due to the higher risks of 'dryland cropping' associated with climate variability (I. Crosthwaite 2012, pers. comm. [interview], 12 March 2012; D. O'Sullivan 2012, pers. comm. [interview], 13 March 2012) and the expansion of irrigation systems in coastal areas (I. Doyle 2012, pers. comm. [e-mail], 17 August). The area for production and plantation forestry under the 'forestry' classification also decreased after 2001/02.

Table 5.3 Area of each Function Unit (FU) class on 1992/93, 2001/02 and 2005/06 FU maps (Figure 5.2) in the Burnett River catchment, Australia (Unit: ha)

Value	FU class	1992/93	2001/02	2005/06
1	Conservation	288,759	288,106	287,398
2	Forestry*	477,071	479,271	394,109
3	Grazing forested	1,282,196	1,072,499	1,310,235
4	Grazing open	1,096,151	1,248,548	1,187,836
5	Dryland cropping	75,103	115,640	37,735
6	Sugarcane	18,302	26,628	17,546
7	Horticulture	5,753	8,405	4,954
8	Irrigated cropping	8,865	10,750	10,150
9	Urban	21,204	20,755	22,939
10	Water	5,799	8,559	8,301
11	Other	6,834	6,876	4,834
TOTAL		3,286,037	3,286,037	3,286,037
3+4	Grazing TOTAL	2,378,347	2,321,047	2,498,071

*Forestry refers to production forestry (2.2.0) and plantation forestry (3.1.0, 3.1.2, 3.1.3) land use categories on the Australian Land Use and Management Classification (ALUM).

In general, similar trends were observed at a sub-catchment scale, although the local change was highly dependent on the sub-catchment land use (Appendix Table A5.5). The land use changes in the coastal sub-catchment (B1) and the inland South Burnett (B3, B4, B7 and B8) were described in Chapter 4 (see 4.3.5, and Figure 4.10 and 4.11). It is important to note that the 'conservation' area increased significantly in sub-catchments in the Central Burnett (B5, B6, B11 and B14) after 2001/02, while it declined in many other sub-catchments. In reality, the Queensland government converted state forests to 'conservation' areas under the extensive Statewide Forests Process between 1999 and 2009 (DSITIA 2012d), and this decline in 'conservation' area was due largely to a change in the mapping methodology for 'conservation and natural environments' classes between *version 3* and *version 4* of *Land Use of Australia* (J. Mewett 2013, pers. comm. [e-mail], 9 August). In addition, a reduction of the 'forestry' area was prominent in the North Burnett sub-catchments (B16, B17, and B18) after 2001/02, when compared to the rest of the catchment.

5.3.2. Water quantity and quality results from *Source*

The average annual run-off volume (ML/year), and the average annual loads of sediment (TSS) (tonne/year), nitrogen (TN) and phosphorous (TP) (kg/year) were simulated for the 40 year model run period for the three land use scenarios 1992/93, 2001/02 and 2005/06. From the outputs, changes in average annual run-off volume, TSS and TN and TP loads in 1992/93-2001/02, 2001/02-2005/06 and 1992/93-2005/06 were obtained for each FU class, and all sub-catchments. In addition, maps were created with the support of ArcGIS 10 to show the distribution of estimated average annual run-off volume, loads of TSS, TN and TP and the changes over the period for sub-catchment scale analysis.

The estimated average annual run-off volume

The estimated average annual run-off volume of the Burnett River catchment showed very little change at a catchment scale between the three time periods (Figure 5.5). Nevertheless, there were variations observed that related to the FU class (Figure 5.6). Corresponding to the land use change data, the run-off from 'dryland cropping' (50.0%), 'sugarcane' (45.5%) and 'horticulture' (36.5%) significantly increased from 1992/93-2001/02, but dropped sharply in 2001/02-2005/06 (-64.8%, -34.1% and -34.5%, respectively). In 1992/93-2005/06, increases in run-off volume were most significant for 'irrigated cropping' (25.8%) and 'grazing open' (7.5%) land uses, and decreases in run-off volume were most significant for 'dryland cropping' (-47.3%), 'forestry' (-22.3%), 'other' (-21.5%), and 'urban' (-15.6%) land uses.

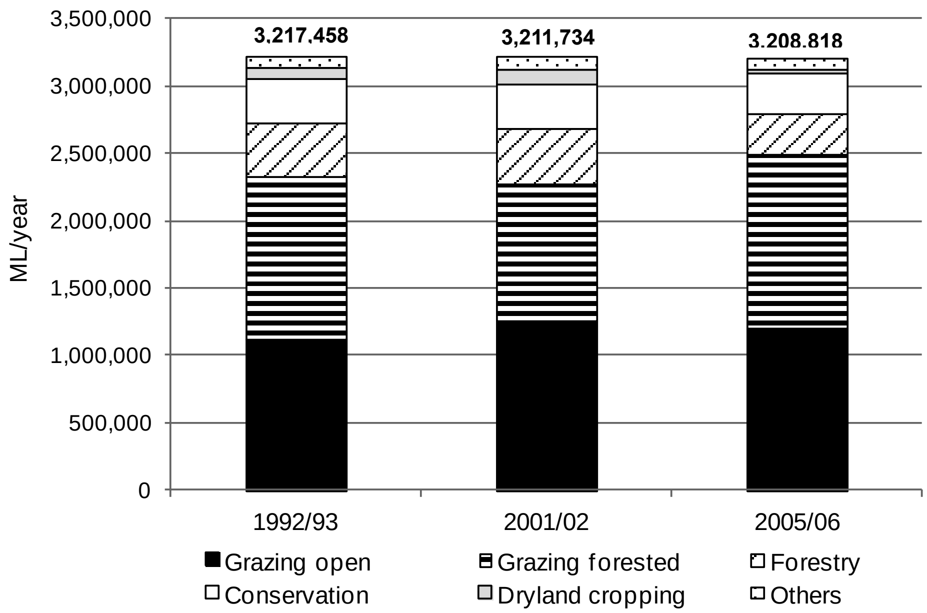


Figure 5.5 Estimated average annual run-off volume of the Burnett River catchment, Australia (Unit: ML/year).

The estimated average annual run-off volume showed little change for most sub-catchments (Figure 5.7), mostly within the range of ± 0.5% between 1992/93 and 2005/06 (Figure 5.9). This was attributed to the fact that there were only minor changes in land use area between these time periods. The two largest sub-catchments B1 and B4 generated the highest run-off volume, contributing 18.9% and 16.2% respectively to the catchment's average annual volume in 2005/06 (3,208,800 ML/year). This was due to their larger catchment areas, higher rainfall and steeper slopes. The upland sub-catchments B15, B14, B5 and the coastal area B8, had runoff in the second highest range (150-300,000 ML/yr). These areas also exhibited higher annual rainfall than the lower runoff yielding areas (Figure 5.8).

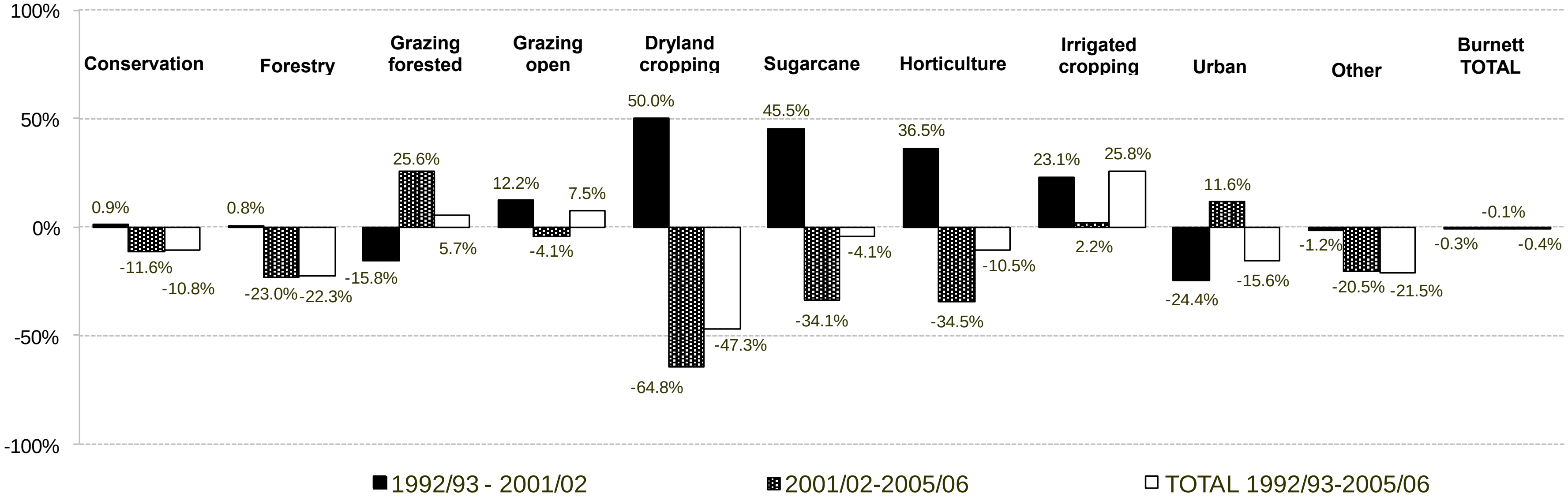


Figure 5.6 Changes in estimated average annual run-off volume by FU between 1992/93 and 2005/06 (%).

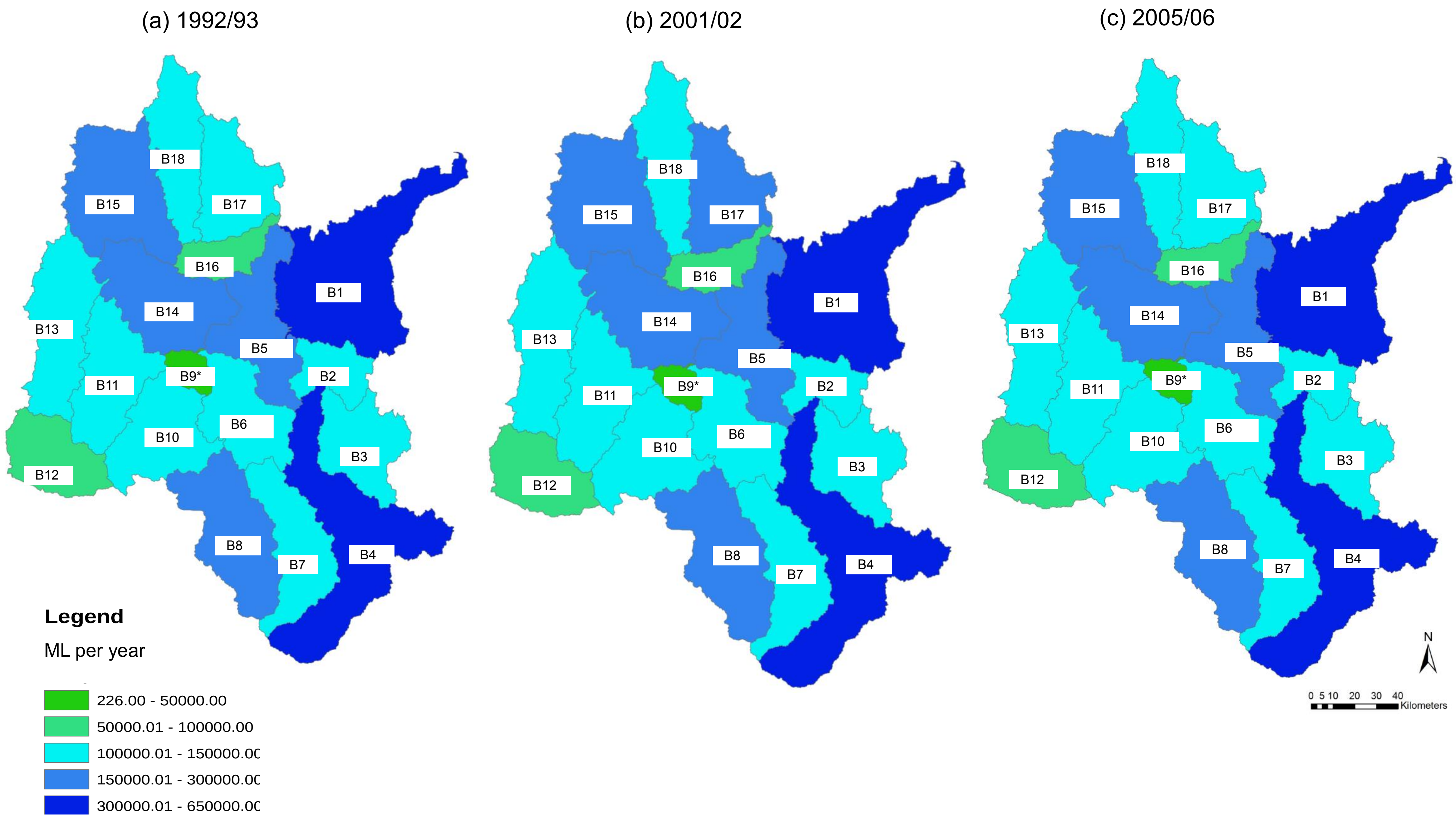


Figure 5.7 Estimated average annual runoff-volume by sub-catchment (ML/year)

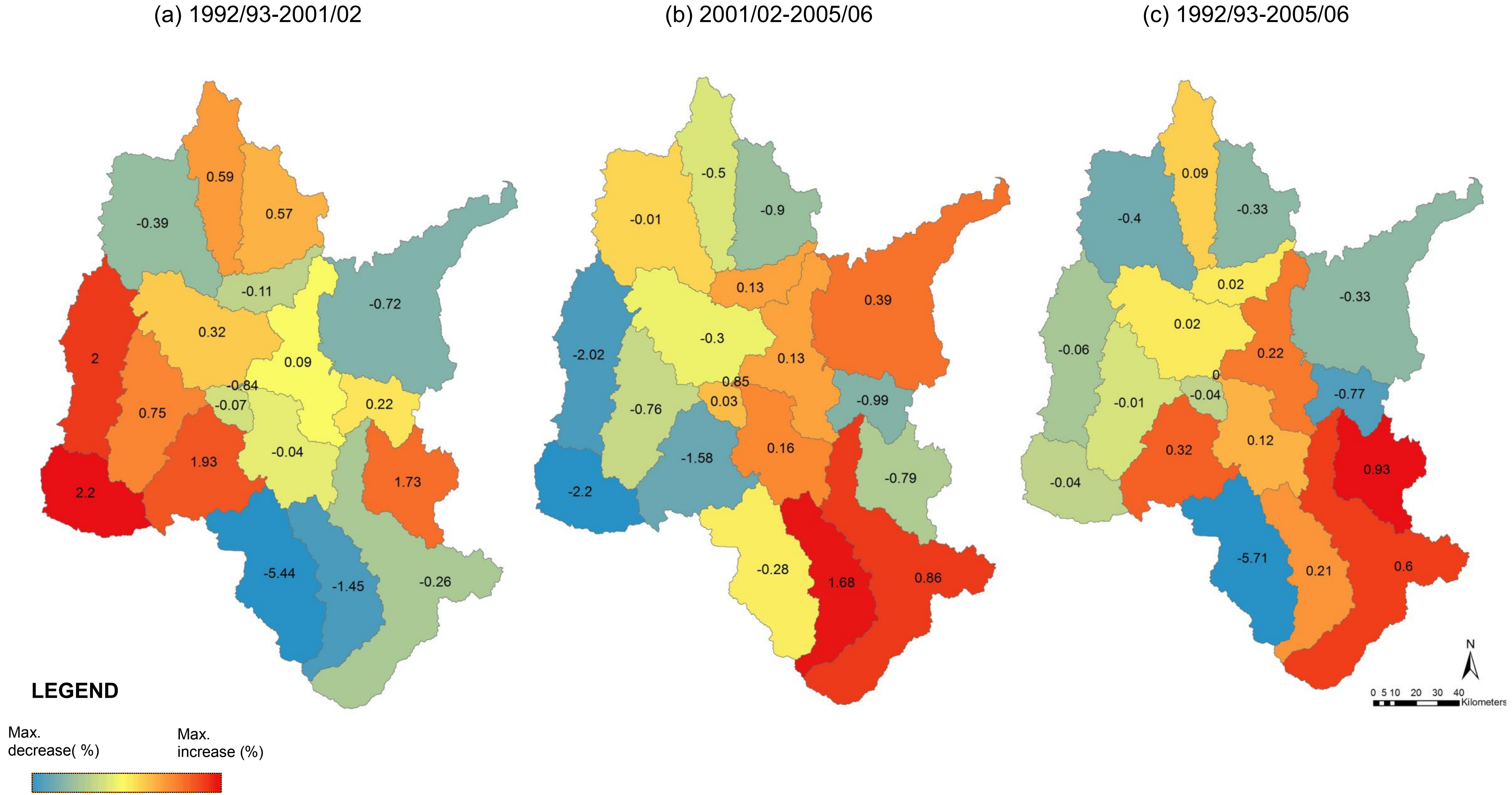


Figure 5.8 Change in estimated average annual runoff-volume by sub-catchment (%).

The estimated average nutrient loads

Source simulated that the 2001/02 period generated the largest amount of TSS, TN and TP loads. These three indicators are very closely related to each other, since nitrogen and phosphorous loads are often sediment associated (Wolfe 2001). In this catchment, sediment load was assumed to be highly related to soil erosion from the grazing land, which is the predominant land use (almost 80%) (Figure 5.9). All results for these three were primarily affected by the land use and its changes since EMC/DWC numbers were prepared corresponding to FU classification.

The estimated TSS for the Burnett River catchment showed a slight decrease (-3.7%) between 1992/93 and 2005/06. Expansion up to 2001/02 and subsequent decline between 2001/02 and 2005/06 in 'dryland cropping' area significantly affected the total changes over the period (Figure 5.9). The TSS load that originated from 'dryland cropping' almost doubled between 1992/93 and 2001/02, while halving (-48.2%) over the total period because of the rapid drop (65.5%) in 'dryland cropping' after 2001/02 (Figure 5.10). 'Sugarcane' and 'horticulture' followed similar patterns, although the changes in percentage were not as significant as for 'dryland cropping'.

Total grazing area (i.e. 'grazing forested' and 'grazing open'), 'irrigated cropping' and 'urban' land uses contributed to the increase in the TSS load during the period 1992/93-2005/6 (Figure 5.10). Cattle grazing was the predominant land use in the Burnett River catchment. Grazing results in vegetation removal, overgrazing and stream bank erosion by cattle access (Brodie & Mitchell 2005) which cause considerable soil erosion. The model suggested that in 2005/06, grazing contributed 82.7% of the total TSS load of the catchment (Figure 5.9). Although it only accounted for less than 1% of the total catchment area, 'urban' land area increased over the period (Table 5.3). The estimated TSS load generated from 'urban' land use increased 62.7% by 2005/06 (Figure 5.10).

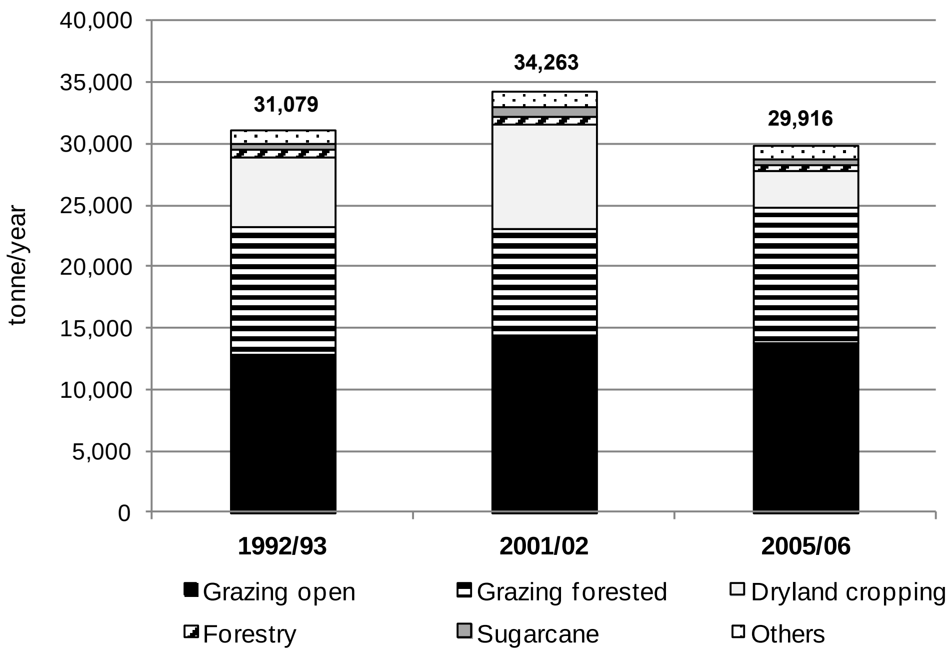


Figure 5.9 Estimated average annual TSS load of the Burnett River in 1992/93, 2001/02 and 2005/06 (tonne/year)

The estimated TN and TP loads from grazing lands increased from 60.6% to 68.2% in TN, and from 67.0% to 75.3% in TP between 1992/93 and 2005/06 (Figure 5.11 and 5.12), due mainly to the expansion of grazing areas. The TN and TP loads from grazing lands in the catchment were predominantly natural, derived from the soils of the area attached to soil particles in runoff. In general, TN and TP from grazing originate from manure from grazing animals and surface-applied fertilisers for modified pastures (especially phosphorus input) (Gourley & Weaver 2012). However, there was very little fertiliser applied to grazing lands in the Burnett River catchment.

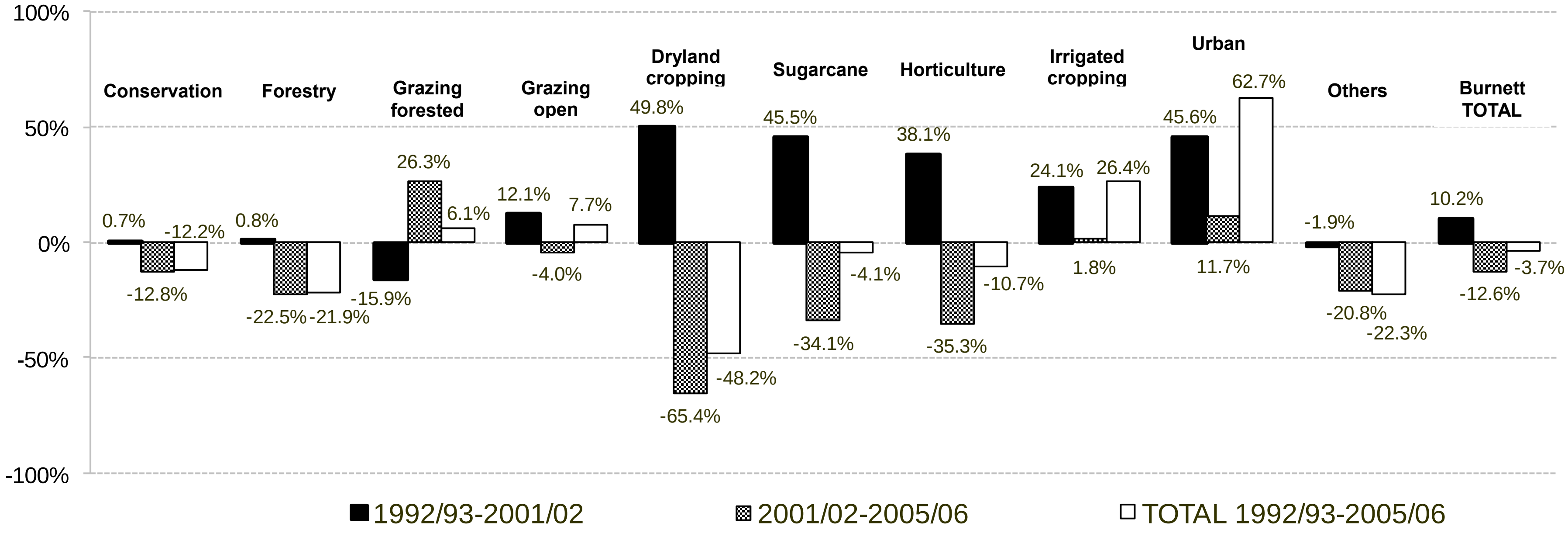
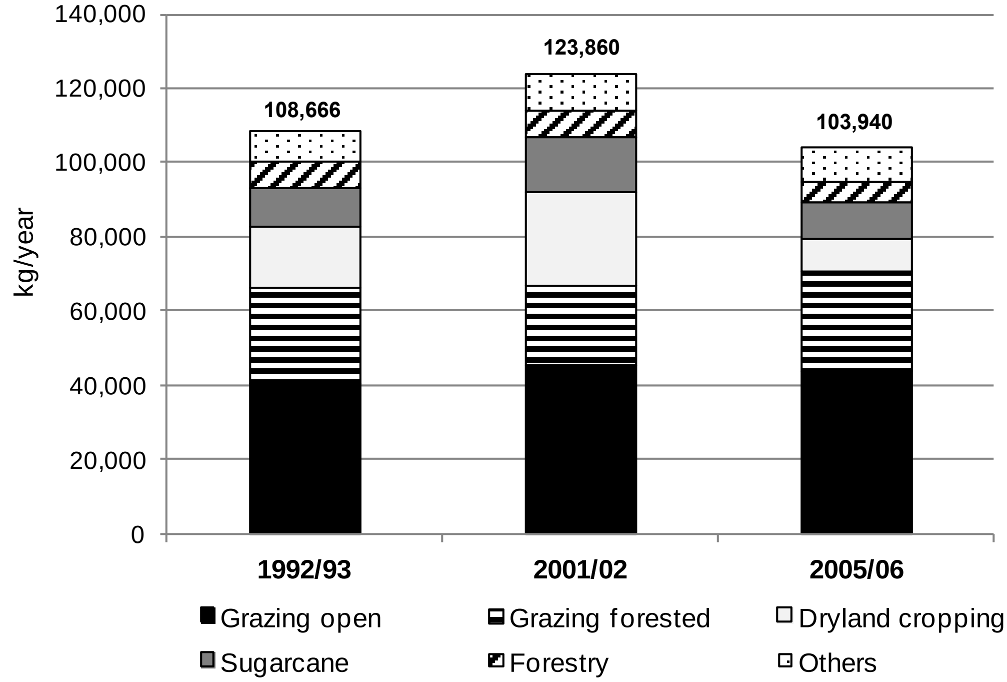
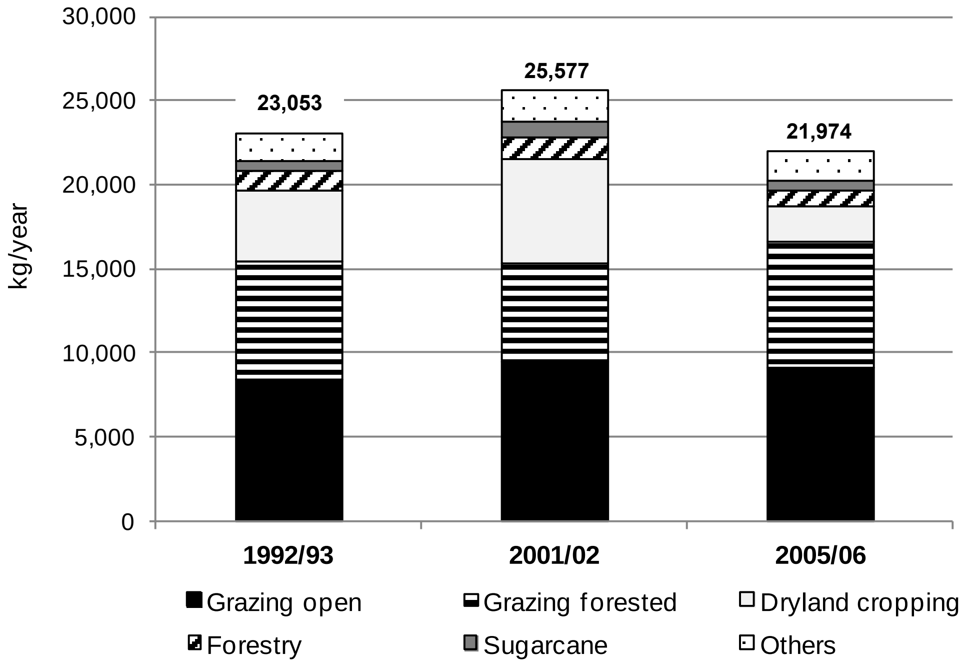


Figure 5.10 Changes in the estimated average annual TSS load of the Burnett River catchment between 1992/93-2001/02, 2001/02-2005/06, and 1992/93-2005/06 (%)



(a) Total Nitrogen (TN) (kg/year)



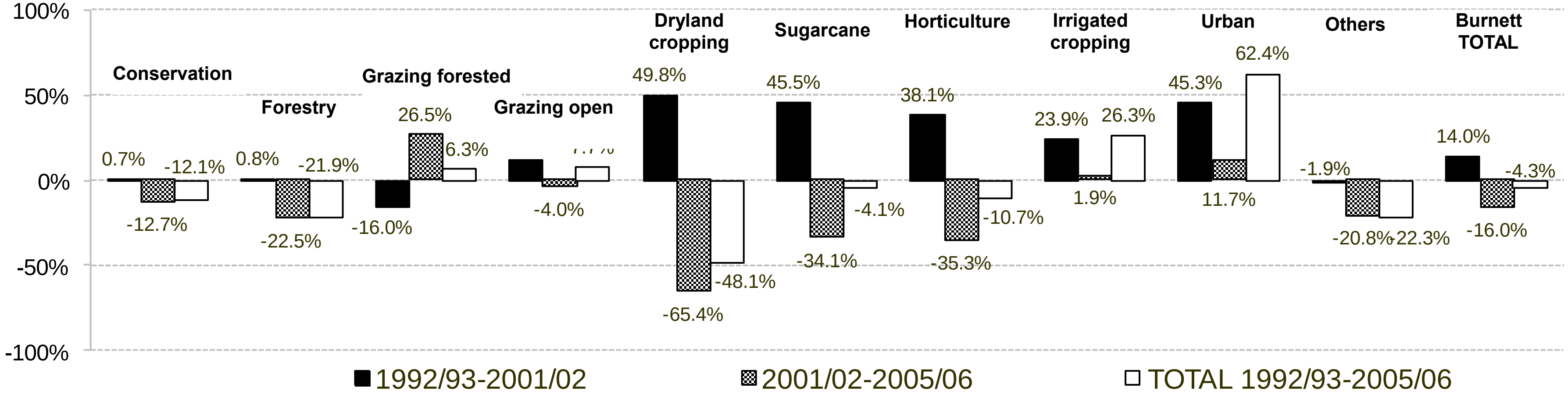
(b) Total Phosphorous (TP) (kg/year)

Figure 5.11 Estimated average annual TN load (a) and TP (b) load of the Burnett River in 1992/93, 2001/02 and 2005/06 (kg/year).

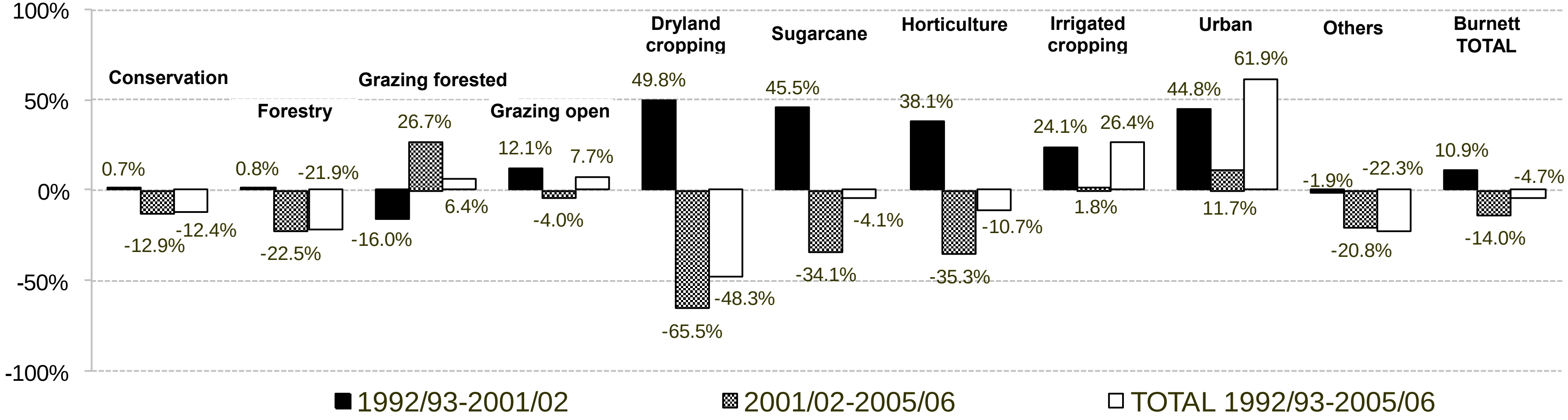
The total cropping area (i.e. total of 'dryland cropping', 'sugarcane', 'horticulture', and 'irrigated cropping') comprised a large fraction of the annual average TN and TP loads. The TN load entering from cropping lands was 26.6% in 1992/93, 34.2% in 2001/02, and 19.7% in 2005/06 (Figure 5.12). Fertiliser is the primary agricultural non-point source of nitrogen (N) and phosphorous (P) on agricultural catchments worldwide (Sharpley 2002), and cropping lands produce larger fluxes of N and P from applied fertiliser through surface water runoff or groundwater (Brodie & Mitchell 2005). Sugarcane production is the major user of nitrogen fertiliser in Queensland. In the Burnett River catchment, sugarcane production occurred in the coastal strip (sub-catchment B1) accounting for 0.9% of the total 2001/02 FU map (Table 5.3). However, a disproportionally high percentage (12%) of the TN load was attributed to 'sugarcane' production (Figure 5.12). Applications of N and P fertiliser increased 10-fold and six-fold respectively between 1950 and 1990 in Queensland cane fields (Pulsford 1996). Brodie and Mitchell (2005) estimated from past studies that approximately 180kg $ha^{-1}year^{-1}$ of nitrogen fertiliser was applied on average to Queensland sugarcane production, of which 110kg ha^{-1} $year^{-1}$ was lost to the atmosphere, water or soil storage, and a large proportion reached adjacent streams and rivers in GBR catchments.

To minimise the sediment and nutrient loads from agricultural activities, a number of programs have been implemented under the Reef Plan (Queensland Department of the Premier and Cabinet 2009b). In the Burnett River catchment, industry organisations are working in partnerships with governments and the Burnett Mary Regional Group for Natural Resource Management Ltd. (BMRG) to improve land management practices. These groups include: the Queensland Dairy Organisation (dairy), Agforce (beef), Cane Growers, BSES Limited etc. (sugarcane) and Growcom etc. (horticulture).

Urbanisation also led to increased TN and TP loads (Figure 5.12). In Australian catchments, high nutrient inputs to urban runoff primarily come from point sources such as municipal sewage treatment plants. The slight increase in the size of urban areas in the catchment from 1992/93-2005/06 resulted in a 62.4% and 61.9% increase in the TN and TP loads respectively (Figure 5.12).



(a) Change in the estimated average annual TN load (%).



(b) Changes in the estimated average annual TP load (%).

Figure 5.12 Changes in the estimated average annual (a) TN load and (b) TP load of the Burnett River catchment between 1992/93-2001/02, 2001/02-2005/06, and 1992/93-2005/06 (%).

As sediment and nutrients are exported in runoff, the sub-catchments with high run-off volumes generally had high exports of TSS, TN and TP (e.g. B1, B4, B8, B15, B14 and B5). As the daily load is a product of the daily runoff and the EMC/DWC values, the land use class also affected the results of the estimated TSS, TN and TP loads substantially since EMC/DWC values reflect the generation rates for different FU classes. Between 1992/93 and 2005/06, the TSS, TN and TP loads increased in sub-catchments B13, B18, B2 and B4, while the rest showed a decrease in various degrees. In particular, B8, B6, B9, B15, B3, B14 and B5 experienced the most prominent decrease (Figures 5.14, 5.16 and 5.18).

Each sub-catchment had specific land use change patterns over the period. Expansion of grazing areas led to the increased sediment and nutrients loads in B13. The later landscape pattern analysis by *Fragstats* provided additional evidence indicating that native vegetation loss and habitat fragmentation occurred in this particular sub-catchment resulting from an increase in grazing areas. On the other hand, a significant increase in cropping areas (particularly 'dryland cropping' and 'irrigated cropping') in B2 and B18 boosted the TSS, TN and TP loads (Appendix Table A5.5). In B1 and B4, the increase in 'urban' area also significantly contributed to the increase in these pollutant loads. Conversely, a reduction in the sediment and nutrient loads in many sub-catchments reflected a reduction in cropping lands (particularly, 'dryland cropping') over the period. Nevertheless, the coastal sub-catchment B1 did not follow these patterns. Here the TSS and TP loads increased 12.9% and 5.0% respectively (Figures 5.14 and 5.18) between 1992/93 and 2005/06, while the TN load showed very little change (Figure 5.16). The reduction of the estimated TN load may have been caused by the reduction of 'sugarcane' and 'dryland cropping' areas after 2001/02, while the sub-catchment showed a consistent increase in grazing and urban areas over the period (Appendix Table A5.5).

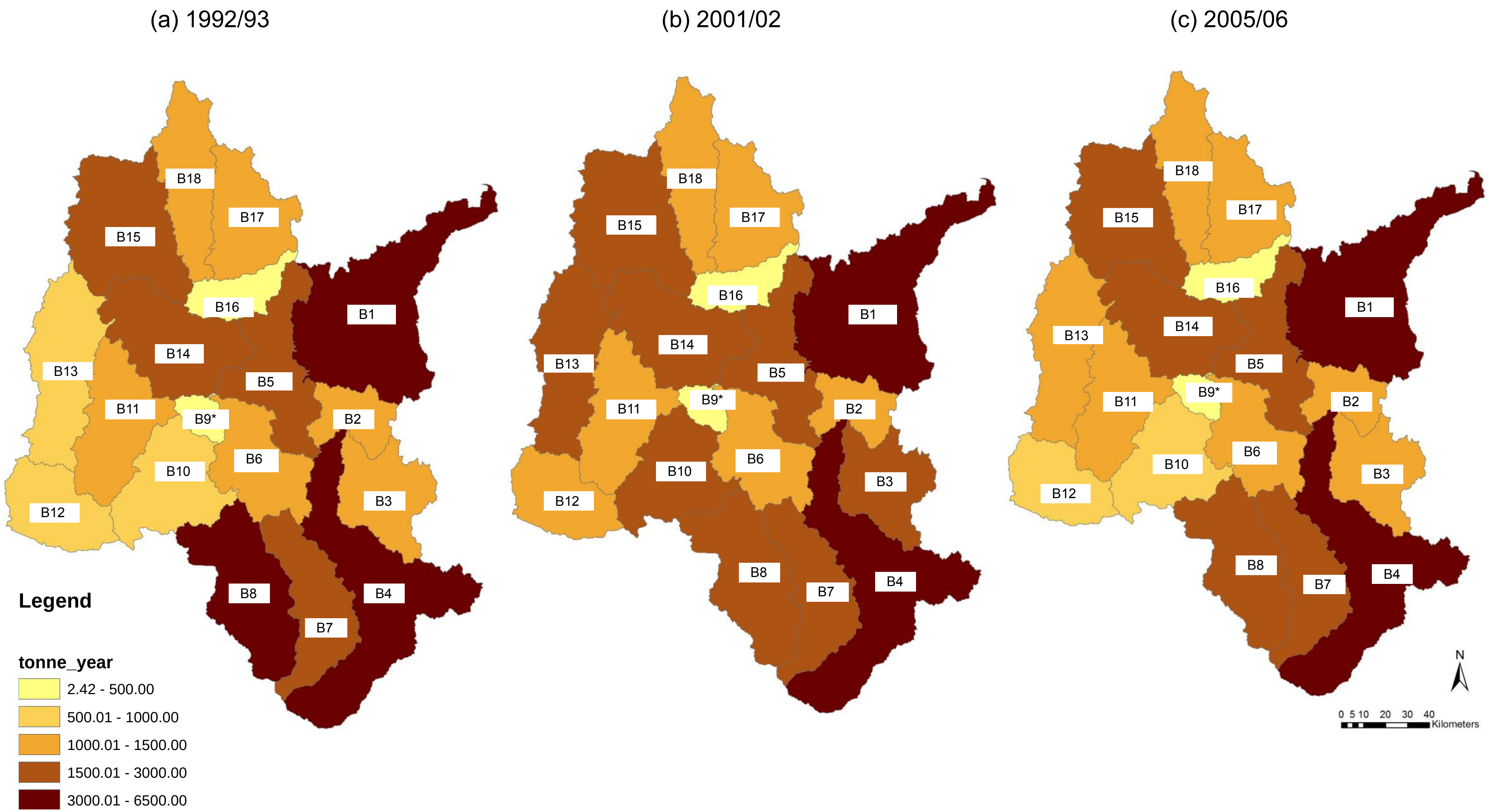


Figure 5.13 Estimated average annual TSS loads by sub-catchment (tonne/year).

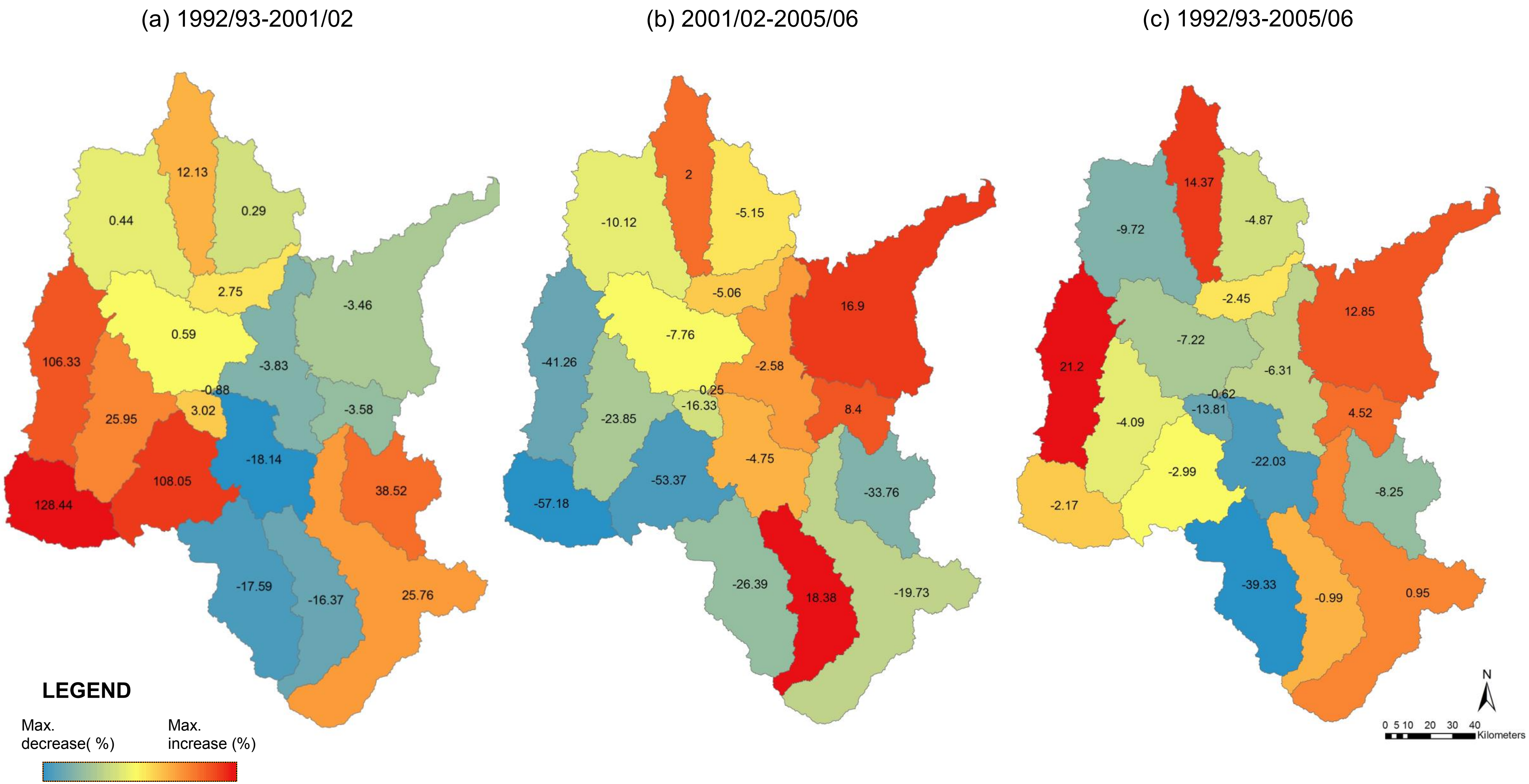


Figure 5.14 Change in estimated average annual TSS load by sub-catchment (%).

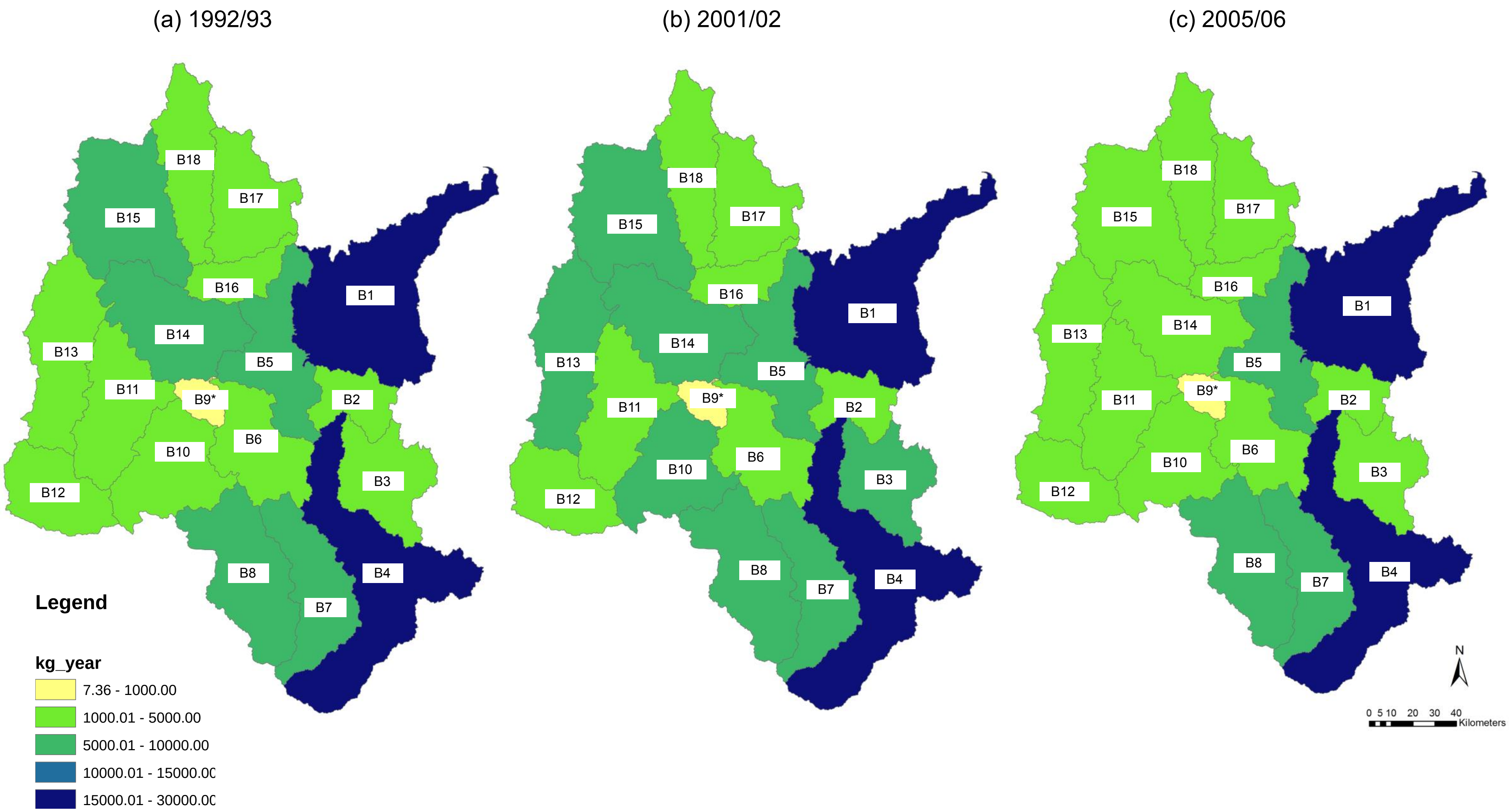


Figure 5.15 Estimated average annual TN loads by sub-catchment (kg/year).

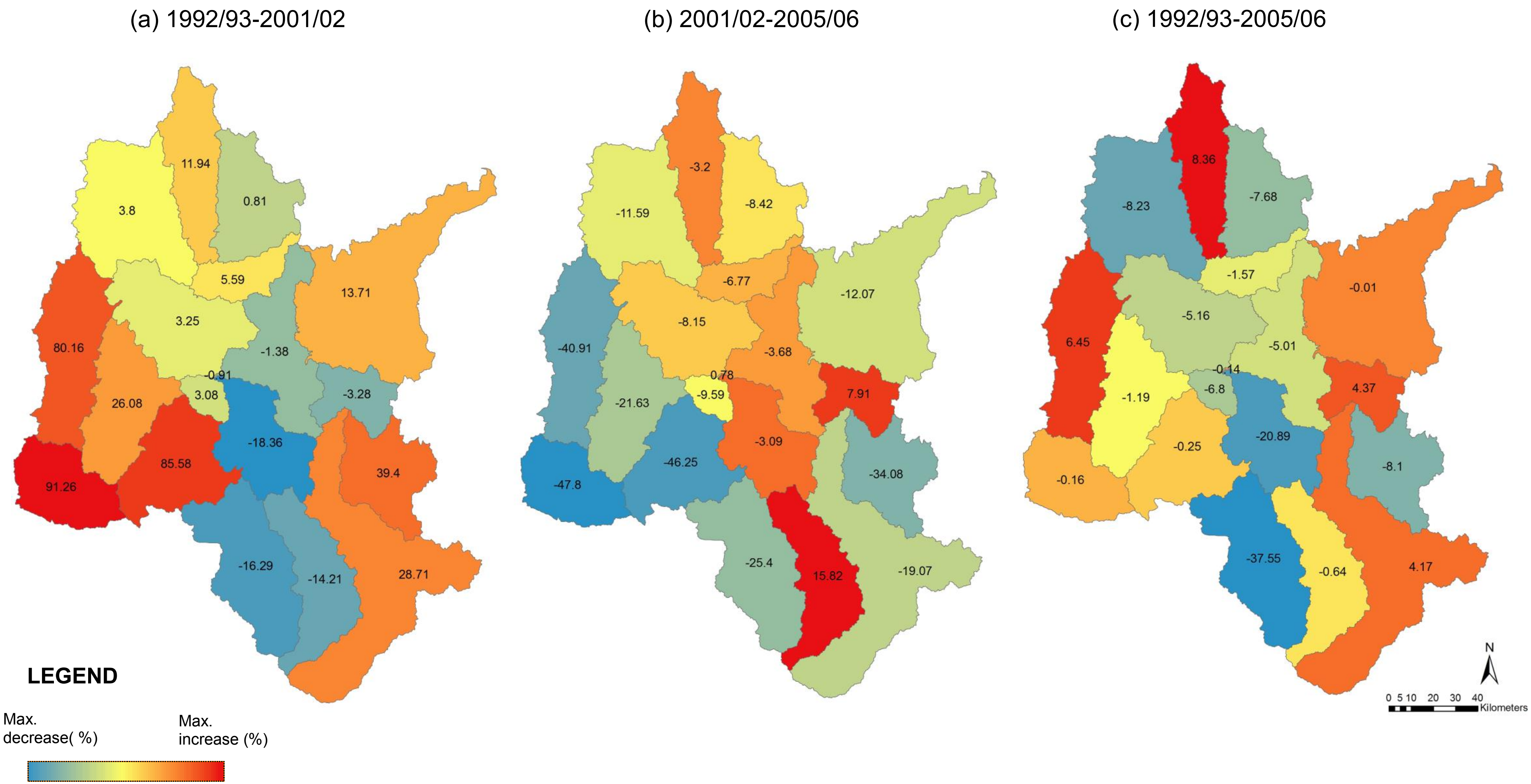


Figure 5.16 Change in estimated average annual TN load by sub-catchment (%).

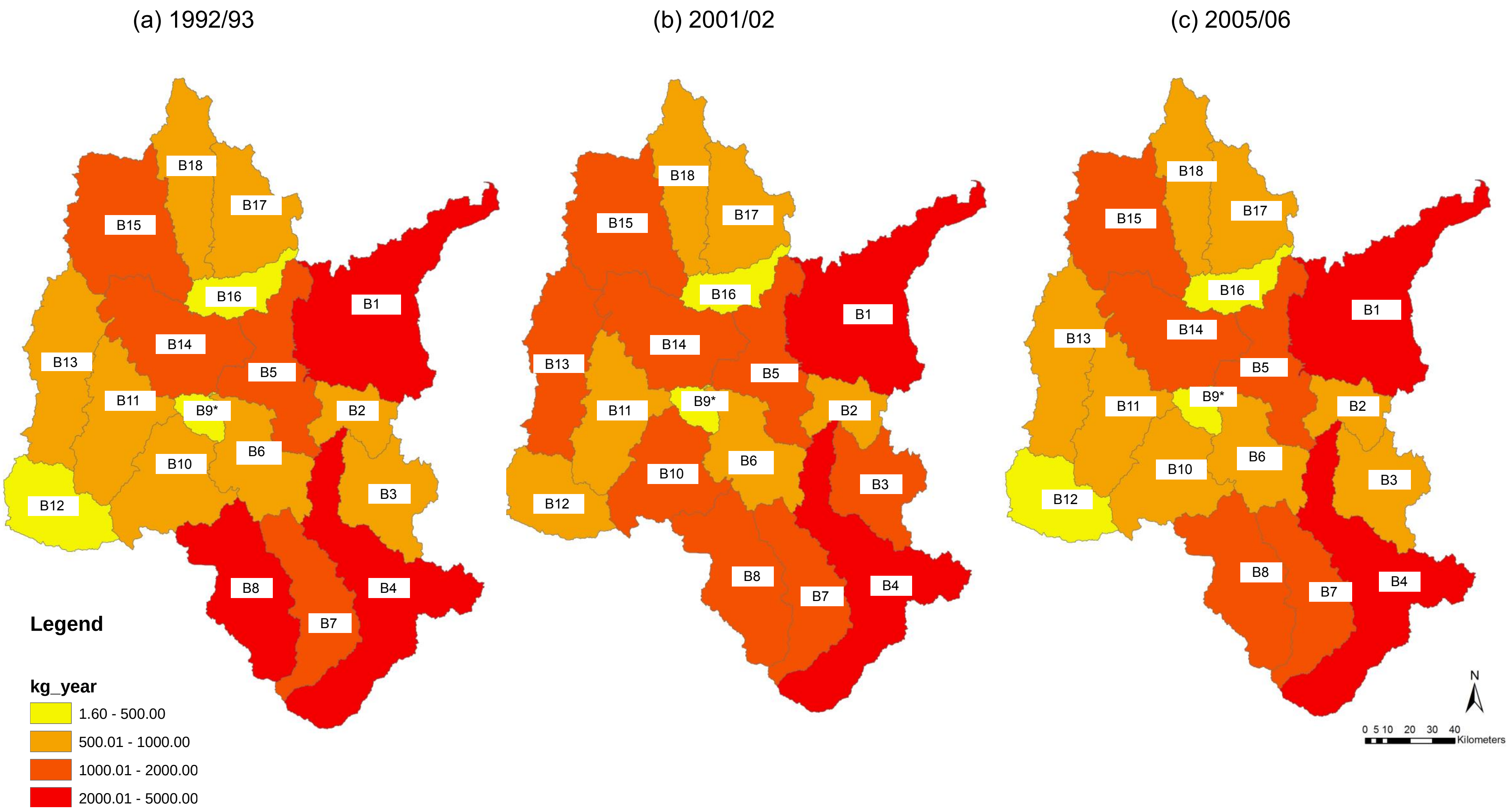


Figure 5.17 Estimated average annual TP loads by sub-catchment (kg/year).

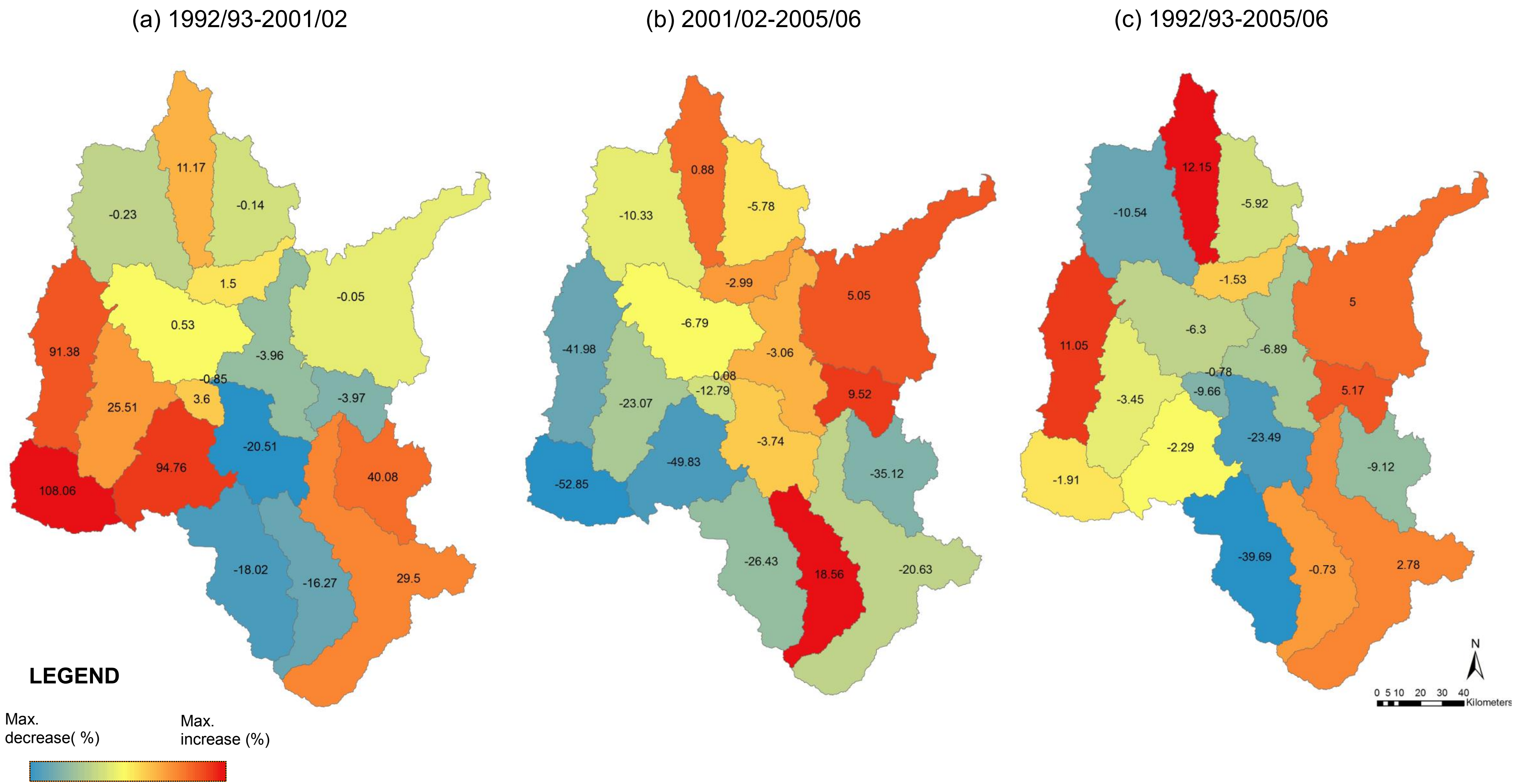


Figure 5.18 Change in estimated average annual TP by sub-catchment (%).

5.3.3 Native vegetation pattern results from *Fragstats*

Clearing of native vegetation is the single most important cause of loss of species and depletion of ecological communities worldwide through habitat loss and fragmentation. To measure the change in the spatial pattern and configuration (structure and composition) of native vegetation communities, *Fragstats* version 4.0 (McGarigal, Cushman & Ene 2012) was employed in this evaluation framework. Three input maps (Figure 5.3) and three Biodiversity Status (BS) maps for 1991, 2001 and 2005 (Figure 5.4) were input to *Fragstats* to compute three class metrics—Class Area (CA), Largest Patch Index (LPI), and Number of patches (NP). These class metrics measure the aggregate properties of the patches of a single class or patch type (i.e. vegetation community, and BS) (McGarigal, Cushman & Ene 2012). Firstly, CA provides an overview of native vegetation by class (i.e. vegetation community and BS for this evaluation). LPI is an indicator of patch size, with values between 0 and 100 to show the percentage (%) of the total landscape that contains the largest patch. To understand the spatial pattern of native vegetation in the catchment, the LPI needs to be interpreted in combination with the number of patches (NP) to better assess whether the landscape process involves habitat loss and/or the possibility of fragmentation. Each index indicates one aspect of fragmentation. For example, a higher NP for a particular vegetation community can indicate a higher rate of disturbance. Nevertheless, information on NP alone does not have any interpretive value because it has no information about area, distribution or shape of the fragments (McGarigal & Marks 1995). For this reason, the NP was calculated together with other metrics such as CA and LPI to enable enhanced interpretation of the data.

Overall, the total area of native vegetation in the catchment decreased from 13,300km^2 (40.1%) in 1991 to 12,800km^2 (38.5%) in 2005 (Table 5.4, Figure 5.19). This result confirmed that no large-scale land clearing had occurred in the catchment since 1991, but gradual loss of small remnant vegetation patches continued over the period.

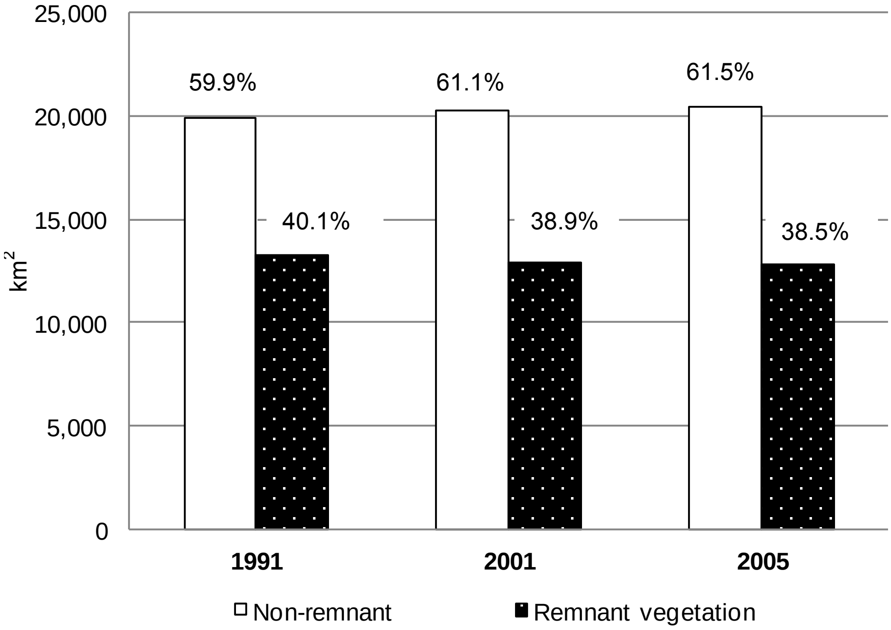


Figure 5.19 The area of change (%) in remnant and non-remnant vegetation in the Burnett River catchment, Australia

Table 5.4 Class Area (CA) and percentage of native vegetation by community in the Burnett River catchment (Unit: ha).

Vegetation community	1991		2001		2005	
Wet forest	74,512	5.6%	70,675	5.5%	70,413	5.5%
Dryland eucalypt forest	1,232,838	92.6%	1,198,508	92.7%	1,185,623	92.6%
Brigalow	7,121	0.5%	5,610	0.4%	5,446	0.4%
Non-eucalypt woodland	7,897	0.6%	7,691	0.6%	7,687	0.6%
Grassland	0	0.0%	422	0.0%	420	0.0%
Coastal communities and wetland	9,386	0.7%	10,317	0.8%	10,288	0.8%
TOTAL	1,331,757	100.0%	1,293,224	100.0%	1,279,879	100.0%

‘Dry eucalypt forest’ was the dominant vegetation community in the Burnett River catchment, accounting for more than 90% of the total native vegetation (Table 5.4). It includes a range of vegetation types, such as woodlands and open-forests, and has as important habitats for small passerine birds[19] many of which are currently in decline (Green

[19] Small passerine birds include Weebill (*Smicrornis breviostris*), White-naped honeyeater (*Melithreptus lunatus*), Buff-rumped thornbill (*Acanthiza reguloides*), Grey fantail (*Rhipidura fuliginosa*), Yellow-faced honeyeater (*Lichenostomus chrysops*), Rufous whistler (*Pachycephala rufiventris*), Striated pardalote (*Pardalotus striatus*), Eastern yellow robin (*Eopsaltria australis*) (Eyre et al., 2009).

& Catterall 1998; Eyre et al. 2009), and native arboreal mammal species, such as gliders[20], possums and macropods (Green & Catterall 1998; Smith & Agnew 2002; Eyre 2006). 'Wet forest' mainly featured as rainforests and accounted for 5.5% of the total native vegetation, with habitat suitable for endangered frog species and rainforest-dependent reptile species (Green & Catterall 1998). The remaining communities accounted for less than 1% each of the total native vegetation.

The change in CA by vegetation community indicated that 3.9% of the total native vegetation area in 1991 was lost by 2005, most of which (-2.9%) occurred between 1991 and 2001 (Figure 5.20). Among native vegetation communities, 'dry eucalypt' and 'brigalow' have been particularly subject to higher levels of grazing, logging and fire disturbance (C. McAlpine 2014, pers. comm., 20 May). Loss of 'dry eucalyptus' was the most significant in terms of scale. However, it was notable that one quarter of the 'brigalow' community was lost between 1991 and 2005, despite its relatively small presence in the catchment (Figure 5.20). The loss of this vegetation community for agriculture since European settlement has been significant in southern Queensland, including in the Brigalow Belt, and has resulted in threats to a number of reptile, bird, and microbat species, which depended on this vegetation community (Australian Government 2013). The analysis by Biodiversity Status (BS) classes also indicated that all remnant vegetation areas declined after 1991 (Figure 5.21). In particular, 10.1% of native vegetation area under 'endangered' status disappeared between 1991 and 2005 (Figure 5.22).

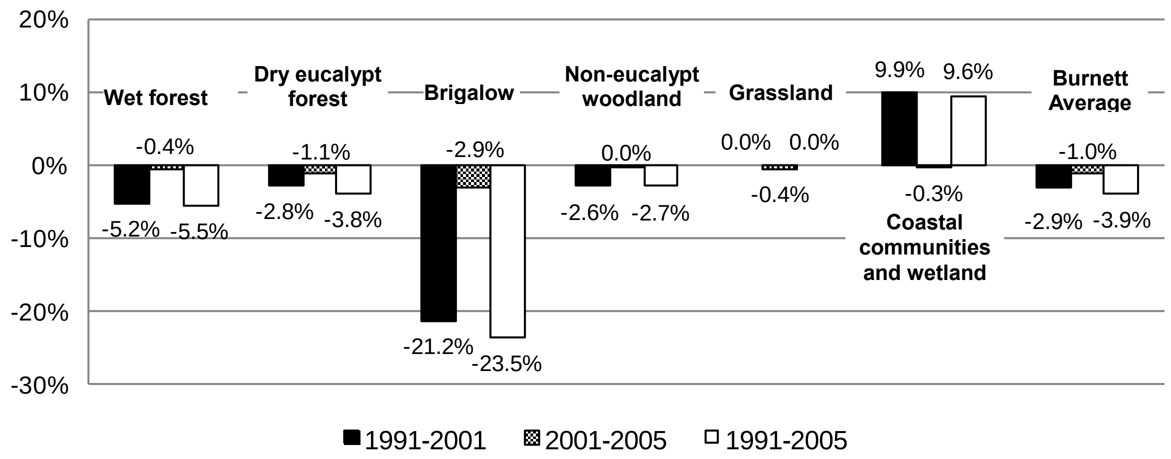


Figure 5.20 Change in Class Area (CA) by vegetation community (%).

[20] Those include Greater glider (*Petauroides volans*), Yellow-bellied glider (*Petaurus australis*) (Eyre, 2006), Feathertail glider (*Acrobates pygmaeus*), Squirrel glider (*Petaurus norfolcensis*), Sugar glider (*Petaurus breviceps*), Yellow-footed marsupial mouse (*Antechinus flavipes*) (Smith & Agnew 2002).

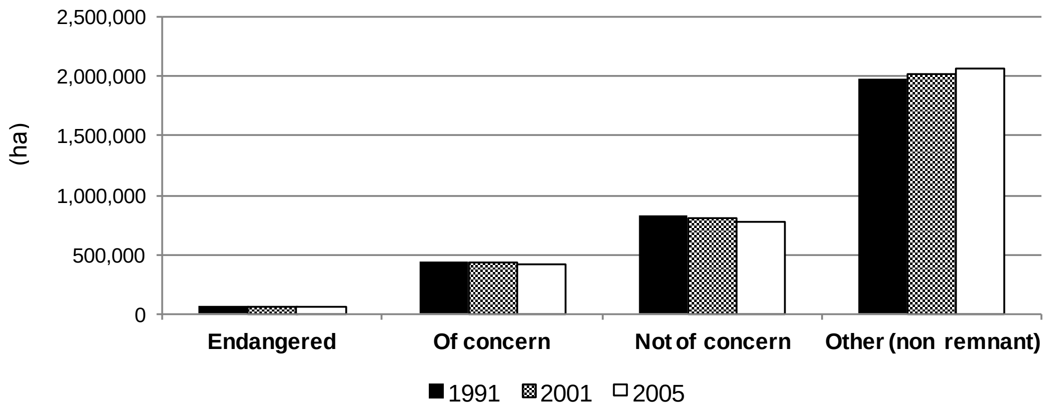


Figure 5.21 Class Area (CA) of remnant vegetation by Biodiversity Status (BS) in the Burnett River catchment.

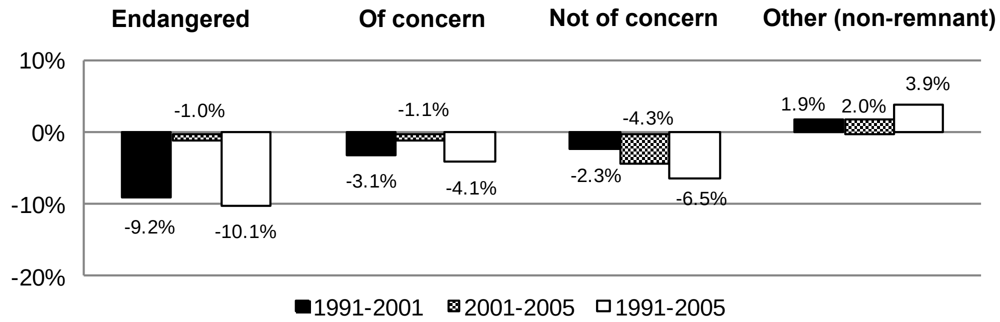


Figure 5.22 Change in native vegetation area by Biodiversity Status (BS) (%).

Vegetation loss rates (ha/year) are often used as an indicator to address trends in the status of biodiversity, and hence they were also calculated from the CA by native vegetation community and by BS class. The average annual loss of native vegetation was 3,850 ha in 1991-2001 and 3,340 ha in 2001-2005. This decline in the loss of most vegetation communities after 2001 was due to the introduction of the *Vegetation Management Act 1999* (Qld). The vegetation loss rates were also expressed in percentages to enable comparisons between vegetation communities (Figure 5.23) and between various BS classes (Figure 5.24) because of the significant difference in CAs particularly among vegetation communities. Among five communities, loss of 'dry eucalypt forest' was the most significant in terms of hectares (3,433ha in 1991-2001 and 3,221ha in 2001-2005) because of its predominance in the catchment. However, 'brigalow' showed the highest vegetation loss rates in percentage throughout the period—2.1% of the annual loss of total vegetation area between 1991 and 2001 (Figure 5.23).

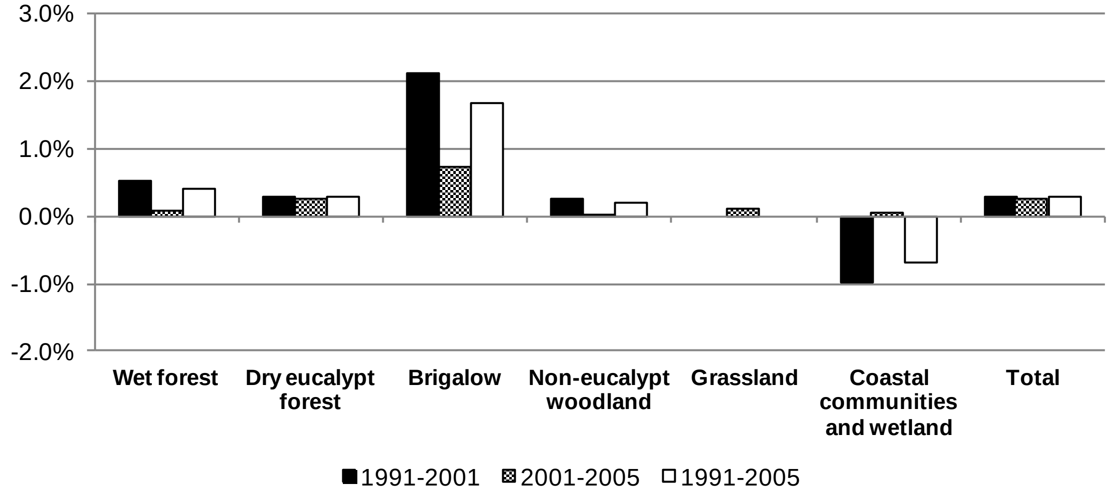


Figure 5.23 Annual rate of vegetation loss of the Burnett River catchment by vegetation community (%).

In addition, the 'endangered', 'of concern' and/or 'not of concern' native vegetation declined over the period despite the introduction of the *Vegetation Management Act 1999* (Qld). After 2001 the loss rate of 'endangered' and 'of concern' areas slowed, while that of 'not of concern' vegetation accelerated (Figure 5.24).

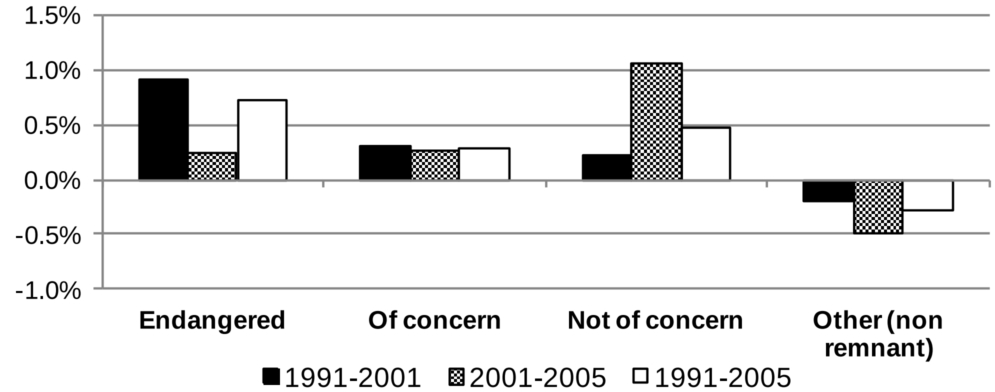


Figure 5.24 Annual rate of vegetation loss in the Burnett River catchment by Biodiversity Status (BS) (%).

For the case study catchment, the LPI and the NP were calculated by *Fragstats* for the five vegetation communities as well as for native vegetation in different BS classes. As both indicators need to be interpreted in combination, the results were presented simultaneously (Figure 5.25 and 5.26). The decline in the size of the largest patches (LPI) was evident in most vegetation communities and BS classes in 1991-2005 at the catchment scale. The NP decreased significantly between 1991 and 2001, and increased slightly in most categories after 2001. Because 'dry eucalypt forest' dominates the catchment's native vegetation, the landscape pattern of this vegetation community affected the overall results (Figure 5.25 [b]). However, the 'brigalow' community showed a slightly different pattern by losing more than one third of its patches between 1991 and 2005 (average -14.1%) with no change in LPI (Figure 5.25 [a]). In terms of the vegetation classed as high conservation status—namely 'endangered' and 'of concern' vegetation—the LPI decreased 1.1% between 1991 and 2001 and stayed the same till 2005, while NP decreased 15.2% between 1991 and 2001, and then slightly increased (0.6%) between 2001 and 2005.

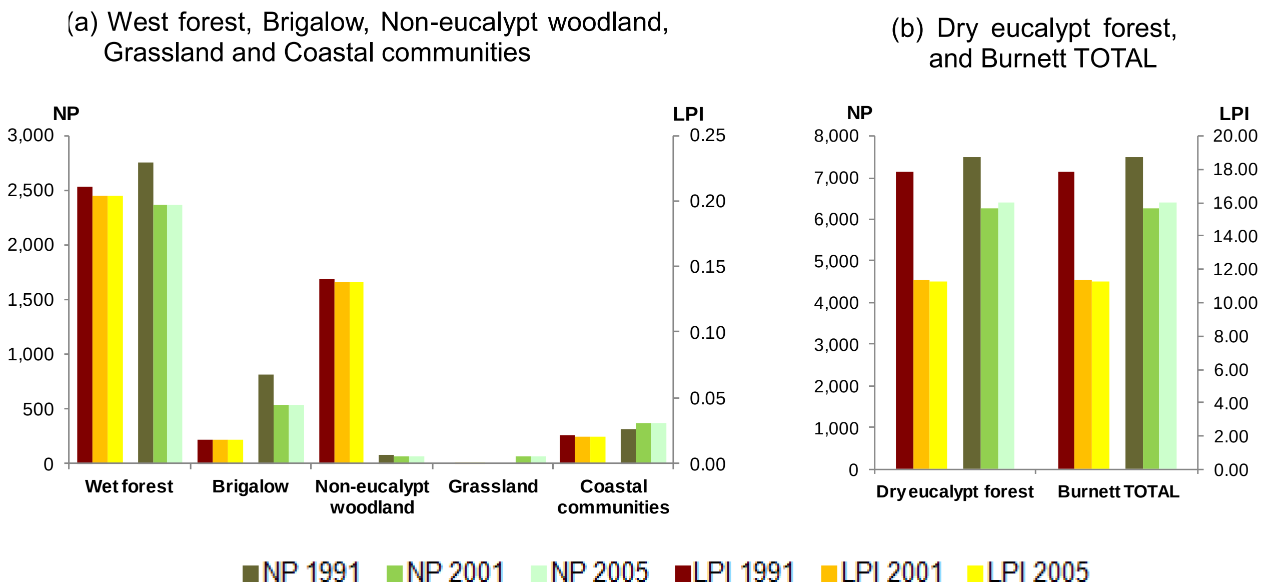


Figure 5.25 LPI and NP of native vegetation in the Burnett River catchment by vegetation community.

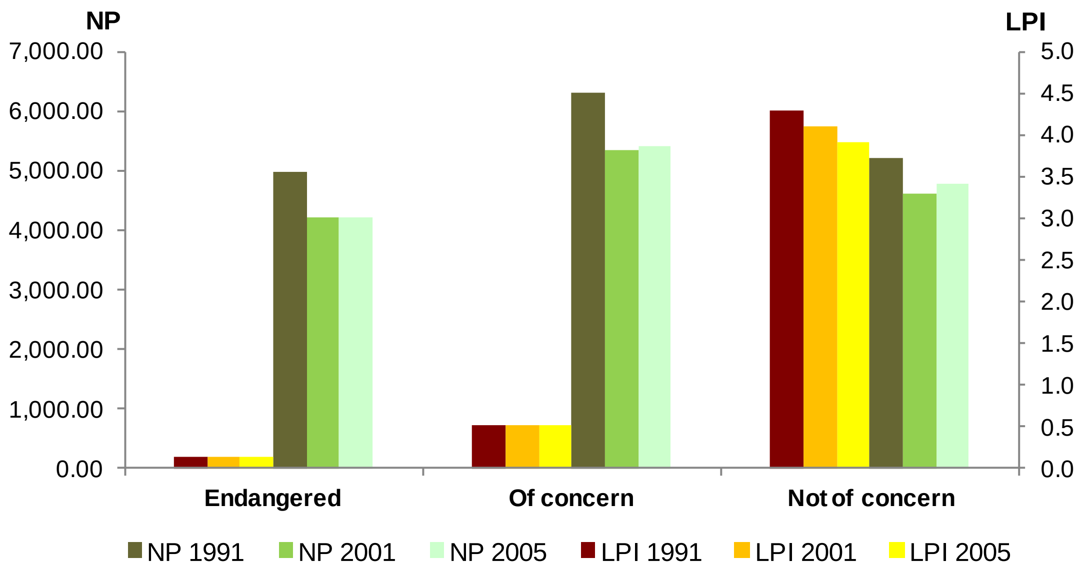


Figure 5.26 LPI and NP of native vegetation in the Burnett River catchment by Biodiversity Status (BS).

The *Fragstats* results were summarised at the catchment scale (Figure 5.27). LPI decreased 36.2% between 1991 and 2005. NP decreased by 15.5% in most vegetation communities between 1991 and 2001 and slightly increased (1.6%) between 2001 and 2005. The reduction of the CA of total remnant vegetation area (3.9%), contributed to a steady loss of habitat in the catchment between 1991 and 2001. The spatial pattern analysis indicated that this resulted from the continuous loss of small patches due to agricultural and urban expansion. Nevertheless, interpretation of a combination of metrics (i.e. CA, NP and LPI) identified that the main landscape process after 2001 was habitat fragmentation, as indicated by a decrease in the amount of habitat (CA) and the size of habitat patches (LPI), and an increase in the number of habitat patches (NP) (Fahrig 2003).

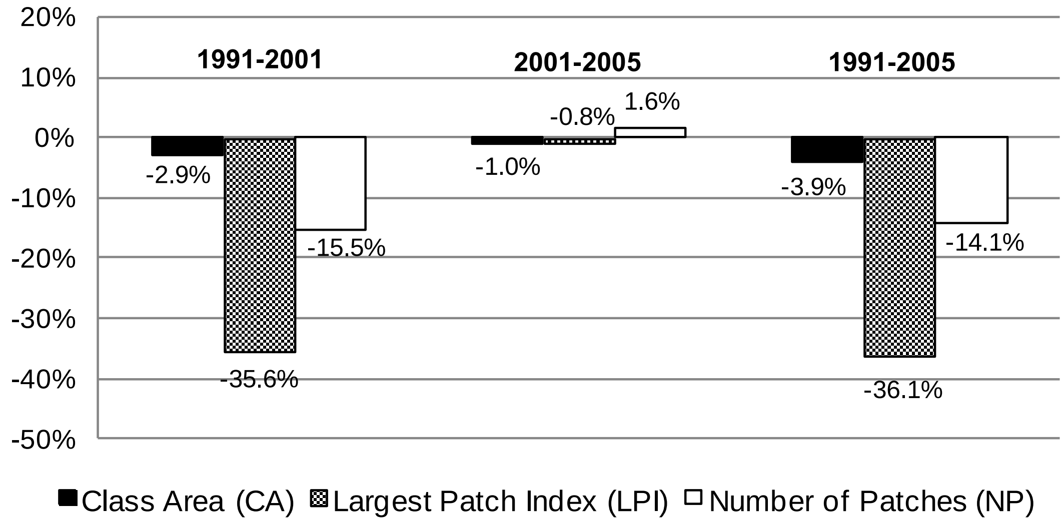


Figure 5.27 Change in CA, LPI and NP in 1991-2001, 2001-2005 and 1991-2005 in the Burnett River catchment.

The results from the sub-catchment scale analysis are visually presented to show the spatial distribution of percentage changes in these indicators. Again, similar trends were observed at the sub-catchment scale (Appendix Figure A5.2). The degree of vegetation loss varied depending on the sub-catchments. After the 1990s, vegetation loss was more significant in inland sub-catchments than coastal ones, as shown in the change in CA and NP (Figure 5.28 [a] and [c]). This reflects that most clearing events were completed by the 1990s in the coastal area for use by agricultural (mainly sugarcane) and urban activities. CA reduced most significantly during 1991-2005 in inland sub-catchments, such as B13 (-9.9%), B7 (-7.2%), B16 (-5.6%), B11 (-4.9%) and B12 (-4.3%) (Figure 5.28 [a]). The decrease in the LPI was most evident in sub-catchments in the Central Burnett region such as B16 (38.3%), B11 (21.9%), B5 (19.3%) and B14 (11.5%) (Figure 5.28 [b]). Compared to these metrics, the fall in the NP of native vegetation communities was even higher. For instance, in B10, 38.4% of native vegetation patches disappeared between 1991 and 2005. This was followed by other inland sub-catchments, B12 (-35.6%), B15 (-33.2%), B9 (-27.3%), B11 (-24.2%), B13 (-23.9%), B6 (-22.1%) and B8 (-20.5%) (Figure 5.28 [c]).

Using the above metrics the landscape processes of individual sub-catchments were examined. For example, around 5% of native vegetation, especially the 'brigalow' community (-32.4%), was lost in B11 over the period, mostly in the 1990s. As both size and number of native vegetation patches reduced, this landscape pattern and associated processes led to habitat loss in B11. B13 is another example where all three indicators showed significant decreases. This sub-catchment presented two different landscape processes before and after 2001. The decline in all three indicators suggested a loss in habitat in the sub-catchment during the 1990s. Between 2001 and 2005 there was evidence of habitat fragmentation: the NP increased (7.1%), while both total native vegetation area (CA) (-4.2%) and patch size (LPI) (-4.9%) decreased within B13 (Appendix Figure A5.2). This indicated that large patches were divided into small pieces to make way for grazing, which expanded sharply by 57% between 2001/02 and 2005/06 in this sub-catchment (Appendix Table A5.5). A similar landscape pattern was observed in other sub-catchments, such as B8, B5, and B14.

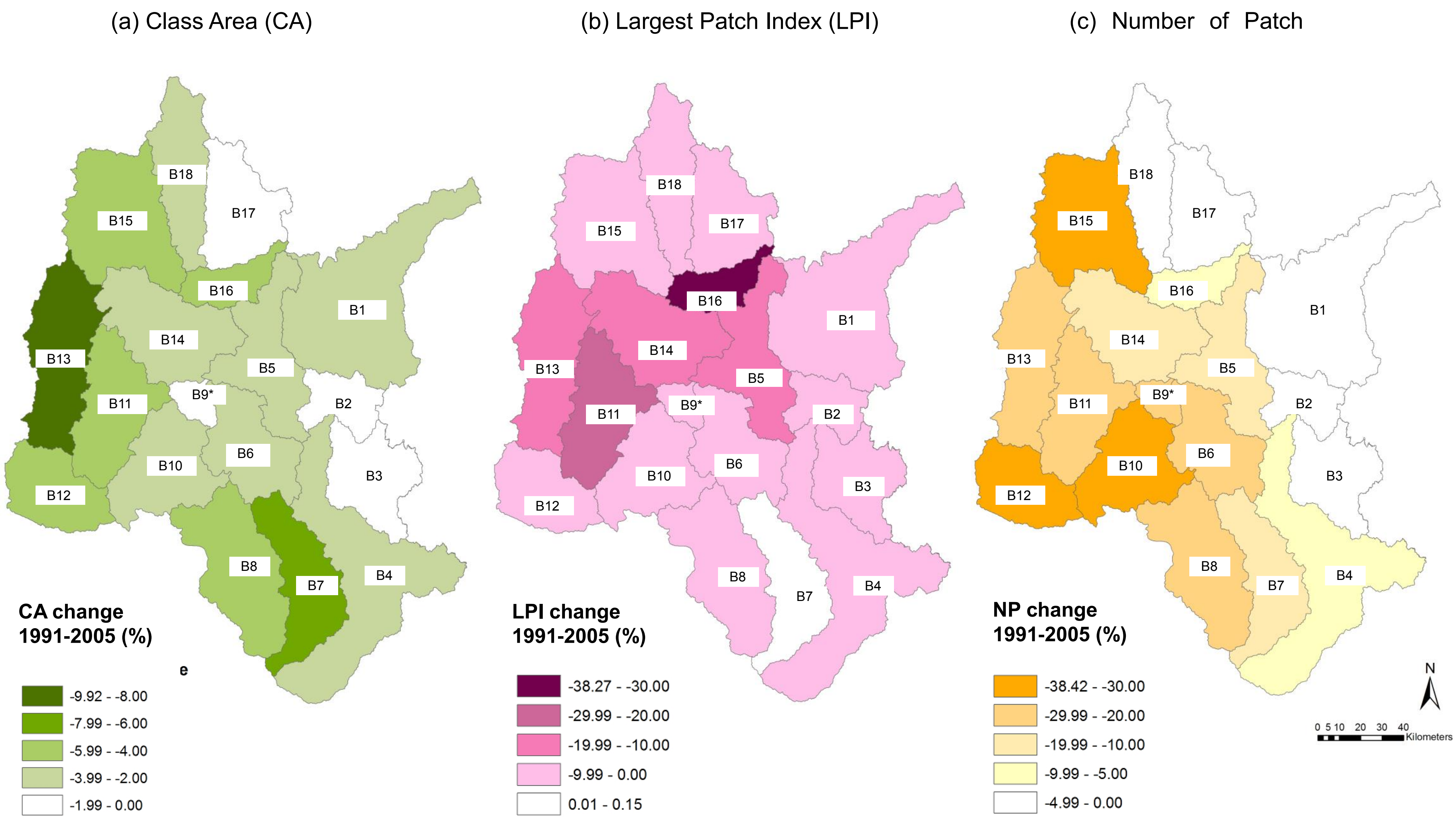


Figure 5.28 Change in class metrics between 1991 and 2005 (%).

5.3.4 Biodiversity conservation value and actual habitat amount

Actual habitat amount (ha) in the Burnett River catchment and 18 sub-catchments was calculated from CAs on three FU/RE maps (Figure 5.3) and biodiversity conservation values corresponding to native vegetation or land use class (on areas disturbed by human activities) (BCV_{LU} [low] and [high]) (Table 5.2) by applying the formula presented in section 5.2.8.

The amount of habitat (ha) was compared over three different time periods. The average actual habitat amount, which was the average value of the results obtained from BCV_{LU} [low] and BCV_{LU} [high], decreased by 3.2 % between 1991 (1,719,720 ha) and 2005 (1,664,236 ha) at the catchment scale (Figure 5.29 and Appendix Table A5.6). During the first 10 years of this period, the amount of habitat dropped 5% in line with the expansion of 'grazing open', 'sugarcane', and 'dryland cropping' areas, which were given low BCV_{LUi}. It showed a slight recovery (1.9%) in the actual habitat amount between 2001 and 2005 (Figure 5.29) despite a slight decrease (1%) in native vegetation area (Figure 5.19). This is attributed to the changes in two non-remnant FU (land cover) classes between 2001 and 2005: increase in 'grazing forested' (29.3%) and a significant decrease in 'dryland cropping' (-66.4%). In this application context, the highest biodiversity conservation value was given to 'grazing forested' (BCV_{LU} [low] = 0.3, BCV_{LU} [high] = 0.7) among non-remnant vegetation areas (Table 5.2), while the lowest value was given to 'dryland cropping' (BCV_{LU} [low] = 0.05, BCV_{LU} [high] = 0.15). This resulted in the overall increase in actual habitat amount between 2001 and 2005.

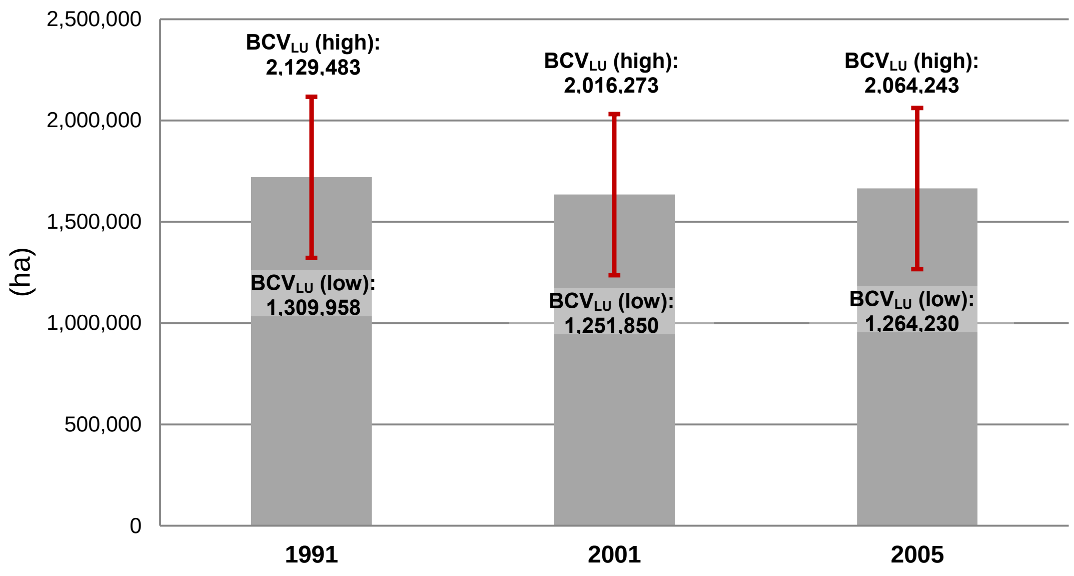


Figure 5.29 Actual habitat amount (average) of the Burnett River catchment.

Nearly all sub-catchments had the least habitat amount (average) in 2001 with a slight recovery between 2001 and 2005 (Figure 5.30). Both results calculated from both BCV_{LU} [low] and BCV_{LU} [high] showed the same patterns. The exceptions were B1, B2 and B13, which showed a continuous decrease in actual habitat amount from 1991 to 2005, and B8, that showed a slight increase (0.6%). The decrease in land with high value vegetation and the increase in 'grazing open', 'irrigated cropping' and 'urban' land use on coastal area were identified as the main causes of the decline of actual habitat amount in B1. Similarly, the loss of high value vegetation accompanied by the expansion of cropping areas led to the decrease in the actual habitat amount in B2. B8 also experienced a continuous loss of native vegetation, while experiencing an increase of land use with relatively high biodiversity conservation values, such as 'conservation' (with non-remnant vegetation) and 'grazing forested'. B13 had the most significant native vegetation loss and fragmentation. This was associated with the expansion of grazing within the sub-catchment. Between 1991 and 2005, B13 showed the largest loss of actual habitat amount (average) with 6.8%, followed by B15 and B1 with losses of 5.7% and 5.2% (Figure 5.31 and Appendix Table A5.7).

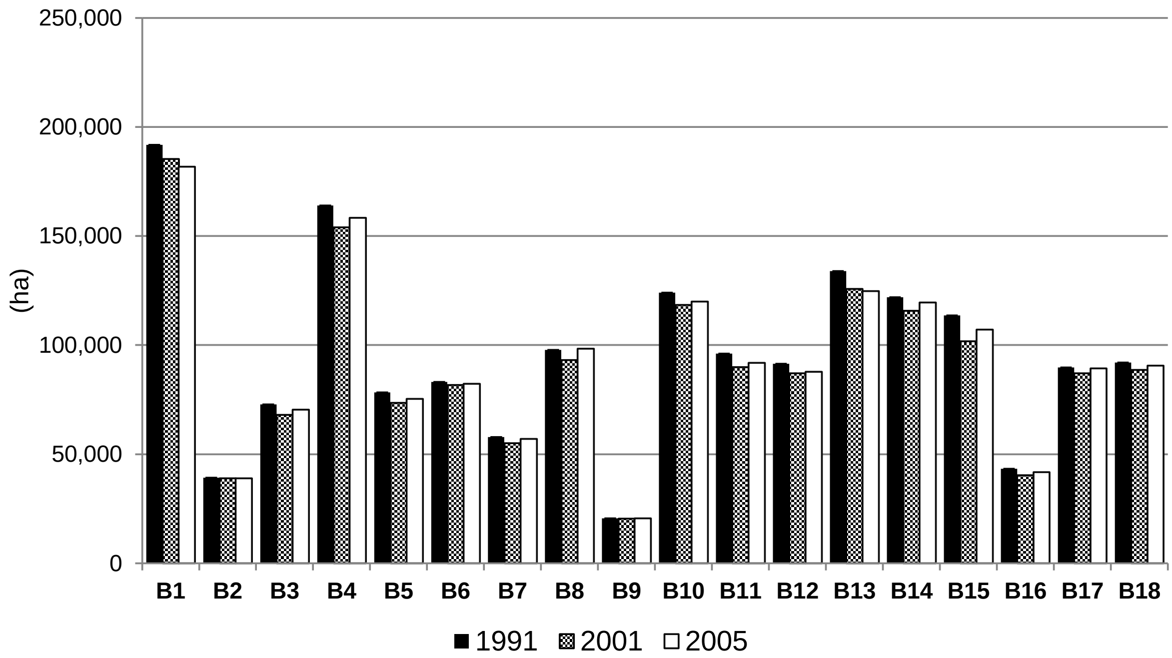


Figure 5.30 Actual habitat amount (average) in the Burnett River sub-catchments.

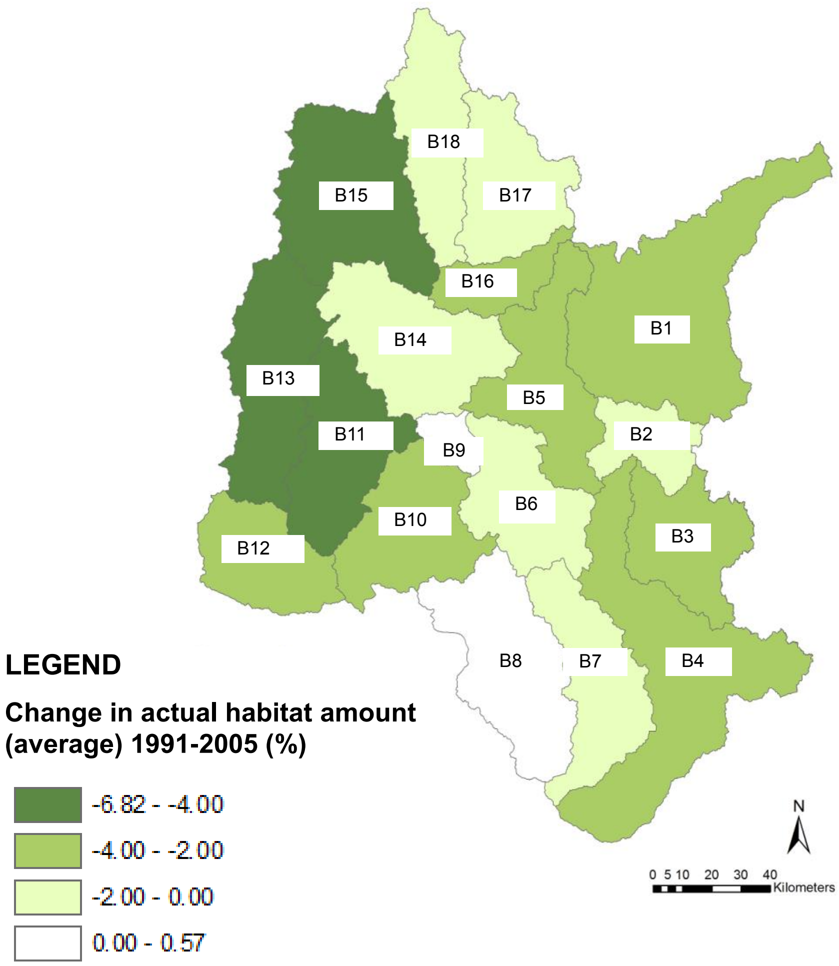


Figure 5.31 Change in actual habitat amount (average) in Burnett River sub-catchments between 1991 and 2005 (%)

5.3.5 Total results for the Burnett River catchment

One of the main objectives of this research was to develop an evaluation framework that assembles key regional-scale environmental indicators closely associated with land use change, and to enable regional policy makers to access this information about the potential environmental impacts of proposed land use change scenarios, especially those using underutilised agricultural land for future bioenergy crop production (Chapter 6). The potential users include policy/decision makers and consultants responsible for environmental, land use and natural resource management policy and planning on a regional scale. Having such an evaluation framework would allow policy makers to better understand and predict the impacts on key environmental values, to guide sustainable land use change options. Radar charts (Figure 5.32) were chosen as an effective way to communicate the results, as they simultaneously presents the results of the multiple indicators. Radar charts offer a graphical representation of the evaluation that is both clear and condensed, allowing the policy makers to easily identify the environmental performance of each scenario from different indicators in a uniform and coherent way (Langeveld et al. 2012).

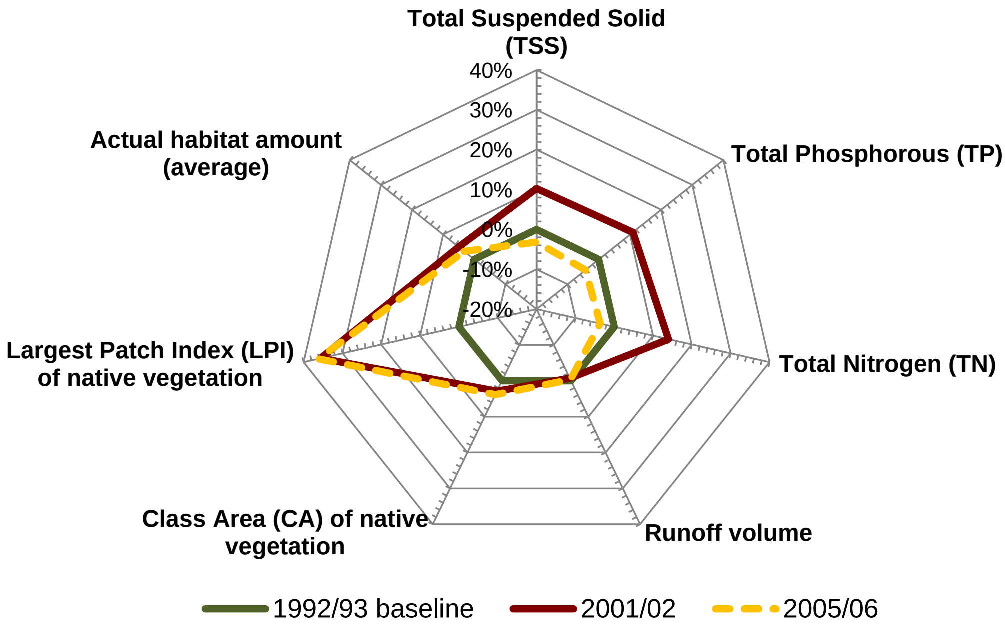


Figure 5.32 Overall environmental outcomes of land use scenarios in the Burnett River catchment: 1992/93, 2001/02 and 2005/06.

A baseline scenario was used as a reference for comparison against different scenarios. In this application, the 1992/93 FU map and/or 1991 RE map were used as the baseline scenario to evaluate whether changes in the overall environmental quality in 2001/02 (and/or 2001) and 2005/06 (and/or 2005) were positive or negative. The chart was generated to make comparisons (i) between the 1992/93 and 2001/02 maps; and (ii) between the 1992/93 and 2005/06 maps (Figure 5.3 and 5.4) using a percentage (%) improvement or a % decline of each indicator (Figure 5.32). Thus results from a reference scenario were set as zero percent (0 %) in all indicators. In the application to future land use change scenarios (Chapter 6), the results of 2005/06 will be used as the baseline scenario.

The area within the graph and the shape of the graph indicate the overall environmental performance. A graph with a larger area indicates a scenario with poor overall environmental performance, while a graph with a smaller area indicates a scenario with better overall environmental performance. For water indicators such as runoff volume, TSS, TN and TP, higher values indicate poorer environmental performance (i.e. more is worse), so for these indicators relative performance was calculated as [indicator *i* value (%) * 1]. For biodiversity indicators, higher values indicate better environmental performance (i.e. more is better), so for these indicators relative performance was calculated as [indicator *i* value (%) * -1]. NP was excluded from the radar chart as it requires interpretation with other metrics such as CA and LPI.

In summary, this chapter aimed to test the effectiveness of the environmental evaluation framework in relation to past land use and environmental trends during the period from the early 1990s to mid-2000s. Overall, the results provided a useful overview of changes in environmental qualities (land use, vegetation, water quality and biodiversity) associated with land use change in the region during the assessed period (Figure 5.32) (Chapter 4). In the case study region, a number of federal and state government environmental and conservation initiatives took effect in the 1990s to early 2000s, including the Statewide Landcover and Trees Study (SLATS); the *Vegetation Management Act 1999*, the Reef Water Quality Protection Plan (Reef Plan), and recovery plans for threatened species under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act). Hence the early 2000s was regarded as a critical turning point for the case study region in terms of curbing environmental degradation.

The results in this chapter captured this trend, illustrating that environmental quality declined between 1992/93 and 2001/02 in the Burnett River catchment (Figure 5.32). The reduction in CA and LPI was the result of the vegetation removal for agricultural expansion, which also increased sediment and nutrient loads, particularly nitrogen, in the water ways. The areas of 'sugarcane' and 'dryland cropping' peaked in 2002/03 and this produced larger fluxes of these pollutants. While the loss of native vegetation decelerated after 2001, habitat fragmentation occurred in sub-catchments, as indicated in the findings from the analysis using *Fragstats*. However, the lower rate of vegetation loss must have contributed to the improvement in the water quality indicators, from the baseline scenario of 1992/93 to 2005/06 as native vegetation cover is an important factor that can affect water quality, especially sediment loads (Figure 3.1). The area under 'grazing' land use across the catchment and 'urban' land uses in certain sub-catchments continuously increased over the period. However, the sharp reduction or disappearance of cropping areas (especially 'dryland cropping') after 2001 may have contributed to the reduction in sediment and nutrient loads in the river. There was evidence of the conversion from cropping to grazing land in recent years on a catchment scale due to a number of reasons (Chapter 4). This also led to the reduction in TP and TN loads because grazing land requires less input of fertiliser than cropping land.

The results of the two scenarios in 2001/02 and 2005/06 show poorer overall environmental outcomes in comparison with the baseline scenario, as seen in the reduction in the graph areas. In conclusion, the Burnett River catchment has experienced environmental degradation in soil, water and biodiversity indicators since early 1990s, although the degradation rate significantly reduced after 2001.

5.4. Discussion

The evaluation framework was developed with a set of generic indicators to enable wider application and thus facilitate sustainable land use decisions in relation to future bioenergy crop production. The potential users include policy/decision makers and consultants in the areas of regional environmental, land use and natural resource management policy and planning. As a result of this application experience, a number of strengths and limitations of the evaluation framework were identified. For this purpose, key criteria were used to discuss the overall effectiveness of the evaluation framework (Table 5.5).

Table 5.5 Key criteria for testing the overall performance of the evaluation framework

Key components	Key criteria
Indicators	• appropriateness • comprehensiveness • versatility • simplicity
Models/tools	• availability of input data and parameters • accessibility to the models/tools • complexity or simplicity • spatial scale

5.4.1 Appropriateness and comprehensiveness of indicators

The key indicators applied in the Burnett River catchment focused on water quantity and quality and biodiversity, because the literature review suggested that these elements can be affected substantially by bioenergy-driven land use change at a regional scale (Chapter 3).

However, it is also important to emphasise that the indicators could be re-selected to assess particular situations in other locations due to the different local concerns and purposes, the varied characteristics of bioenergy systems, the range of stakeholders and their priorities, diverse regional environments, and differing scales of application (Efroymson et al. 2013). Although the results cover important regional-scale environmental issues associated with land use changes in the case study region (i.e. applicable not only for bioenergy-driven land use change but also land use change in general), the scope of indicators could be expanded to include GHG (e.g. evaluation for peatland) and soil quality in future applications.

However, most indicators included in this evaluation framework for the Burnett River catchment are essential for any case study region. The water quality indicators, namely TSS, TN and TP are the most widely used indicators for today's global water quality and ecological studies as measuring the loads of major nonpoint source pollutants exported to downstream environments (e.g. GBR catchments). They are greatly influenced by land use change, native vegetation loss, and the intensity of human activities on the catchment, and they were highly advantageous in representing impacts from human-induced land use changes. As they are commonly used indicators, data availability is one of the advantages. Compared to these water quality indicators, results on runoff volume did not highlight the particular characteristics in the case study catchment, but that is most likely to be caused by (i) setting of parameters of Simhyd (rainfall-runoff model) incorporated into *Source*, (ii) the catchment size, and (iii) the minor scale of land use change between the three scenarios. The limitation associated with Simhyd parameters will be discussed below. Nevertheless, indicators for water quality can be tailored for different catchments. For example, herbicide runoff from agricultural activities is an issue in the GBR catchments, and pesticide is an issue in many catchments. Such relevant indicators can be added to the framework depending on future needs. *Source* is fully capable of simulating a range of indicators related to water quantity and quality (e.g. Particle Phosphorous (PP), dissolved inorganic nitrogen (DIN), and photosystem II herbicides (PSII)) as long as there is water quality data that enable model calibration and validation.

In terms of landscape pattern analysis, indicators/metrics were selected to measure habitat loss and fragmentation. These are the two biggest challenges in biodiversity and conservation worldwide (Fahrig 2003). The metrics were selected based on their 'simplicity', 'versatility' and 'comprehensiveness'. CA is one of the most effective metrics at class level to indicate status and change in total area by a particular patch type. In this application, habitat loss was relatively well represented by a decrease in a couple of area metrics, such as CA (total area) and LPI (patch size). However, there were limitations to representing habitat fragmentation using only a few metrics, as it is a much more complex process that involves size, shape, distribution, and connectivity of patches (Ewers & Didham 2006). In landscape ecology research, a range of metrics is commonly used to measure fragmentation, which include edge metrics (e.g. edge density [ED]), contagion/interspersion metrics (e.g. contagion [CONTAG] and mass fractal dimension [MFRAC]), isolation/proximity metrics (e.g. mean nearest neighbourhood distance, proximity index [PROX]), and shape metrics (e.g. perimeter-area fractal dimension [PAFRAC]) (Hargis, Bissonette & David 1998). They need to be interpreted in combination. Considering the potential target users, the number of indicators was minimised to avoid the complexity associated with the application and interpretation of the outputs. More precise and possibly simplified identification of habitat fragmentation is an interesting area for future research.

In addition, biodiversity conservation value and actual habitat amount were introduced for efficiency and effectiveness in quantifying the biodiversity status of the total case study area. They have the strong advantage of being able to provide a good overview of the status and changes in the biodiversity conservation value of the area. On the other hand, limitations were also recognised. Firstly the subjectiveness of the biodiversity conservation values of different land uses/land covers (BCV_{Lu}) requires that the values must be established carefully in consultation with experts who are familiar with local ecology. Also, there is always strong criticism against 'over-simplification' of the complexity of biodiversity and ecosystems status in such scoring systems. Regardless of the introduction of value ranges in BCV_{LU}, the same range of value was allocated to the same land use class in the same catchment, even though in reality the species abundance and diversity could be very different depending on a number of site conditions attributed to climate, topography, location, surrounding environment/adjacent land use, vegetation type, original species composition, and agricultural practice and management. The values were not intended to highlight individual species, yet in reality it often happens that some land use (e.g. plantation forestry of dry Eucalypts in the subtropics) could support certain taxa (e.g. arboreal mammal, bats) and species, but not be beneficial to the others (e.g. reptiles, frogs). Thus consultation with local experts is essential to minimise or avoid inappropriate biodiversity conservation values. It is also important to note that the selection and interpretation of biodiversity indicators is regionally specific because the local taxa that are of special concern, such as key species, vary by region (McBride et al. 2011; Efroymson et al. 2013).

5.4.2 Data availability for models/tools

This evaluation framework aims to cater for future application to other geographical regions with less data intensity and availability (3.1). Thus the availability of input data required by *Source* and *Fragstats* is important to its wider application. In short, the evaluation framework cannot function without input data. *Source* and *Fragstats* both require spatial information, primarily land use/land cover and vegetation datasets. For the selected case study catchment in Queensland, Australia, the finer scale spatial dataset based on Landsat imagery was developed by State Government departments and were publicly available without cost. The only limitation was availability of data prior to the 1990s, a problem common in most other areas of the world. All state-level mapping programs based on fine resolution satellite imagery started in Queensland in the late 1990s. Hence it was impossible to conduct the analysis of land use mapping prior to the 1990s. The earlier dataset released by the Australian Government in the early 1990s was used for this application for a few reasons (section 5.2.2), but the resolution of national-scale data is much coarser than that of state or catchment scale datasets.

Thanks to recent developments in satellite-based technology, land-cover and vegetation datasets, and digital elevation models (DEM) are readily available for most areas in the world. These include Global Land Cover 2000 (GLC2000) database (European Commission Joint Research Centre 2003) and the Shuttle Radar Topographic Mission (SRTM) data for DEM (NASA Jet Propulsion Laboratory 2000). However, data availability at finer scales for land use and vegetation mapping depends on geographical location. Spatial datasets are well developed at government levels in many developed countries. For example, one stop GIS portals are provided by government websites in the U.S.A to allow free data download. This includes the U.S. Department of Agriculture (USDA) website at the federal level, and New Jersey, North Dakota, Oklahoma, Texas, and Georgia at state levels. Iowa State also provides a one stop source for state-wide environmental and natural resources data, including watershed and land use/land cover (since 1975) (Iowa Department of Natural Resources 2013).

In Europe, Land Use Data Centre (LUDC) under European Environmental Agency (EEA) offers 100m resolution raster land-cover data, Corine land cover (CLC) inventories for 1990, 2000 and 2006 (European Environment Agency 2013). This free dataset indicates 44 land-cover classifications including vegetation types. At the same time, individual EU member states prepare data for local uses. For example, the UK's Centre for Ecology and Hydrology (CEH) under the Natural Environmental Research Council, produces high resolution land cover maps for 1990, 2000 and 2007 for the UK, although the European CLC maps were actually produced based on them (Centre for Ecology & Hydrology 2013). Japan is another country with readily available datasets on urban, transport and land use in GIS format. A 100m resolution land use dataset has been available since 1976 from the website of the National Land Information Division, National and Regional Policy Bureau under Ministry of Land, Infrastructure, Transport and Tourism (MLIT) (国土交通省国土政策局国土整備課 2013).

It is more challenging to obtain localised datasets in developing regions. Brazil and Indonesia have experienced the highest deforestation rates among tropical countries (FAO 2006, 2010), and thus developing high-quality land cover and land use datasets is an urgent matter in these countries. In Brazil, coarse scale land use datasets for 1996 and 2010 (1:5,000,000) were produced by a national agency, Instituto Brasileiro de Geografia e Estatística (IBGE), for the entire the country (IBGE 2003, 2010b), and the shapefiles are available from the IBGE's website. Finer catchment-scale land use maps exist only for a few selected catchments for specific purposes. For example, a detailed land use map (1:100,000) was prepared for four sub-catchments in the São Francisco River Basin in north-eastern Brazil under an international development project led by the Organisation of American States (OAS) and the United Nations Environment Programme (UNEP) —the Integrated Management of Land-based activities in the Sâo Francisco River Basin project (Companhia de Desenvolvimento dos Vales do São Francisco 2002; MMA 2006). However, this is not applicable to other catchments in the country. A vegetation cover map for 2006 was also produced by IBGE at a national scale (1:5,000,000) using six vegetation cover categories and 25 sub-categories (IBGE 2010a). While the dataset is also available on the same website, Brazil is a large country and the dataset is too coarse for a catchment-scale evaluation.

In Indonesia, obtaining land use mapping for multiple time periods is even more challenging. There are web-based digital national scale land cover and land use maps generated by the Ministry of Forestry (MoF 2010). These online digital maps include catchment boundaries, and could provide a basis for input data for the evaluation, and the original data may be available from the government agency. Nevertheless there is controversy concerning this dataset due to the unclear definitions and boundaries of forest areas and a lack of fair procedures in designing forest areas (Ardiansyah & Barano 2012; Margono et al. 2012). Agreed land cover maps have been required by different sectors in accordance with the objectives of the United Nations Framework on Climate Change (UNFCCC) Reducing Emission from Deforestation and Forest Degradation (REDD and REDD+) program (Ardiansyah & Barano 2012; Margono et al. 2012). In other developing regions where getting access to high quality land cover and/or land use datasets is difficult, the use of the global scale datasets such as GLC2000 for the baseline scenario could be an excellent alternative. For example, the dataset for Africa and South America (1,000×1,000 metre resolution) dramatically upgraded the previous maps due to the increased availability of higher quality satellite data. GLC2000 for Africa provides 27 land-cover and/or vegetation classifications on a continental scale (Mayaux et al. 2003), and that for South America provides 42 land-cover and/or vegetation classes on a continental scale (Eva et al. 2002). The classification is suitable for both hydrological and spatial pattern analysis in this evaluation framework.

It is also important to be aware of uncertainty in relation to the input data used in this evaluation framework. In spatially explicit modelling approaches based on GIS, there is always uncertainty in the use of input data in different data types, such as format and resolution, which significantly influence the outputs (Verstegen et al. 2012). A difference in mapping methodologies and resolution resulted in slight differences in land use classes in the case study catchment between those by the Australian Government (Land Use of Australia, *version 3*) (1:2,500,000) (ABARES & BRS 2006) and by Queensland Government (the Queensland Land Use Mapping Program [QLUMP] (DSITIA 2012g) (4.3.3.3). In this evaluation, the national scale land use maps with coarse resolution (0.01×0.01 degree, or approximately 1,000×1,000 metres) were used due to the availability of different time periods. From the national scale land use maps, FU maps (Figure 5.2) were generated for *Source* input, and then they were overlayed with Queensland vegetation maps (RE maps) (vector format, which was converted into raster) to produce FU/RE maps (Figure 5.3) for *Fragstats* input. For example, due to different format and resolution of these datasets, this data processing may cause errors or reduce the quality of the original data without careful understanding the process. In general, the use of data sets from different sources can lead to greater risk of uncertainty (by both software and human). The uncertainty associated with parameters is further discussed in the next section.

5.4.3 Strengths and limitations of models and tools

Fragstats is a simple tool for users. Its versatility has been proven by numerous past applications globally. The only input required for the tool was a land-cover map or vegetation map for this evaluation (i.e. FU/RE maps). On the other hand, *Source* requires many more inputs than *Fragstats* to simulate more complex hydrological events. The input data includes gridded climate data for the case study region for time periods in excess of a decade (longer period data is preferable to average out the influence of climate variability). Acquiring this data is a challenge in many parts of the world. In the case of Australia, the gridded climate data has been relatively well managed over the past decades by the Bureau of Meteorology (BoM). However, limitations could occur at inland catchments in large countries like Australia where the density of population and weather stations is significantly low. The low spatial density of climate data for the larger catchments could increase the uncertainty in the estimates.

More importantly, obtaining high quality parameters for the rainfall-runoff model and constituent generation model is the most challenging process of running *Source*, as they directly influence the model outputs. One of the strong advantages of this modelling platform (Chapter 3) was that it has less complexity and fewer input requirements compared to other equivalent spatial hydrological models (e.g. SWAT, HSPF). This application received support from Queensland Government hydrologists working within the Paddock to Reef program in model development and calibration. The parameters for these models had been tested against observed data and finer scale modelling conducted in the program. However, the availability of high quality parameters and the associated uncertainty are the most challenging issues in relation to a wider application of *Source* to other parts of the world, particularly where limited water quality monitoring data exist and limited hydrological expertise is available. However, the application of this model outside Australia is in progress, such as its recent applications to Singapore and Lake Tai in China (eWater CRC 2012a), and a recent technical and partnership agreement with the Indian government for future development and application of *Source* in the Indian context (eWater CRC 2013). Such recent efforts and progress in relation to *Source* are in harmony with the aim of this study, and significantly enhance the chance of future applications of this evaluation framework.

To limit the number of parameters, the simplest catchment model was developed within the *Source* platform for this application (5.2.3). The limitations of this simplified model need to be acknowledged. For example, the 11 FU classes were grouped into three for Simhyd parameters, namely forested land ('conservation', 'forestry' and 'grazing forested'), cleared land ('grazing open', 'urban', and 'others') and cropping land. This was because very little scientific data was available to justify applying different parameters to each FU class (Fentie in press). As a result, runoff volume highlighted few characteristics associated with land use change in the case study catchment. For the constituent generation model (EMC/DWC), it should also be noted that there is very limited data available on EMC/DWC for other catchments. For this application, 11 values corresponding to each FU class were derived and used. Consequently the changes in pollutant loads were primarily driven by land use changes, as opposed to changes in hydrology. However, it is important to emphasise that obtaining the most accurate runoff volume and pollutants loads was not the prime objective of this evaluation framework or this research project. The main objective was to develop a platform to inform policy makers on the general environmental outcomes of various land use change scenarios to assist their decision making processes. Therefore, the models and tools incorporated into the framework had to be relatively simple and applicable for many cases. The results for loads generated had a fair degree of accuracy, considering that Simhyd parameters and EMC/DWC values applied to the model were extracted from the more advanced, calibrated model developed by the Queensland hydrologists for the Reef Plan (5.2.3). However, it is important to emphasise that the goal in relation to Reef Plan was distinct and different from the objective of this research project.

Lastly, both *Source* and *Fragstats* were easily accessible. The latest version of *Fragstats* could be downloaded from the website without cost (McGarigal, Cushman & Ene 2012). Detailed user guides and tutorial materials were also available from the website. *Source* (formerly *Source Catchments*) was initially offered as free software. However, the latest version requires membership and a small licence fee depending on organisation type (eWater CRC 2012c). The membership offers full support, training and resources for the users.

5.4.4 Appropriate spatial scale

In this application, the evaluation framework was applied at catchment and sub-catchment scales. The results indicated repetition of the same pattern at both scales for some indicators (e.g. landscape structure and change indicators). Even though the Burnett River catchment has differences in local climates and land use characteristics between the coastal area (i.e. sub-catchment B1) and inland areas, it can be concluded that most areas in the catchment experienced similar land use change patterns and environmental consequences between 1991 and 2005. To simplify matters in future studies, the catchment scale application may be sufficient for regions where little climate and land use variety are observed, since the application to the sub-catchment scale may require substantial time and effort (especially when the number of sub-catchments is large). However, all results from the sub-catchment scale analysis in this application were extremely valuable in helping to understand the variety of patterns within the catchment and this data often aided understanding of the catchment scale analysis by providing more detailed information. More importantly, the application to the sub-catchment scale is essential when evaluating consequences of the land use change scenarios that are likely to affect sub-catchments in various ways.

5.4.5 Overall performance

In conclusion, the largest strength of the evaluation framework was its effectiveness in providing a good overview of changes in environmental quality associated with land use changes. Results from the application captured the main characteristics of the environmental quality of each land use scenario relatively well. This argument was supported by comparison of results with existing quantitative and qualitative data, such as existing water quality monitoring data and information from local experts. The other strengths included flexibility that will allow for future development, and its logical structure. Its flexibility enables the framework to be customised for future applications (e.g. re-selection of indicators), which take account of particular local circumstances.

Opportunities for improvement and future development were identified from the application, such as the limitations of the indicators for habitat fragmentation, and the challenges associated with the availability of data and parameters where it is difficult to get access to the knowledge of experts.

5.5. Summary

This chapter covered the testing of the evaluation framework in the Burnett River catchment in Queensland, Australia using existing land use datasets 1992/93, 2001/02 and 2005/06. As a result, eight environmental indicators—runoff volume; total suspended solids (TSS); total nitrogen (TN); total phosphorous (TP); class area (CA), largest patch index (LPI), number of patches (NP) of native vegetation, and biodiversity conservation value (or 'actual habitat amount') —were quantified for 1992/93, 2001/02 and 2005/06 maps.

The main input data required by the spatial hydrological model, *Source* (ver. 3.2.3 beta), and the spatial pattern analysis program, *Fragstats* (version 4.0), were land use and vegetation maps of the area. Thus the original maps were processed into Function Unit (FU) maps based on hydrological responses, making them suitable forms for model inputs. This process included reclassification of the map categories and conversion to raster format using ArcGIS. *Source* also required parameters for its incorporated rainfall-runoff model and constituent model. For this application, the parameters were adapted based on a finer calibrated model also prepared within the *Source* platform by Queensland Government hydrologists. The team had extensive expertise and experience in hydrological modelling of the Great Barrier Reef (GBR) catchments under the Paddock to Reef program.

The results from the evaluation framework provided a useful overview of changes in environmental quality associated with land use change in the region from the 1990s to mid-2000s. While these changes were identified in Chapter 4 based on literature (e.g. water quality, terrestrial biodiversity), existing data (e.g. remnant vegetation, land use) and interviews with local stakeholders, the results from the application of the evaluation framework to this region, showed close alignment with the reported trends in land use, vegetation, water quality and biodiversity over this period. The results captured that the Burnett River catchment experienced environmental degradation from the early 1990s, although the rate of degradation significantly reduced after 2001/02. The 1992/93 FU map was used as a baseline scenario to evaluate the change in the overall environmental quality in 2001/02 and 2005/06. Between 1992/93 and 2001/02, the reduction of CA and LPI was the result of vegetation removal for agricultural expansion in the catchment (especially grazing and dryland cropping), which also increased TSS, TN and TP in the waterways. The loss of native vegetation decelerated after 2001/02, which possibly contributed to the improvement in the water quality indicators, as they became better in 2005/06 compared to the baseline scenario. These results provide evidence that the evaluation framework can provide reasonable predictive capacity on future environmental impacts from land use change within the case study catchment.

In the last part of this chapter, the overall effectiveness of the evaluation framework was discussed, and its strengths and limitations were identified for future applications. The strengths of the evaluation framework included its effectiveness in providing a good overview of changes in environmental quality associated with land use changes, and its flexibility, relative simplicity and logical structure. Its flexibility enables the framework to be customised for future applications, which take account of particular local circumstances. On the other hand, use of the framework may be limited to some extent by the uncertainty and the availability of appropriate input data, particularly parameters for the models utilised in the *Source* platform. It also depends on the availability of field data and high expertise of local ecology when scoring biodiversity conservation values of different land uses/land covers (BCVLu). Regarding the *Source* platform, however, intensive efforts to enable wider application of the platform to catchments outside Australia are currently in progress, and are expected to help solve several issues associated with the future application of the framework.

Chapter 6: Evaluation of the regional environmental consequences of bioenergy-driven land use changes [21]

6.1 Introduction

This chapter evaluates the regional environmental impacts of several land use change scenarios that incorporate 'underutilised agricultural land', and identifies how and under what conditions the conversion of these lands for bioenergy crop production can deliver better environmental sustainability outcomes. The results will support decision making on whether the use of such lands can minimise the impacts of bioenergy crop production. To address the research questions and aim of this research, the environmental outcomes of a set of bioenergy-driven land use change scenarios in the Burnett River catchment of Queensland, Australia were quantified by applying the spatially explicit evaluation framework (Chapter 3). The changes were assessed relative to the 2005/06 baseline land use scenario and threshold values currently used in the case study region, and compared with selected land use scenarios based on different land use change pathways, which incorporated underutilised agricultural land.

The six land use change scenarios were developed in the context of bioenergy development in Australia. Bioenergy has made a very small contribution to electricity and transport fuel production in Australia[22] (Geoscience Australia & ABARE 2010), and the bioenergy industry is in the early stages of large-scale commercialisation. Moreover, due to the recent change in the political climate related to climate change and renewable energy, the future of Australia's bioenergy industry is highly uncertain. However, interest in bioenergy was growing due to the imperative for climate change mitigation strategies by the former governments, rising oil prices, and a strong capacity for producing bioenergy feedstock. The CSIRO estimated a high potential for bioenergy feedstock production in Australia. It estimated that lignocellulosic feedstock from agriculture and forestry, algae oil and Pongamia oil combined could replace 23% of diesel use (4.2GL/year), and that SRC eucalypts could replace 15% of gasoline use (4.3GL/year) or 9% of electricity usage (20.2TWh/year) (Farine et al. 2012).

---

[21] This chapter is based on a manuscript submitted to *Agricultural Systems*. Miyake, S, Smith, C, Waters, D, Peterson, A, McAlpine, C, & Renouf, M, 'Environmental sustainability of using 'underutilised agricultural land' for future bioenergy crop production'.

[22] In 2007-08, bioenergy accounted for 3.9% of Australia's total primary energy consumption, and only 0.4% of the liquid transport fuels (Geoscience Australia and ABARE 2010).

6.2 Methods

6.2.1 Evaluation framework

A spatially explicit GIS-based evaluation framework (Chapter 3) was developed to offer a methodology for quantifying the regional-scale environmental effects of land use change, and to facilitate decision making on sustainable land use options. The framework generated information about eight environmental indicators (Figure 3.2). Water quantity and quality (and soil erosion) and terrestrial biodiversity were identified as critical environmental areas in relation to the regional-scale environmental effects of land use changes. These consequences are commonly experienced in many geographical regions (3.3.2), including the case study catchment. Thus this evaluation framework focuses on these indicators.

For the water indicators, a simple catchment scale water quantity and quality model was built using a physical hydrological modelling platform *Source* (version3.4.0) (eWater CRC 2011) to estimate run-off volume (ML/year), total suspended solids (TSS) (tonne/year), total phosphorous (TP) (kg/year) and total nitrogen (TN) (kg/year) for each land-use change scenario. To date, *Source* has been applied to a number of important hydrological projects in Australia and internationally to assess hydrological responses due to climatic, land use and land management changes (eWater CRC 2012). For biodiversity indicators, the spatial pattern analysis tool *Fragstats* (version 4.0) (McGarigal & Marks 1995) was used to calculated Class Area (CA) [total area], Largest Patch Index (LPI) [patch size] and number of patches (NP) of each native vegetation class. *Fragstats* has been widely used by landscape ecology professionals globally due to its versatility, applicability, simplicity and effectiveness in computing a wide range of metrics at different spatial and temporal scales.

In addition, ‘actual habitat amount’ was calculated using the biodiversity conservation value (BCV_{LU}) of each land class determined in the Southeast Queensland context (3.2.8 and Table 5.2 in Chapter 5). Values, with ranges, were given to all land use categories, including bioenergy crops (BCV_{LU} [low] = 0.15, BCV_{LU} [high] = 0.3). They were determined from a review of spatial ecology literature on specific taxa and species native to the region, including mammals, birds and reptiles (Martin & Catterall 2001; Kanowski, Catterall & Wardell-Johnson 2005; Kanowski et al. 2006; Martin et al. 2006; Eyre et al. 2009) and discussions with an expert familiar with the ecology of the region’s ecosystems and fauna (C. McAlpine 2014, pers. comm., 20 May). With an absence of field data and the subsequent high levels of uncertainty in terms of the ecological outcomes of individual bioenergy crops, a common value (i.e BCV_{LU} = 0.15 - 0.3) was assigned to both Pongamia and eucalypts. Using specific values for each crop would be controversial without justifications supported by sufficient field data, which should be obtained through future research. The assigned value for bioenergy crops was assumed to be higher than that for 'open grazing' (BCV_{LU} [low] = 0.1, BCV_{LU} [high] = 0.2) and for other cropping land categories (BCV_{LU} [low] = 0.05, BCV_{LU} [high] = 0.15), but lower than that for the conventional forest plantation (BCV_{LU} [low] = 0.25, BCV_{LU} [high] = 0.5). Many forest plantations in the region are long rotation (i.e. no less than 25 years) and have reduced disturbance from forest management activities. Other issues taken into consideration included: suggested similarities in the management intensity between woody perennial bioenergy crops and agricultural crops, particularly those for sort rotation coppice (SRC) of fast growing species (Bouget, Lassauce & Jonsell 2012; D. Lee 2013, pers. comm., 21 March); and opportunities of bioenergy plantation for creating habitats, shelters and movement corridors for certain species of wildlife compared to open land (C. McAlpine 2013, pers. comm., 17 July).

6.2.2 Land use change scenarios

6.2.2.1 Case study region

The Burnett River catchment (33,257 km^2) is predominantly agricultural with beef cattle grazing comprising over 75% of the total area. It is located in one of the most ecologically diverse and sensitive regions in subtropical Queensland, Australia. It is at the southern end of the Great Barrier Reef (GBR), and within the Brigalow Belt Bioregion, which is one of 11 National Biodiversity Hotspots (Australian Government 2003) (Figure 6.1). The coastal area has higher rainfall and a lower range of temperature than inland areas. The 2050 climate data indicates that over half of the inland areas have an annual rainfall of less than 700mm (Mitchell & Osborn 2005). Land clearing and agricultural expansion have resulted in the removal of almost 70% of the original native vegetation within the catchment since European settlement, including threatened brigalow (*Acacia harpophylla*) (Wide Bay Burnett Environment and Natural Resources Working Group 2012). This resulted in the loss of important habitat for threatened species (e.g. small woodland birds) (Australian Government 2003; Martin et al. 2006; Maron & Kennedy 2007; Eyre et al. 2009), significant degradation of groundwater and surface water quality in the GBR lagoon, and serious threats to the reef's valuable marine ecosystem (Queensland Department of the Premier and Cabinet 2009a, 2009b). In response, a number of planning and institutional improvements were achieved under federal and state government policies over the last two decades. In the case study region, Queensland's *Vegetation Management Act 1999* restricted broad-scale clearing of remnant (structurally intact) native vegetation (Queensland Government 2009b), and a number of a joint government initiatives were launched to increase the adoption of improved land and agricultural management practices (Queensland Department of the Premier and Cabinet 2009a, 2009b). However, subsequent to a change in State government, Queensland's vegetation laws were amended significantly in December 2013 to increase landholders' ability to clear land for agricultural and development activities (Queensland Government 2013). There is currently no mandate for bioenergy production in Queensland, yet a number of research and development projects in biofuels and bioenergy have been supported by the Queensland Government, such as the research field in sugarcane and second generation feedstock (Queensland Government 2014).

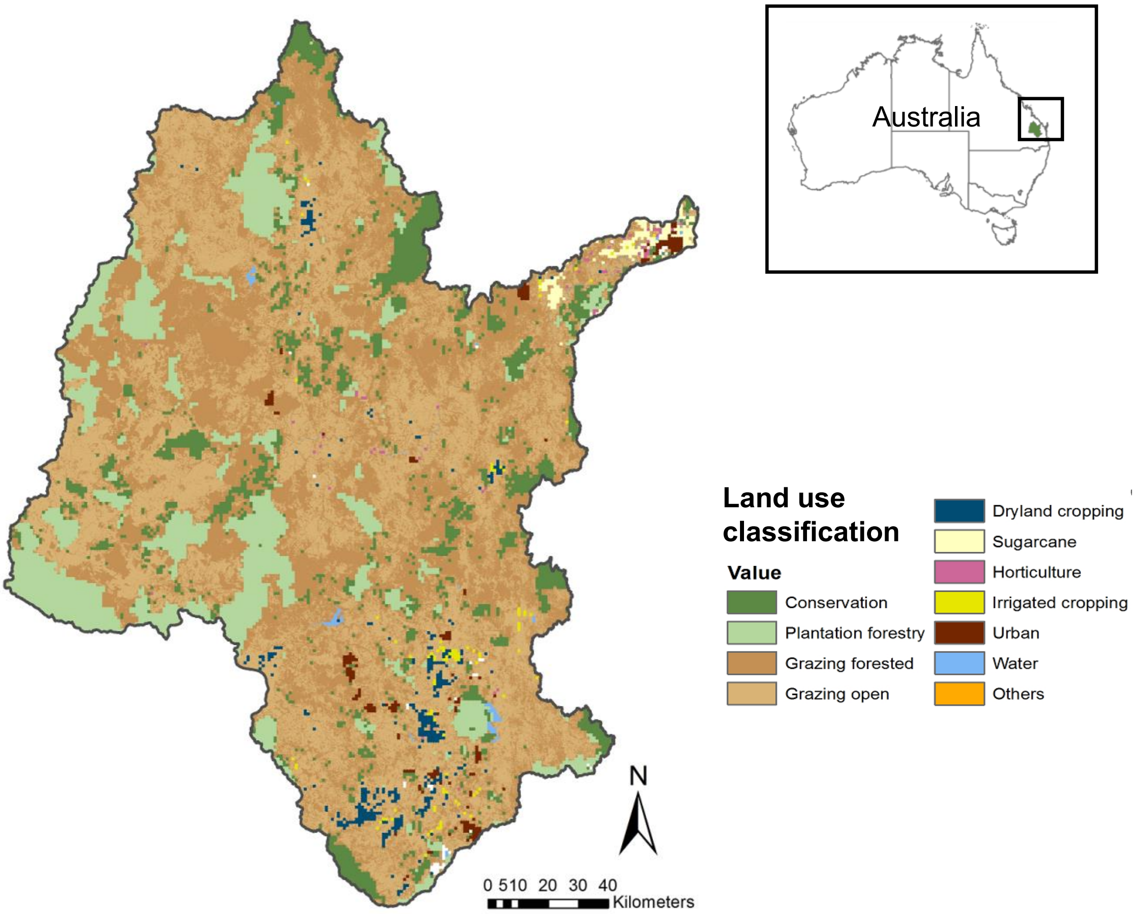


Figure 6.1 Land use of the Burnett River catchment 2005/06.

6.2.2.2 Key drivers affecting land use change scenarios

Regional-scale agricultural land use change decisions are driven by interacting factors - biophysical, demographic, economic, technological, political, institutional, and social/cultural (Geist et al. 2006; Seabrook, McAlpine & Fensham 2006). All drivers often act in synergy, yet economic opportunities frequently are the primary drivers of land use change for individual landholders in many situations (Lambin et al. 2001; Seabrook, McAlpine & Fensham 2006; Bryan, Ward & Hobbs 2008; Bryan King & Wang 2010; van der Hilst et al. 2010; Odeh, Tan & Ancev 2011). Economic viability is determined by establishment cost (land price, preparation, planting and maintenance in the early years of plantation), annual operational costs (costs of fertiliser, pesticide, maintenance, transport and harvesting), and total revenue influenced by market demand and value, yield, and the productivity of the crops (Bryan, Ward & Hobbs 2008; Odeh, Tan & Ancev 2011; van der Hilst et al. 2012b). Other important factors are related to infrastructure, such as accessibility to roads, transport, water-related infrastructure, and processing plants (and their capacity), and political and institutional factors which can substantially influence land use change through regulatory and economic (e.g. taxes, subsidiaries) instruments, as exemplified by bioenergy targets established in the U.S.A. and the EU (Miyake et al. 2012; van der Hilst et al. 2012b).

Biophysical properties, such as soil type, topography, vegetation and climatic conditions are the most obvious constraints to agricultural development and regional-scale landscape change (Seabrook, McAlpine & Fensham 2006). Land use change scenarios for this spatially explicit evaluation were developed based on biophysical suitability, and not on economic viability. This was due to: (i) uncertainty about bioenergy/biofuel policy and future production projections in the absence of clear targets at the national and state levels in Australia; and (ii) lack of field data from commercial scale production of Pongamia and SRC eucalypts, such as production and transport costs. In the absence of an economic rationale for the scenarios, ‘extreme’ land use change scenarios were assessed to highlight the potential environmental effects and implications caused by bioenergy production on underutilised agricultural land.

The process for land use change scenario development involved the following three steps: selection of bioenergy crops and identification of areas for crop production from biophysical suitability; identification of land use change pathways most suited for the case study context; and identification of underutilised agricultural land.

6.2.2.3 Bioenergy crops for subtropical Queensland

This research focused on the environmental advantages of converting underutilised agricultural land to perennial bioenergy crops (Pathway 4 in Figure 2.2, Chapter 2). The bioenergy crops were selected from species native to the case study region and/or tropical regions in Australia. Two different crops were considered in the scenarios: Pongamia; and two eucalypt species (Spotted gum and Chinchilla white gum). Those crops have high potential for producing bioenergy in subtropical areas of Queensland, and can grow with low rainfall, high temperatures, poor soils and with minimal water and agrochemical inputs (Odeh & Tan 2007; Odeh, Tan & Ancev 2011; Shepherd et al. 2011a; Murphy et al. 2012).

Pongamia is a leguminous oilseed crop. Research and development is currently in progress for biodiesel and other liquid biofuel (e.g. jet fuels) production from its bio-based oil (Murphy et al. 2012). It is not a major source of lignocellulosic biomass, although the seedpod left after extracting oil, and the prunings can generate a useful quantity of biomass (P. Scott 2013, pers. comm., 14 October). It is native to the Indian sub-continent, central and southeast Asia, and northern Australia (DEEDI 2010). Several field trials across southeast Queensland have demonstrated that the growth, seed yield (around 12.6 tonne of seed per hectare (ha) with around 20,000 seeds from 10 year old trees), and oil content (around 40%) are promising (Scott et al. 2008). The greatest benefits expected from the crop are the low requirement for annual rainfall (500-800 mm) and its ability to fix nitrogen from the atmosphere (Kazakoff, Gresshoff & Scott 2011; Murphy et al. 2012).

Eucalypts (some 900 species) are also native to, and common species in, Australia. Spotted gum in particular is very common in the case study region and has been used traditionally for hardwood or pulp production, and more recently has been considered as a source of woody biomass for firewood, charcoal, and electricity production. There is also an emerging potential for biomass as feedstock for second-generation biofuel production (Shepherd et al. 2011a; Shepherd et al. 2011b). Many eucalypt species coppice and provide highly competitive growth performance in plantations (Shepherd et al. 2011a). Species selection trials for hardwood production in subtropical Queensland indicated that Spotted gum was very reliable across a range of site types and was more resilient to pests and diseases than many other species (Lee et al. 2010). Spotted gum can grow in areas with an annual minimum rainfall of 700 mm. Chinchilla white gum is suited to more marginal climate conditions (Queensland CRA RFA Steering Committee 1998) and can grow in areas with annual rainfall of 650 mm and above (D. Lee 2013, pers. comm. [e-mail], 3 April).

The environmental outcomes from large-scale production of these crops require rigorous further evaluations, particularly field-based research. Although this study did not aim to compare the environmental performances of Pongamia and the eucalypts because of lack of existing field data, there are several characteristics of these crops that could affect environmental outcomes. For example, Pongamia plantations are likely to involve fewer disturbances associated with harvesting and fewer requirements for N fertiliser (P. Scott 2012, pers. comm., 28 November). However, commercial-scale production of this crop is new in the subtropical context, and its biodiversity implications are largely unknown even though the crop is identified as native to tropical regions in Australia (DEEDI 2010). In comparison, Spotted gum is a very common species in both native forests and forestry plantations in the case study region. Its SRC plantations, however, have the potential to produce greater environmental impact compared to conventional forestry due to higher amounts of agrochemical inputs, shorter harvesting cycles (2-3 years) and extensive residue removal from sites (D. Lee 2013, pers. comm., 21 March).

6.2.2.4 Identification of 'underutilised agricultural land' and three land use change pathways most suited to the subtropical Australian context

The definition of underutilised agricultural land in the Australian context may be different from that in other geographical regions. Most cleared land in Australia, however marginal, has been used traditionally for some form of agriculture (mostly grazing) and/or forestry in rural communities. Hence, to some extent, bioenergy crop production may involve trade-offs in relation to food production (Farine et al. 2012), which suggestes a potential risk for displacement of existing activities to other locations/countries, or what is termed indirect land use change (iLUC). Such a risk would be much smaller in the Australian context than developing regions such as Brazil (Chapter 2), due to relatively effective political, institutional, and enforcement capacities in environmental and land use policies and planning. However, it is important to be aware of the potential risk of displacement, even though the land is classified as marginal and unsuitable for most agricultural activities.

The definition of underutilised agricultural land used in this study included ‘ambiguous lower quality land’, which is “land that is not necessarily unsuitable for food production but where food production is less productive” (Shortall 2013, pp23). It encompassed land that has limitations for agricultural activities, particularly food crop production, and included ‘abandoned agricultural land’, ‘marginal land’ and ‘low productivity land’ (Figure 1.1 in Chapter 1). Such lands were identified from the Queensland Government’s land capability maps for the Burnett region (Vandersee & Kent 1983; Donnollan & Searle 1999; Kent 2002), which indicate each land unit from Class I (most suitable land for all agricultural and pastoral uses) to Class VIII (land with severe limitations, unsuited for any agricultural uses). In this study, underutilised agricultural land included land in Class V (land that is arable but has limitations which, unless removed, make cultivation impractical and/or uneconomical) to Class VIII (Appendix Figure A6.1).

In the Australian context, conversion from beef cattle grazing to bioenergy crops is a possible land use change option largely due to land availability. There were also a number of proposed environmental and socio-economic benefits for farmers. Grazing is the most widespread land use in Australia accounting for 55% of the continent’s total land area (Bureau of Rural Science 2010; State of the Environment Committee 2011). The grazing industry, especially beef cattle grazing, plays an important role in the country’s economy. Due to the biophysical constraints (mostly rainfall patterns in Australia), suitable land for settlements and cropping are constrained to subtropical and temperate regions along the coastline of the continent. ‘Marginal’ agricultural lands in arid and semi-arid regions of inland Australia are often used for some form of grazing. In the Burnett River catchment, expansion of beef cattle grazing (5.2% between 1992/93 and 2005/06) has been a notable land use change over the past few decades (Chapter 4). Generally, ‘unproductive’ or ‘marginal’ cropping lands previously used for dryland agriculture, were converted to beef cattle grazing due to a combination of socio-economic factors experienced in the region (I. Crosthwaite 2012, pers. comm., 20 August; D. O’Sullivan 2012, pers. comm., 22 August).

In addition, due to increasing global beef consumption, expansion of beef cattle production is one of the greatest environmental challenges in Australia (State of the Environment Committee 2011). These challenges relate to GHG emission, deterioration in soil and water quality, and loss of biodiversity through alteration of habitat (e.g. vegetation clearing, changes in mid- and lower storey forest structure), changes in the proportions and mixtures of species, and the grazing activities themselves (McAlpine et al. 2009; State of the Environment Committee 2011). Open grazing land with modified pasture incurs additional impacts, such as invasive species and changes in fire regimes (McAlpine et al. 2009; State of the Environment Committee 2011). In this context, new farming models that integrate bioenergy production with grazing landscapes have been proposed for rural Australia (Shepherd et al. 2011b; Murphy et al. 2012). The potential benefits discussed in the existing literature include diversification from by-products and a reduction in the risk associated with farming. So far, little is known about the economic viability of production, scale, and value chains of these bioenergy crops grown in a complementary way with grazing due to the scarcity of field trials. However, the scenarios developed in this research are in line with such a proposed production model.

6.2.2.5 Defining six land use change scenarios

Six land use change scenarios were prepared based on the three land use change pathways (Figure 6.2 and 6.3).

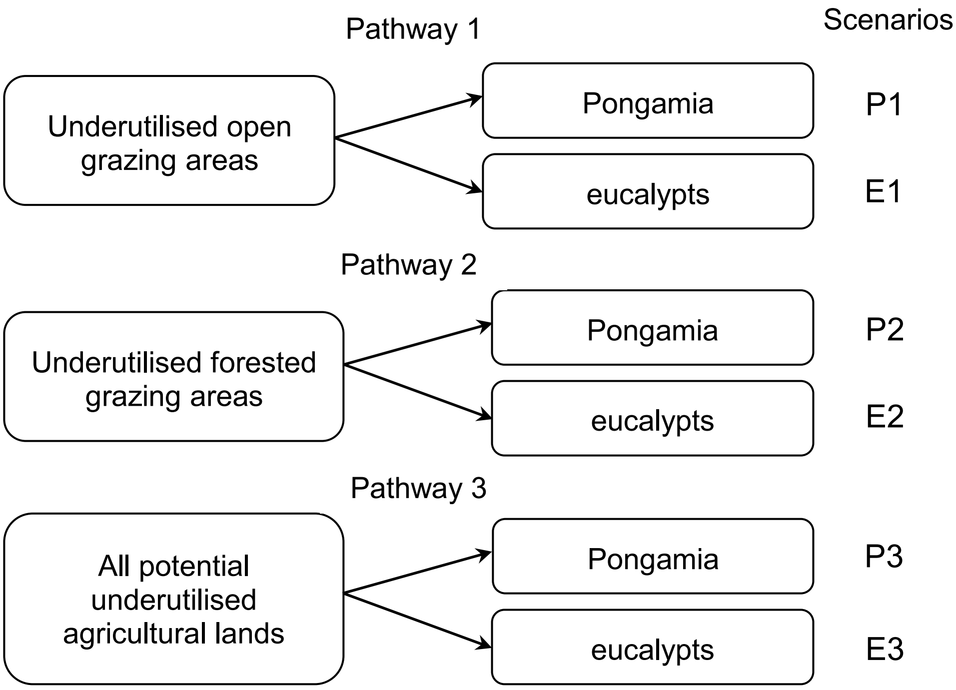


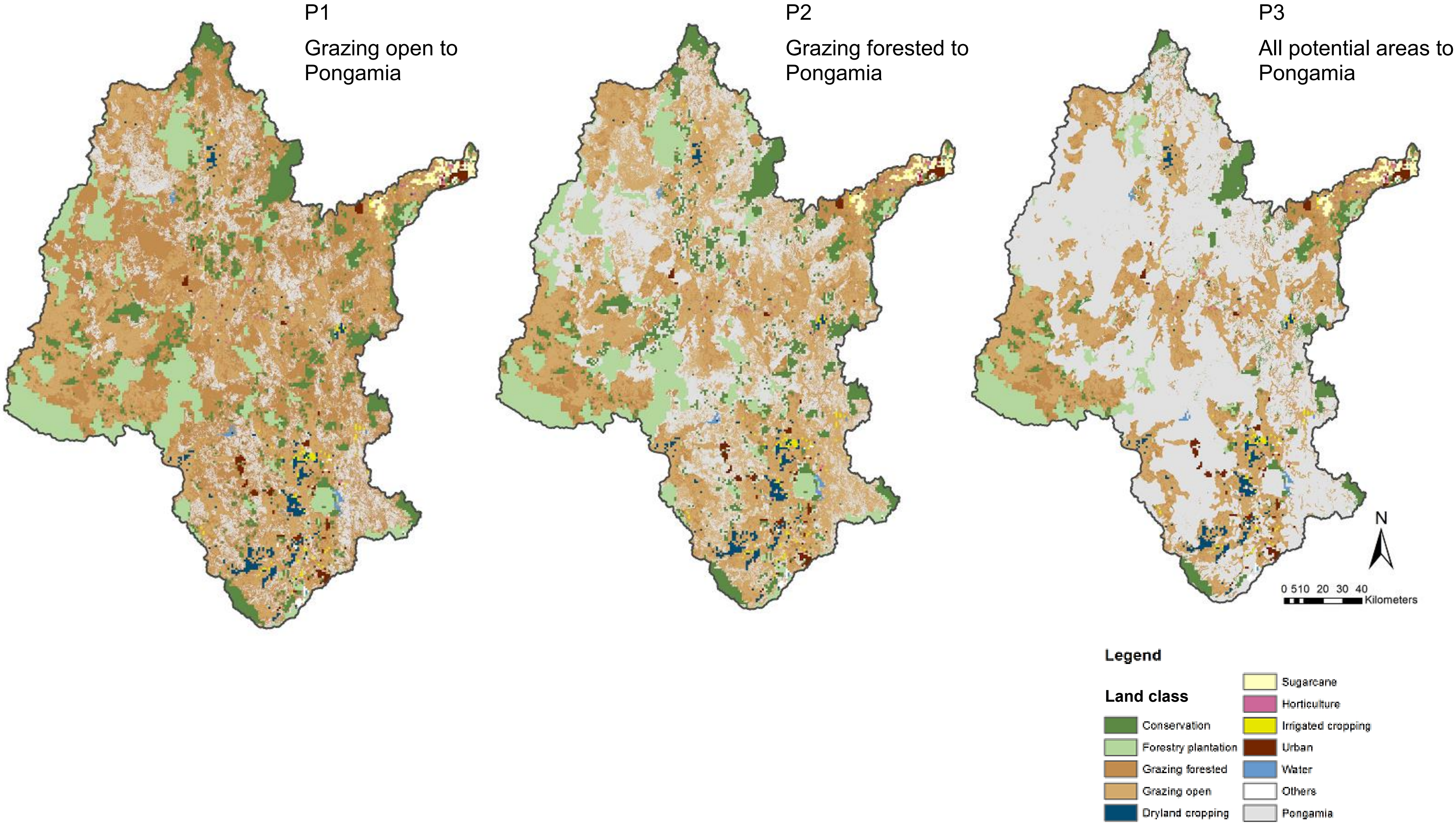
Figure 6.2 Land use change pathways and land use change scenarios investigated in this study.

Pathways 1 and 2 assumed the establishment of these plantations on grazing land identified as underutilised agricultural land. Pathway 3 was prepared as an extreme case to achieve maximum bioenergy production. It involved conversion of all areas, identified as suitable areas for each bioenergy crop within the underutilised agricultural land (i.e. land in Class V to Class VIII). Thus six scenarios were examined. This involved conversion to Pongamia through three land use change pathways - P1 (from open grazing), P2 (from forested grazing) and P3 (all potential underutilised agricultural land), and three pathways involved conversion of grazing land to eucalypt species (E1, E2 and E3). Pongamia scenarios allocated 0.6 million ha (16.9% in P1), 0.9 million ha (28.5% in P2) and 1.9 million ha (56.4% in P3) of the total land area to crop production, while eucalypt scenarios included less crop extent since the crop requires higher rainfall than Pongamia, thus allocating 0.4 million ha (11.9% in E1), 0.6 million ha (18.9% in E2) and 1.2 million ha (37.2% in E3) of the total land area to crop production.

6.2.2.6 Potential bioenergy production from these land use change scenarios

Bioenergy feedstock production from each scenario was calculated from broad assumptions of yields and area of utilised land (Table 6.1). However, these numbers are indicative only because yields were estimated from a small number of plantings on trial sites under very different conditions.

The Pongamia scenarios were assumed to produce bio-based oil (feedstock for biodiesel), seed cake (which can be digested or fermented to methane gas or ethanol respectively), and seed pods (feedstock for lignocellulosic ethanol) (Klein-Marcuschamer et al. 2013). Oil yield was estimated to be 5.0 tonne/ha/year (Scott et al. 2008), and seed cake and seed pod yields were estimated to be 4.5 - 7.5 and 7.5 - 12.5 tonne/ha/year, respectively (P. Scott 2013, pers. comm. [e-mail], 3 October). However, these estimates were based on ‘elite’ trees planted on highly productive land, good agricultural soils and under irrigation in the southern Queensland context (Gatton). Pongamia is genetically diverse and has significant variation in many traits including yield (Sharma et al. 2011). Thus the extrapolated oil and biomass yields may be significantly less than those estimates, particularly on plantations established on underutilised agricultural land in rain-fed conditions.


P1
Grazing open to Pongamia
P2
Grazing forested to Pongamia
P3
All potential areas to Pongamia
N
0 510 20 30 40
Kilometers
Legend
Land class
Conservation
Forestry plantation
Grazing forested
Grazing open
Dryland cropping
Sugarcane
Horticulture
Irrigated cropping
Urban
Water
Others
Pongamia

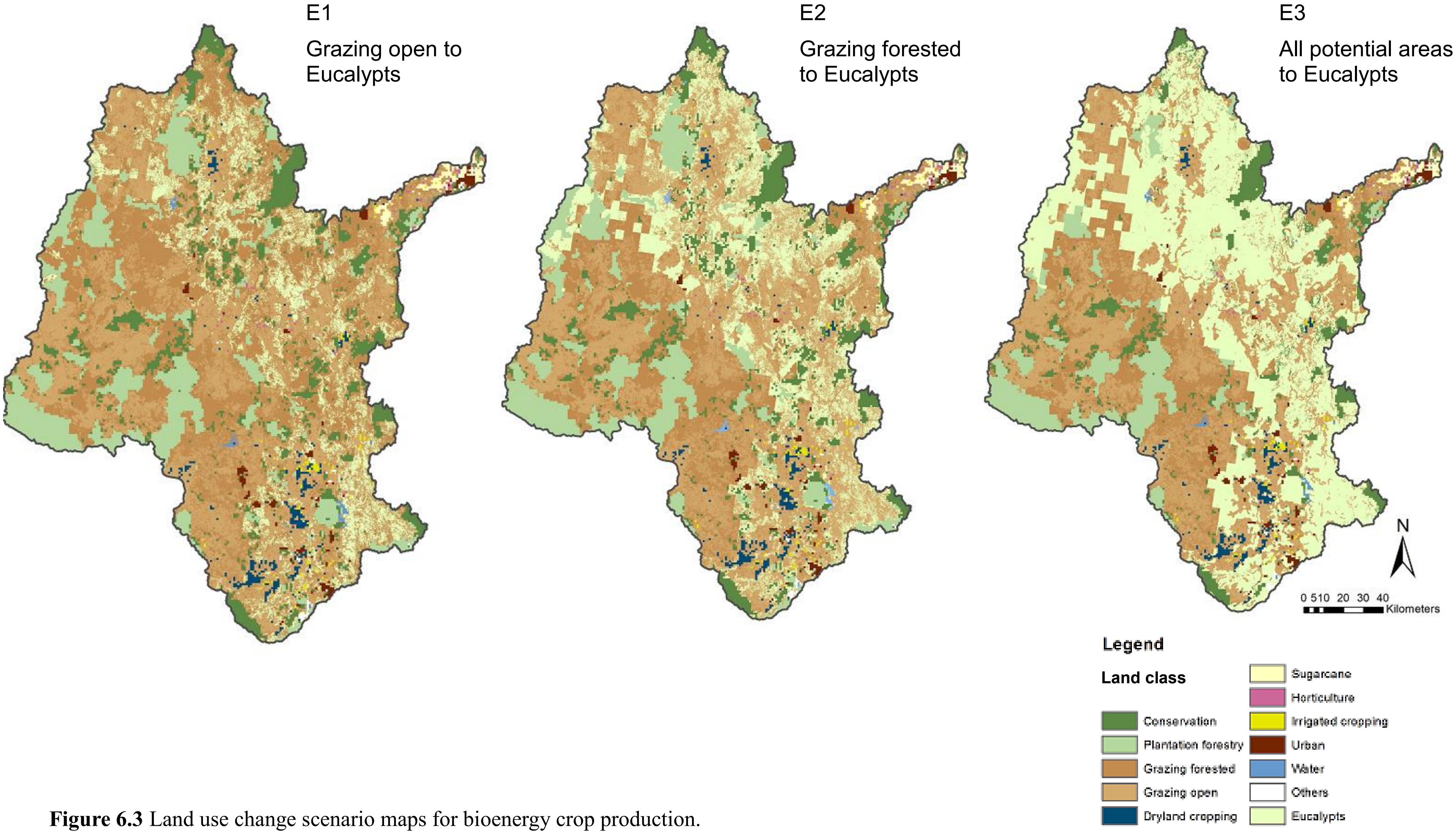


Figure 6.3 Land use change scenario maps for bioenergy crop production.

Biomass production from the two eucalypt species (feedstock for lignocellulosic ethanol) was estimated in a much more conservative way, based on field data for the inland Burnett region on areas without irrigation (Lee et al. 2011). The total biomass yield per tree was estimated to be approximately 5.0 tonne/ha/year, of which 4.0 tonne/ha/year (80%) is above-ground biomass[23] (D. Lee 2013, pers. comm. [e-mail], 28 August) and 1.0 tonne/ha/year (20%) is root biomass (D. Lee 2014, pers. comm. [e-mail], 25 February). The above-ground biomass yield was based on the volume mean annual increment (MAI) of trees for hardwood production (Lee et al. 2011). The biomass yields of these eucalypt species may be more than double this in the coastal areas where higher rainfall is expected (Lee et al. 2011).

Table 6.1 Estimated bioeneregy production (by bioenergy products) from land use change scenarios (indicative only).

	Bio-based oil (MT/year)	**Biomass** (MT/year)	**Ethanol** (ML/year)	**Synfuel** (ML/year)	**Electricity** (TWh)
2005/06 Baseline	0	0	0	0	0
P1 (Pongamia)	2.8	4.1-7.0	1,086-1,810	1,028-1,713	5.6-9.4
P2 (Pongamia)	4.7	7.1-11.8	1,834-3,057	1,735-2,892	9.5-15.9
P3 (Pongamia)	9.3	14.0-23.3	3,633-6,056	3,438-5,730	18.9-31.4
E1 (Eucalypts)	-	2.0	512	484	2.7
E2 (Eucalypts)	-	3.1	813	770	4.2
E3 (Eucalypts)	-	6.2	1,600	1,514	8.3

*Pongamia: oil and biomass yields were estimated from data on best performance trees planted on highly productive land good agricultural soils and under irrigation in southern Queensland (Gatton) (P. Scott 2013, pers. comm. [e-mail], 3 October).

** eucalypts: above-ground biomass yield was estimated from the volume MAI of trees at a taxa trial site in the inland Burnett region (without irrigation) for hardwood plantations (Lee et al. 2011). The approximate allocations of biomass yields of age 6-10 year old trees were estimated to be 20% from root, 60% from stem, and 20% from leaves (D. Lee 2014, pers. comm. [e-mail], 25 February).

Based on these assumptions, the Pongamia scenarios could produce 2.8 - 9.3 million tonnes (MT)/ha/year of oil, and 4.1 - 23.3 MT/ha/year of biomass, from which 1,086 - 6,056 million litres (ML)/year of ethanol, 1,027 - 5,730ML/year of synfuel, or 5.6 - 31.4 terawatt hours (TWh)/year of electricity[24] could be derived. The eucalypt scenarios could produce 2.0 - 6.2 MT/year of biomass, from which 512 - 1,600 ML/year of ethanol, 484 - 1,514 ML/year of synfuel, or 2.7 - 8.3 TWh/year of electricity could be derived. However, it is important to

[23] The biomass production was based on the premise that the average volume of aboveground forest biomass (stem and leaves) (MAI) of two species was 5.1m^3/ha/year, and average basic density was 863.75kg m^3 (Lee et al. 2011).

[24] Production of each bioenergy product was estimated based on values for estimated energy and fuel yields of 'bioenergy plantation', which were used by O'Connell et al. (2009). The values were 260L/t (ethanol), 246L/t (synfuel) and 1.35MWh/t (bioelectricity).

emphasize that these estimates were derived under very different assumptions and field trial conditions. In particular, those for the Pongamia scenarios were based on ‘elite’ trees planted on highly productive land, good agricultural soils and under irrigation. Yields for production on underutilised agricultural land could be significantly lower than this. They are provided to give some context to the scenarios evaluated.

6.2.3 Mapping procedure

The land use change scenarios were prepared using ESRI ArcGIS 10. The national scale land use map for 2005/06 from *Land Use of Australia, version 4* (Bureau of Rural Science 2010) and the Queensland 2005 Regional Ecosystem (RE) map (*version 6.0b*) (DERM 2009) were used as base maps, and also as the baseline to evaluate the change in the overall environmental performance of the land use scenarios. Grazing lands were categorised into open grazing (i.e. cleared pastures) and forested grazing (i.e. grazing in woody native vegetation) areas. The distinction between the two categories was made by using the threshold of 12% Foliage Projective Cover (FPC) based on the dataset from the Queensland Government’s major vegetation monitoring initiative, Statewide Landcover and Trees Study (SLATS) (DSITIA 2012a; 2012b). FPC represents the level of woody vegetation coverage. The 12% FPC definition approximates a 20% crown cover, which has been used commonly as the threshold for vegetation clearing in remote sensing studies (DNRM 2003).

In parallel, underutilised agricultural land was identified from land capability maps (Appendix Figure 6.2), and land for Pongamia and the eucalypt species was based on their specific crop requirements in relation to climate, topographic and agronomic conditions (Table 6.2). The suitable areas for growing Pongamia and the eucalypt species were then overlaid with the following maps:

- The national land use map (Bureau of Rural Science 2010) to exclude nature conservation areas, water (including a buffer zone of 200 metres to avoid the possibility of seed dispersal), and urban land uses;
- The Digital Elevation Model (DEM) to exclude areas with slopes greater than 20 degrees; and
- The gridded climate dataset projections from WorldClim for the 2050s (Hijmans et al. 2005) to exclude areas with rainfall and temperatures which will be above and beyond the crop requirements.

As a result, 1,866,833 ha of land was identified as suitable for Pongamia production, and 1,234,053 ha for eucalypt production within the catchment. Pathway 3 involved conversion of all these lands to bioenergy crops.

Table 6.2 Specific crop requirements of Pongamia and the eucalypt species in relation to climate, topographic and agronomic conditions

Requirement	Pongamia*	Eucalypts**
Min. average temperature in July (^{o}C)	≧1	≧0 (Spotted gum only)
Frost days in July	< 4	≦5-10 (Spotted gum only)
Max. average temperature in January (^{o}C)	tolerant to hot temperature	tolerant to hot temperature
Annual rainfall (mm)	500-2,000	700-1,600 (Spotted gum) 650-850 (Chinchilla white gum)
Slope (%)	≦20	≦20
Buffer from 'water' and 'Nature conservation' land class (m)	200	-

*The specific requirements of Pongamia were derived from Murphy et al. (2012), Odeh & Tan (2007), Odeh, Tan & Ancev (2011) and finalised by an expert opinion (P. Scott 2012, pers. comm., 28 November).
**The requirements of eucalypt species were derived from Barton and Perekh (2005), Javanovic and Booth (2002), Queensland CRA RFA Steering Committee (1998) and finalised by an expert opinion (D. Lee 2013, pers. comm. [e-mail] 27 March and 3 April)

6.2.4 Application of the evaluation framework

Data pertaining to the land use change scenarios (Figure 6.1 and 6.3) were the main input data to the spatially explicit models/methods incorporated into the evaluation framework – spatial hydrological modelling platform *Source*, spatial landscape pattern tool *Fragstats*, and calculation for the actual habitat amount. All input maps for the models/methods, including the base land use maps, were prepared in ASCII grid format with a cell size of 100 metres. In this evaluation, as *Source* required the 12 major land use classes corresponding to different hydrological responses (called Function Units [FU]), the land use classes on the base land use map were reclassed (Fentie in press). At the same time, the input data for *Fragstats* were prepared by overlaying the FU map prepared for *Source* inputs and the vegetation map (Regional Ecosystem [RE]) map [version 6.0b]) (Queensland Herbarium 2012) using six classes for native vegetation area and the 12 FU classes outside the native vegetation area.

6.2.5 Sensitivity and uncertainty estimates

In spatially explicit modelling based on GIS, there is always uncertainty in input data, resolution and parameters, which significantly influence model outputs (Verstegen et al. 2012). In this research for example, difference in mapping methodologies and data resolution between the Australian Government (Land Use of Australia, *version 3*) (1:2,500,000) (ABARES & BRS 2006) and the Queensland Government (the Queensland Land Use Mapping Program [QLUMP] (DSITIA 2012g) (4.3.3.3) resulted in slight differences in land use classes in the case study catchment. Thus awareness of uncertainties in input data is important in understanding and interpreting the model results (5.4.2).

The catchment model developed within the *Source* modelling platform required more input datasets than *Fragstats*. This included long-term daily rainfall and potential evapotranspiration (PET) data (climate data from 1 January 1970 to 31 December 2009 was used in this evaluation), and parameters for the rainfall-runoff model and constituent generation model. For this evaluation, the results of finer scale modelling were available as a base calibrated model for the case study catchment (Fentie in press), which had been developed as part of the Queensland Government's Paddock to Reef program (Waters & Carroll 2013). The parameters developed for the base calibrated model were regarded as well tested against the observed data and other modelling results by the Queensland Government hydrologists, and thus less uncertainity was involved in relation to parameters for land use classes on the existing land use maps. Neverthless, the model development process required adjustment of the parameters to enable their specific application to the case study catchment in consultation with Queeensland hydrologists (5.2).

Identifying the best alternative parameters for Pongamia and eucalypts plantations was the most challenging process for this evaluation due to high levels of uncertainty stemming from the lack of actual field monitoring data. Conventional methods, such as Monte-Carlo sensitivity analysis were not regarded as suitable for this analysis because of the difficulty in estimating probability ranges. To overcome this challenge, results for the hydrological and water quality models were generated for low and high management intensity options using a set of parameter ranges (Table 6.3).

Table 6.3 Parameter ranges and proxy parameters used for the constituent generation model (Source: generated from values in Fentie in press; Queensland Department of Natural Resource and Mines).

Bioenergy crop	Constituents	Parameter range (mg/L)
Pongamia	TSS	EMC*: 235.7(forested grazing) - 275 (horticulture) DWC**: 100 (forested grazing) - 137.5 (horticulture)
	TN	EMC*: 0.6 (forested grazing) DWC**: 0.17 (forested grazing)
	TP	EMC*: 0.17(forested grazing) - 0.54 (horticulture) DWC**: 0.02 (forested grazing) -0.28 (horticulture)
Eucalypts	TSS	EMC*: 44 (forestry plantation) - 275 (horticulture) DWC**: 22 (forestry plantation) - 137.5 (horticulture)
	TN	EMC*: 0.48 (forestry plantation) – 3.01 (horticulture) DWC**: 0.24 (forestry plantation) – 1.5 (horticulture)
	TP	EMC*: 0.08 (forestry plantation) – 0.54 (horticulture) DWC**: 0.04 (forestry plantation) – 0.28 (horticulture)

*EMC: The Event Mean Concentration (average constituent concentration over a storm event)
** DWC: The Dry Weather Concentration (constituent concentration during dry weather or baseflow conditions)

The parameter ranges for these two options had to rely largely on expert opinions on the agricultural and/or forestry production and management systems. For example, the sediment and nutrient generation rates of SRC eucalypt plantations were assumed to be in the range between conventional plantation forestry (‘low management intensity option’) and woody perennial horticultural crops, e.g. orchard plantation (‘high management intensity option)’. The majority of conventional plantation forestry in the case study region cultivates spotted gum for hardwood and fibre production with a long rotation, and involving much less maintenance, agrochemical inputs and harvesting activity than the SRC plantation. Bioenergy crops in all scenarios are produced without irrigation. However, the SRC plantation may require a similar amount of fertiliser inputs and disturbance for maintenance and harvesting as some horticultural crops (D. Lee 2013, pers. comm., 21 March). Likewise, all proxy parameters for other pollutants and all values for Pongamia were identified using the same approach, which involved adopting the original parameters for the land categories that involved similar agricultural and/or forestry management systems (P. Scott 2013, pers. comm., 17 April). The original values were based on those developed by Fentie (in press) for existing land uses in the Burnett River catchment, as part of the Queensland Government’s Paddock to Reef Program.

Range were also derived for the biodiversity conservation values corresponding to land cover/land use categories (BCV_{LU}), used to calculate the 'actual habitat amount' of each land use change scenario (5.2.8, Table 5.2). This was important to address the high level of variance and uncertainty in these values, which were estimated from the literature and an expert familiar with the ecology of the region's ecosystems and fauna (C. McAlpine 2014, pers. comm., 20 May) (5.2.8 and 6.2.1).

6.3 Results

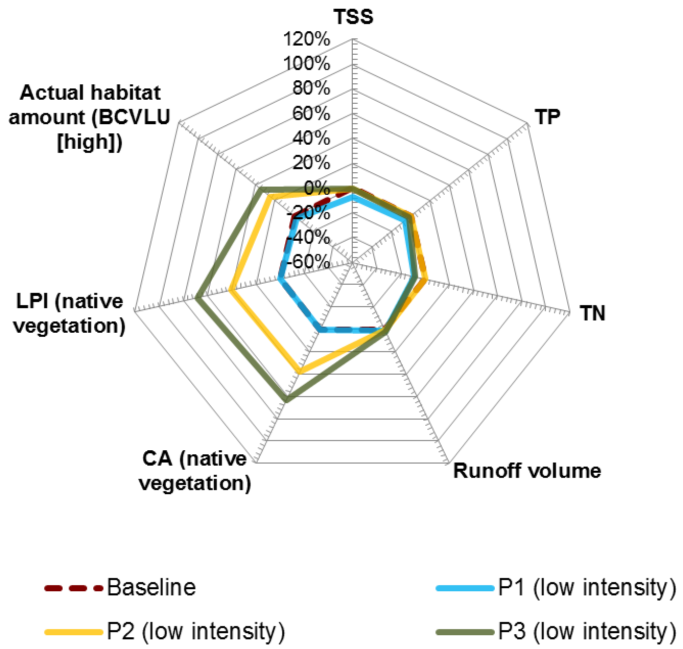
6.3.1 Radar charts

The environmental performance of all scenarios was compared to the baseline scenario (2005/06) using percentage change for each environmental indicator. It was presented in radar charts to facilitate comparison and interpretation of multiple indicators simultaneously (Figure 6.4 [a] and [b]). The application of the evaluation framework was implemented for both catchment and sub-catchment scales, but results are presented for the catchment scale. Maps were generated from the results at the sub-catchment scale to facilitate better understanding of the geographic distribution of the results for each indicator (Appendix Figure A6.3 to A6.6).

6.3.2 Key factors influencing the overall results

Both the extent of the land use change and the management intensity influenced the overall results. For example, the eucalypt scenarios (E1, E2 and E3) involved less crop extent/land use change compared to the Pongamia scenarios, due to the smaller amount of land identified as suitable for the production of Pongamia (Figure 6.3). The management intensity significantly influenced the water quality indicators (i.e. TSS, TN and TP) of each scenario, with the low management intensity options performing much better than the high management intensity options.

Low management intensity



High management intensity

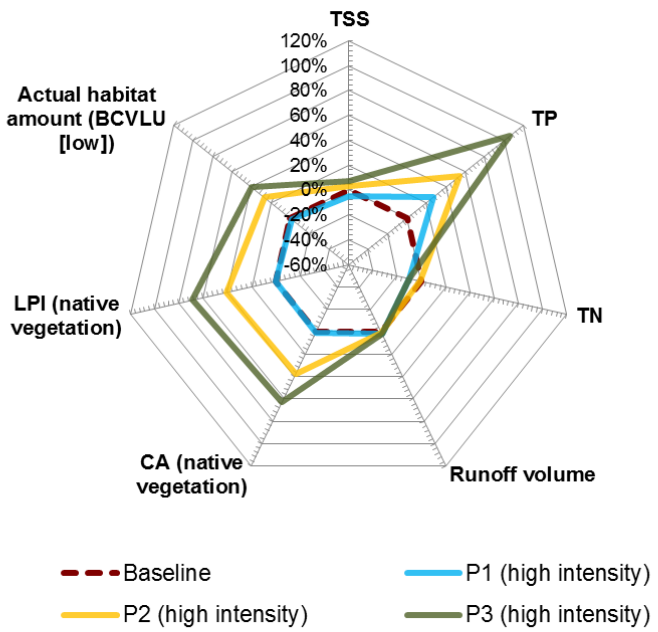
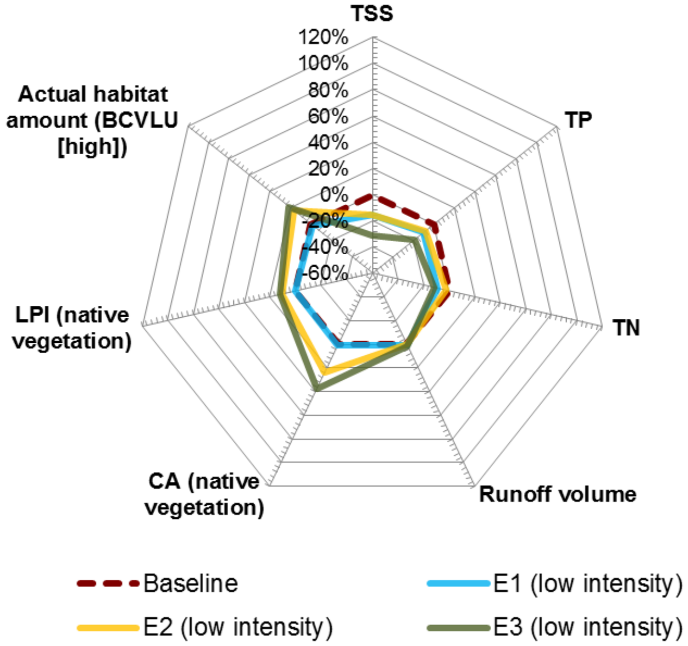


Figure 6.4 [a] Overall environmental consequences of Pongamia land use change scenarios in the Burnett River catchment. A graph with a larger area indicates a scenario with worse overall environmental performance.

P1—conversion of 'underutilised' grazing open. P2—conversion of 'underutilised' grazing forested. P3—conversion of all potential 'underutilised agricultural lands' to Pongamia production

Low management intensity



High management intensity

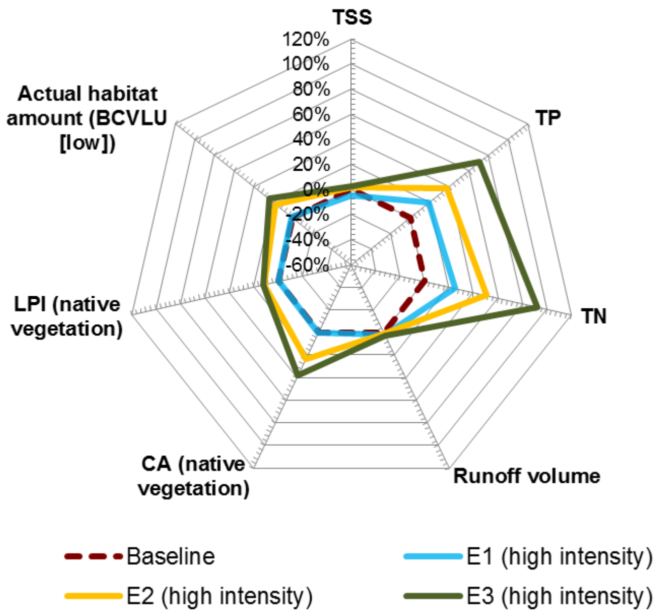


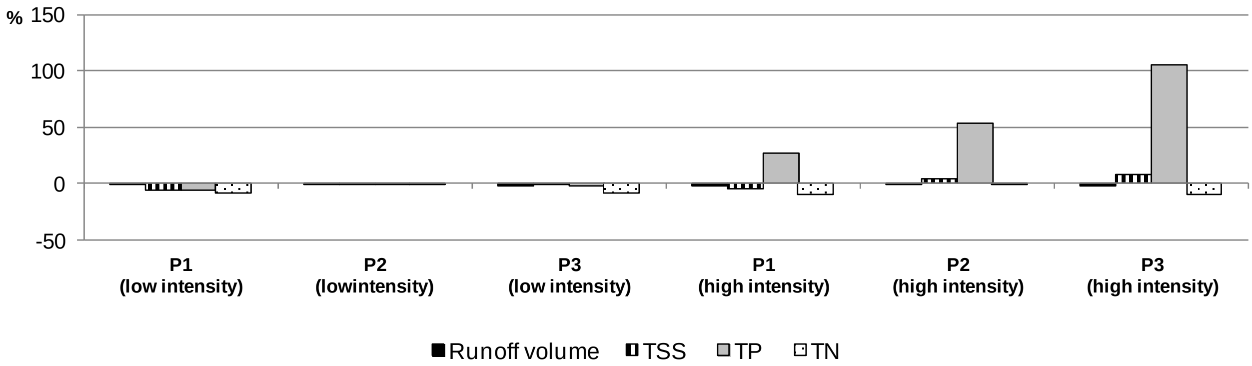
Figure 6.4 [b] Overall environmental consequences of eucalypt land use change scenarios in the Burnett River catchment. A graph with a larger area indicates a scenario with worse overall environmental performance.

E1—conversion of 'underutilised' grazing open. E2—conversion of 'underutilised' grazing forested. E3—conversion of all potential 'underutilised agricultural lands' to Pongamia production

6.3.3 Pathway that enhances environmental quality

The results indicated that Pathway 1 (conversion of 'underutilised' open grazing lands to bioenergy crops: P1 and E1) under low management intensity was the only scenario that slightly enhanced the overall environmental qualities in the case study catchment (Figure 6.4 [a] and [b]). Scenario E1 (low intensity management option) produced the best environmental outcome by reducing TSS (-15.1%), TP (-9.9%) and TN (-8.8%) loads from the baseline (Figure 6.5).

(a) Pongamia



(b) Eucalypt species

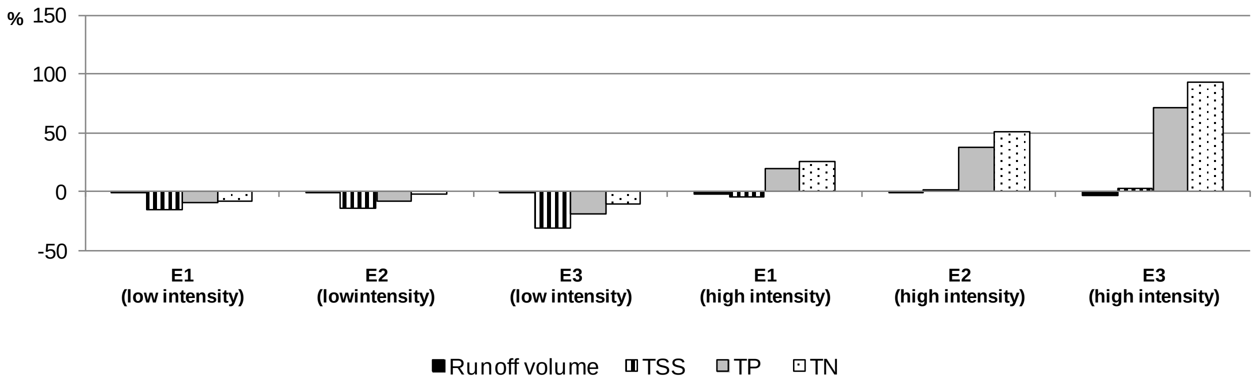


Figure 6.5 Change in estimated average annual run-off volume, TSS load, TN load, and TP load relative to the baseline scenario (%) for (a) Pongamia and (b) Eucalypt species.

This was accompanied by the least disturbance to native vegetation (0.6% reduction in CA, and very little change to LPI) (Figure 6.6) and a slight increase (1.6%) in the actual habitat amount (Figure 6.7). However, these biodiversity indicators require careful and integrated interpretation. Further spatial pattern analysis indicated a slight loss (1.1%) of native vegetation and ecosystems of high conservation status (i.e. ‘endangered’ and ‘of concern’ status on the RE maps) in this scenario (Figure 6.8). Thus overall, conversion of open grazing areas to woody bioenergy crops can benefit many species, but it is unlikely to support species in decline that are vulnerable to native vegetation loss and fragmentation. The same interpretation applies to scenario P1.

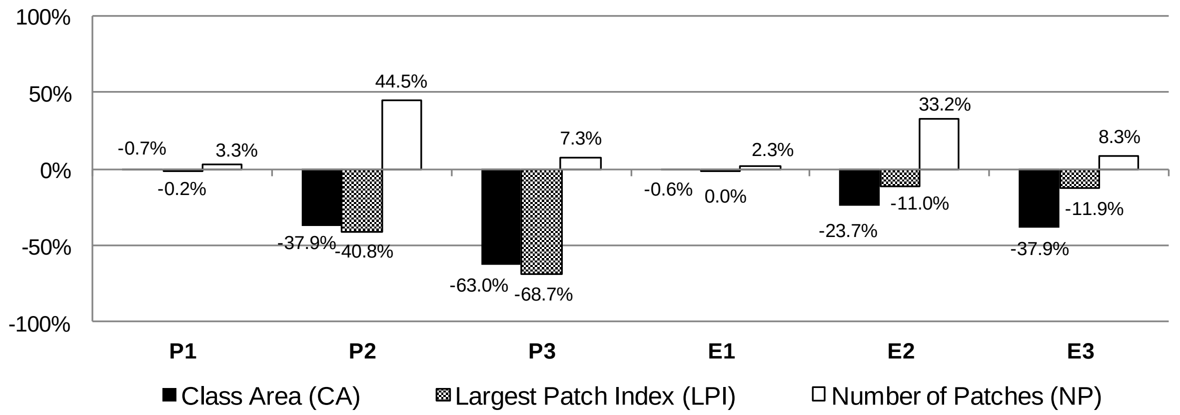


Figure 6.6 Change in Class Area (CA), Largest patch Index (LPI) and number of patches (NP) of native vegetation areas relative to the baseline scenario (%).

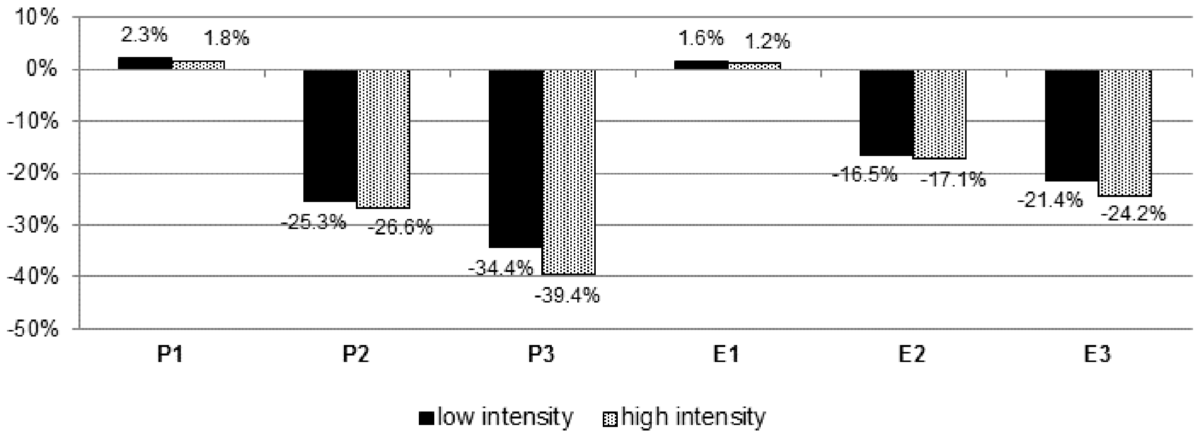


Figure 6.7 Change in actual habitat amount (%) relative to the baseline scenario (%).

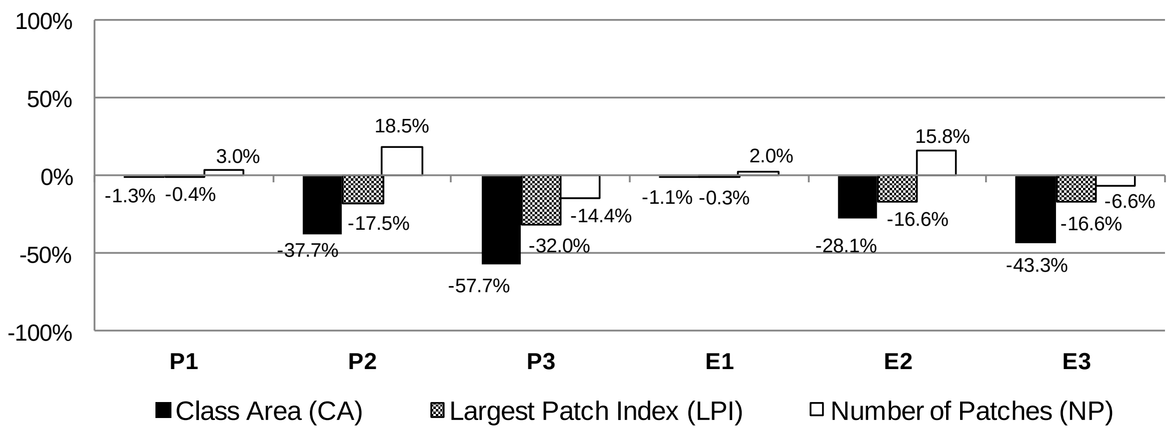


Figure 6.8 Change in Class Area (CA), Largest patch Index (LPI) and number of patches (NP) of native vegetation with high conservation status relative to the baseline scenario (%).

6.3.4 Pathways that reduce environmental quality

All high management intensity options resulted in an increase in the pollutant loads to receiving waters. Scenario P1 was the most intensive management option and increased TP (19.2%) and TN (24.9%) loads in waterways in spite of a slight reduction in TSS (-4.7%). In terms of TN loads, all Pongamia scenarios had definite advantages over eucalypt scenarios since the crop requires no or much less N fertiliser input. TP and TN loads could increase by over 50% in scenarios based on Pathway 2 and over 100% in Pathway 3 for both crops. These examples include 54.1% and 105.5% increase in TP in P2 and P3, and 50.5% and 92.4% increase in TN in E2 and E3 (Figure 6.5).

Irrespective of the management intensity, scenarios based on Pathway 2 (P2 and E2: conversion of ‘underutilised forested grazing land’) and Pathway 3 (P3 and E3: conversion of all potential ‘underutilised agricultural lands’) produced less favourable outcomes. This is attributed mainly to the native vegetation loss associated with the large scale land use change that is envisaged in these scenarios (Figure 6.6). The largest reduction in total native vegetation (CA) (63%) occurred in Scenario P3. In comparison, the eucalypt scenarios showed a smaller reduction in CA of native vegetation (e.g. -37.9% in E3) (Figure 6.6), since a smaller land area was subject to land use change. Almost one third (scenarios based on Pathway 2) or one half (Pathway 3) of ‘endangered’ and ‘of concern’ native vegetation and ecosystems were lost in the case study catchment (Figure 6.8). As a consequence, considerable habitat loss (partly fragmentation) is predicted for Pathway 3 due to a combination of the following effects (Fahrig, 2003): substantial reduction in CA (-63.0% [P3], -37.9% [E3]), substantial reduction in LPI (-68.7% [P3], -11.9% [E3]), and a slight increase in NP (7.3% [P3], 8.3% [E3]) (Figure 6.6). On the other hand, the spatial pattern of scenarios based on Pathway 2 can be described as habitat fragmentation rather than habitat loss per se, which involved a decline in both CA (-37.9% [P2], -23.7% [E2]) and LPI (-40.8% [P2], -11.0% [E2]), and a significant increase in NP (44.5% [P2], 33.2% [E2]).

Despite the potential destruction of wildlife habitats in Pathway 2 and 3, the large-scale conversion does not alter the catchment hydrology significantly since woody perennial crops such as Pongamia and eucalypts can replace the woody vegetation cover in a different form. The estimated average annual run-off volume of all scenarios showed little decrease (±2%), and more importantly, under low management intensity, the water quality indicators showed improvements in these scenarios. Scenarios based on Pathway 3 presented even better outcomes than those based on Pathway 2 in relation to these water quality indicators. For example, E3 showed the largest decrease with -30.7% in TSS, -18.9% in TP, and -11.2% in TN loads on a catchment scale in the low management intensity option (Figure 6.5).

All results showed evidence that the high management intensity option could degrade water quality, in particular in sub-catchments where a high proportion of land is changed to bioenergy crops. TP and TN loads could increase by over 50% in scenarios based on Pathway 2 and over 100% in Pathway 3 for both crops. These examples include 54.1% and 105.5% increase in TP in both P2 and P3, and 50.5% and 92.4% increase in TN in E2 and E3 (Figure 6.5). The sub-catchment scale analysis also showed that the increase of these pollutants could exceed more than 100% in a few sub-catchments (e.g. sub-catchment B16) (Appendix Figure A6.3[g], A6.3[g] and [h]).

6.3.5 Comparison with region-specific threshold values on water quality and biodiversity

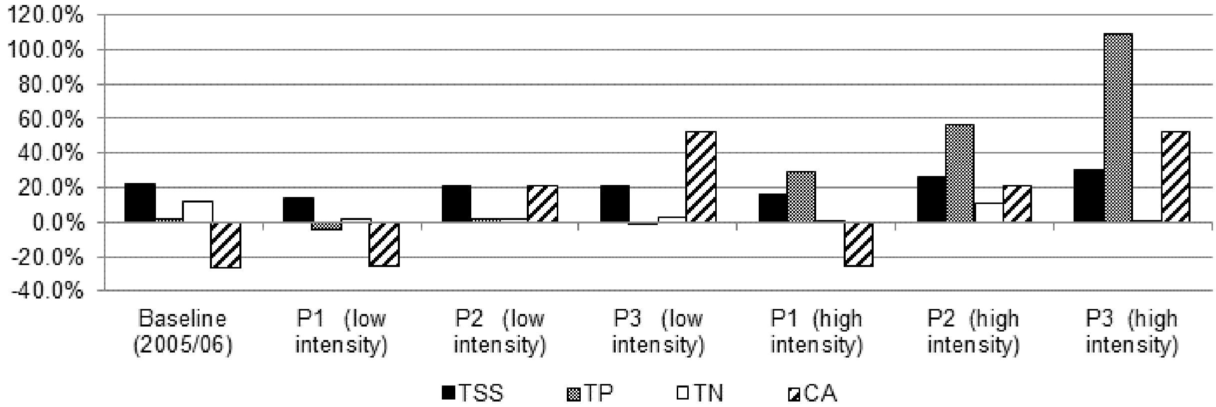
The results were compared with region-specific threshold values for water quality and biodiversity for the case study regions. These thresholds were based on water quality targets under the Reef Water Quality Protection Plan 2009 (Reef Plan) (Queensland Department of the Premier and Cabinet 2009b), and thresholds used for remnant (native) vegetation management by Queensland Government.

For the Burnett River catchment, there are pollutant reduction targets on TSS (20% reduction by 2020), TP and TN (50% reduction by 2013) under Reef Plan (Queensland Department of the Premier and Cabinet 2009b). The method for load reduction calculation was specified under the Paddock to Reef Integrated Monitoring, Modelling and Reporting Program (Paddock to Reef) (Waters & Caroll 2013). In this analysis, results from the 2005/06 scenario were used as the reference case to calculate a 'baseline' for the load reduction calculation (current [2005/06] – pre-development) and targets on each pollutant, due to differences in models and scenario assumptions. Those target values were compared with results from the six land use change scenarios.

In regards to biodiversity, literature on landscape ecology (Fahrig 2001) supports the concept of an extinction threshold between survival probability and habitat amount (e.g. native vegetation cover). Although there is no common threshold value across species, thresholds ranging from 10 to 30% have been suggested by Andrén (1994) and used widely for conservation and habitat protection planning (Radford, Bennett & Cheers 2005), including by the Queensland Government (Table A5.4). For this reason, the threshold value of 30% was used for remaining remnant vegetation areas (i.e. sum of CAs of remnant vegetation) to the total catchment area. The results were expressed in graphs, setting the threshold values as zero percentage (0%), and showing the results for the six land use change scenarios as a percentage (%) in meeting (-) or exceeding/not meeting (+) the thresholds (Figure 6.9).

Overall, there was no single bioenergy land use change scenario that could meet all thresholds simultaneously, even water quality targets specified in the Reef Plan. This indicates clearly that significant efforts in land management practices in all agricultural sectors are required to meet the specified targets. Nevertheless the results confirmed the findings in the earlier sections, and highlighted the advantages of the P1 and E1 scenarios that could help to achieve the environmental objectives of the region. However, it must be emphasised that the land use change scenarios must also be accompanied by the best agricultural management practice (i.e. low intensity management). Results indicated that these scenarios could achieve better outcomes than the baseline (2005/06) for most indicators, except for a minor decline (0.9% for P1, 0.7% for E1) in the CA of remnant vegetation. However, such losses could be prevented relatively easily by careful planning of bioenergy plantations. The results also confirmed that other scenarios are not recommended because the potential impacts are significant (exceeding 20%) in one or more indicators, or would not help to achieve the environmental objectives in the region.

(a) Pongamia



(b) Eucalypts species

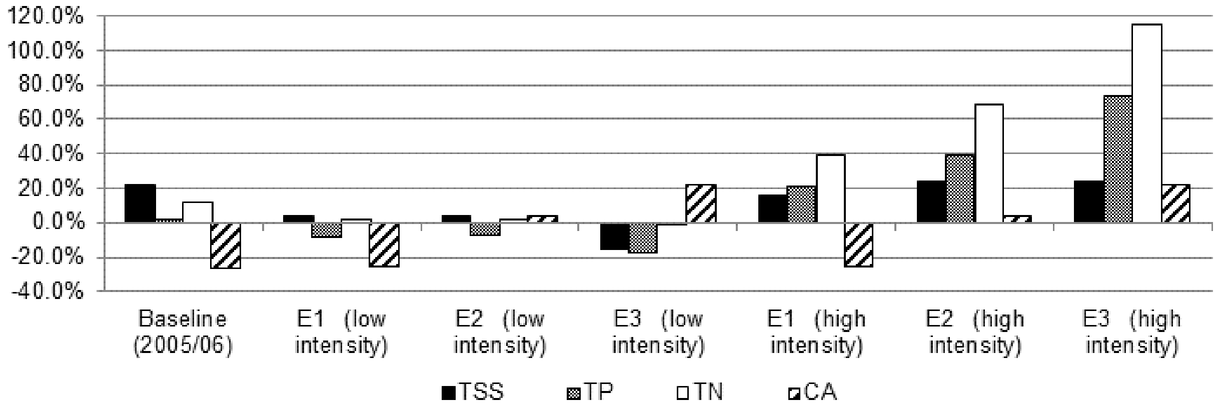


Figure 6.9 Comparison with threshold values in the Burnett River catchment. The threshold values were set as zero percentage (0%), and the results for the six land use change scenarios are shown as a percentage (%) in meeting (-) or not meeting (+) the thresholds.

6.4 Discussion

This is the first study to evaluate the potential regional-scale water and biodiversity effects of using 'underutilised agricultural land' for bioenergy production, using a spatially explicit modelling approach. While use of these lands has been suggested (e.g. the UK's 2009 Renewable Energy Strategy by HM Government 2009), no study has attempted to model, quantify and compare them with current or different land use change scenarios to date. In short, the use of 'underutilised agricultural land' for bioenergy crop production could be positive or negative, and the following suggestions and recommendations were derived from the findings.

6.4.1 Policy implications for bioenergy production on 'underutilised agricultural land'

A policy direction of simply encouraging bioenergy crop production on underutilised agricultural land will not necessarily result in improved environmental sustainability. To be effective, future bioenergy policy requires a guideline that includes more detailed prescriptions concerning the use of underutilised agricultural land. This could be achieved through very careful planning and management strategies. Emphasis should be placed on: (i) the selection of the most sustainable land use change pathways and scenarios; (ii) stringent controls on native vegetation clearing and consideration of local ecosystems; and (iii) the identification and implementation of 'optimum' management strategies that result in the least impact.

Key strategy 1: selection of the most sustainable land use change pathways and scenarios

As crop and land use change pathways vary significantly from one country/region to another, bioenergy crop production using underutilised agricultural land may involve a wide variety of land use change scenarios. In this evaluation, crops were carefully selected for the case study area from woody perennial native species. The results, however, suggested that pollutant loads from those crops can differ significantly depending on the site management. They were generally in concert with previous studies on the hydrological effects of woody perennial bioenergy production, such as SRC poplars and willows in Europe (Dimitriou et al. 2009; Langeveld et al. 2012) and dryland salinity mitigation in South Australia's mallee production (Bryan, Ward & Hobbs 2008). These studies reported an increase in water use, a reduction in surface run-off and soil erosion, and an improvement in water quality through reduction in sediment and nitrogen loads. Compared to the hydrological and water quality effects, the biodiversity benefits of the bioenergy crops incorporated in the scenarios were largely unknown from existing studies. However, bioenergy crops should be selected from species native to the region in order to protect and even enhance native biodiversity and ecosystems, particularly in the Australian context.

Designing land use change pathways also requires consideration of several factors and involves trade-offs. The land use change pathways in this study were developed in line with new production models proposed by existing Australian research for the integration of Pongamia and eucalypts into grazing landscapes (Shepherd et al. 2011b; Murphy et al. 2012). The results indicated that the conversion of underutilised agricultural land must be limited to open grazing land/pasture to ensure environmental benefits, and grazing land with native vegetation cover should be excluded from future bioenergy crop production.

The introduction of bioenergy crops into grazing-dominated rural landscapes can provide socio-economic and environmental benefits. For example, agroforestry systems (the integration of southern mallee eucalypt production in the dryland wheat and sheep regions) of South Australia (SA) and Western Australia (WA) was designed to remedy multiple natural resource management problems (e.g. salinity induced by dryland agriculture, biodiversity loss, water quality degradation), increase farmers' profits, reduce the risks to producers through product diversification into biomass and oil (Shepherd et al. 2011a; Harper et al 2009), and to explore carbon-offset opportunities (Bryan, Ward & Hobbs 2008; Harper et al 2009). These projects also indicated that well-planned and integrated native woody perennial bioenergy production in a grazing landscape has potential to provide successful outcomes in rural Australia. This could include major benefits at the regional level in terms of land remediation, and rural and regional development, but also at the national level in terms of potential for improved fuel security, significant savings in the future cost of imported oil and GHG mitigation (Farine et al. 2012). In subtropical Australia, the bioenergy industry based on planted eucalypts and other woody perennial crops has not been developed to extent of that for SA and WA. However, the basic principles of the production system for mallee eucalypts can be applicable to other regions, including the case study region.

Key strategy 2: Stringent controls on native vegetation clearing and careful consideration of local ecosystems

The results confirmed the importance of stringent control and careful planning to prevent native vegetation clearing for crop expansion, including forested grazing areas. This is perhaps the most challenging issue, especially in developing regions, where political, institutional, and enforcement capabilities are limited (Miyake et al. 2012). Even in the case study region in Australia where relatively effective vegetation management laws have been enacted, they have not guaranteed to prevent all clearings. Regardless of the *Vegetation Management Act 1999*, 1.7 million ha of woody remnant vegetation was cleared between 2001 and 2011 in Queensland (DEHP 2014) under certain conditions, such as clearing of regrowth and certain species (e.g. leguminous acacia) and illegal clearing (McAlpine et al. 2009). Moreover, the existing governance arrangements are unlikely to guarantee stringent control under the recent political climate in Australia. The Queensland Government has recently created more exemptions to facilitate clearance of native vegetation on lands which are suitable for 'high-value' and 'irrigated high-value' agricultural activities. Those include broad scale cropping (e.g. sugarcane), annual horticulture, perennial horticulture and irrigated pastures (Queensland Government 2013). Plantation forestry is not included in this exemption. Nevertheless, bioenergy plantations may be considered as high-value' agriculture once greater demand for bioenergy production emerges. Similar amendments on native vegetation clearing regulations were also made in WA, which allow farmers to clear native vegetation for farm management and infrastructure projects (Government of Western Australia 2013). Therefore, the evaluation of extreme scenarios (as envisaged in Pathway 3) is worthwhile to consider especially in the Australian context as it highlights the impacts directly caused by the replacement of native vegetation with the bioenergy crops.

First, the results indicated where native vegetation clearing occurred to establish woody perennial bioenergy plantations there were no significant alterations to the regional hydrology, because the plantation replaced the lost native vegetation cover in a different form. Therefore the water quality indicators did not show any negative impacts provided the management intensity was low. Nevertheless, a monoculture bioenergy plantation, including eucalypt plantations, will be unable to provide the same high level ecological functions as the primary vegetation. Hence they are highly unlikely to offer a solution for the conservation of threatened species that are currently constrained to limited areas of remaining native vegetation patches in wider landscapes.

Second, the results confirmed the following evidence for biodiversity conservation. The land use change pathway from beef cattle pasture on cleared land to woody perennial bioenergy crops can provide an ecological restoration benefit with additional habitats and opportunities for the movement of many species between habitats. The benefits however, are likely to be limited to 'generalists' or 'open or developed land' species. In the subtropical Queensland context, existing eucalypt plantations for hardwood production can support some native mammal species, such as possums, gliders, macropods and bats to a certain degree provided the plantation has matured (Smith & Agnew 2002). Compared to pastures, future SRC eucalypt plantations can contribute to restoration of habitat for these species by re-connecting the fragmented remnant vegetation patches. The benefits obtained would depend on the planning and management strategies on site. However, due to the highly intensive management characteristics of SRC eucalypt plantations, such as short harvesting cycle and extensive residue removal, the same degree of conservation benefits as the conventional forestry plantation cannot be expected from the SRC eucalypt plantations. Another important biodiversity consideration was the potential invasiveness of the crops. For example, Spotted gum is a very common native tree species in the region, however Pongamia is not commonly found in native vegetation in subtropical Queensland. Thus the ecological implications of large scale production are largely unknown and need to be investigated in future research.

Key strategy 3: identification and implementation of 'optimum' management strategies with the least impacts

There was a clear contrast of outcomes between high and low management intensity options in the case study. This evaluation applied value ranges in hydrological parameters and biodiversity conservation values (BCV_{LU}) due to a lack of data and in consideration of the large degree of variety and uncertainty. The 'optimum' management strategy for these bioenergy plantations needs to be identified through ongoing and future long-term field trials, including the harvesting cycle and method, proportion of residue removal, irrigation system, chemical inputs (amount and frequency), and other ancillary human activities such as grazing and logging. In fact, such field trials have started recently in North America and Europe for bioenergy plantations (e.g. SRC) (Bouget, Lassauce & Jonsell 2012; Langeveld et al. 2012; Lattimore et al. 2013), but few have been conducted in Australia. Moreover, the management strategies for bioenergy crop production on underutilised agricultural land are also unknown. The crop production on these lands may require additional inputs to raise land productivity compared to productive agricultural land. Thus 'optimum' management strategies of these crops on underutilised agricultural land might be significantly different from the assumptions in this analysis, but this needs to be understood and evaluated rigorously in future research. As a result, relevant standards and operational guidelines on management strategies for bioenergy plantations must be developed to incorporate the research findings from these field trials. They need to be linked with the international certification schemes and sustainability requirements for bioenergy production.

6.4.2 Potential role of the evaluation framework in regional land use planning

Planning strategies for bioenergy plantations require consideration of a number of local-specific conditions such as climate, topography, and soil quality. To enable the most environmentally sustainable outcomes, the bioenergy crop production plan will need to be incorporated into regional land use planning strategies. The planning and decision making process must include a wide range of stakeholders, such as communities, agricultural industry groups, catchment management groups, regional and local government departments, and various environmental experts. In the Australian context, the regional natural resource management organisations (e.g. the Burnett Mary Regional Group [BMRG] for the case study catchment) are pivotal in any future processes aimed at integrating consideration of bioenergy crop production into regional planning processes.

While the current evaluation framework focuses on a minimum number of critical environmental indicators, it has the potential to be developed to enable more comprehensive future applications. It offers a basis for an applicable methodology for land use change scenario evaluation suitable to various regions with slight modifications, and facilitates decision making on bioenergy land use. It results in maps and radar charts that enable clear presentation of land use change options. The need for such an evaluation framework is becoming more important, as bioenergy-driven land use change and related decision making is increasing worldwide. In future applications of this evaluation framework, the indicators can be re-selected to reflect specific regional situations, the varied characteristics of bioenergy systems, the range of stakeholders and their priorities, diverse regional environments, and differing scales of application (Efroymson et al. 2013). Currently, economic and social indicators are not included in the evaluation framework, but it can be linked to those indicators such as biomass yield in the future. The indicators can be also expanded to a range of other regional-scale environmental indicators such as air, soil, biodiversity and other water quality indicators (e.g. herbicide and pesticide loads) that can be linked to spatial information. The future development of this evaluation framework can be undertaken in line with the goals of sustainability requirements for bioenergy production in various regions.

6.4.3 Limitations and priorities for future research

This evaluation incorporated assumptions gained from the literature and expert judgements to overcome the lack of actual field data and a large degree in uncertainties associated with new crops and their bioenergy products, land use changes, and the associated impacts in the subtropical Australian context. However, field trials for production of the evaluated crops are underway. Since 2008, several Pongamia sites have been established across the region by the ARC Centre of Excellence for Integrative Legume Research (CILR), at The University of Queensland (UQ), (P. Scott 2012, pers. comm., 29 November). Numerous field trials for Spotted gum have been undertaken in subtropical Queensland. While the majority of trials have been designed for conventional hardwood and fibre production (Lee et al. 2010; Shepherd et al. 2011a), a field trial of SRC eucalypts for biofuel was launched in 2012 by the Queensland Alliance for Agriculture and Innovation (QAAFI), UQ, under their 'Future Biofuels' research program (R. Henry 2012, pers. comm., 18 December). The current trials have various objectives, such as species selection for commercial production and identification of the management strategy and bio-product processing. Thus in the coming years ecological and hydrological data will be collected. This will help to improve significantly the overall reliability of the estimates of bioenergy products and the production system of these crops, and also the quality of the parameters and results from hydrological modelling using *Source* as well as biodiversity conservation values (BCV_{LU}).

The second limitation of this study relates to the land use change scenarios that were assessed. Firstly, it would be valuable for future bioenergy sustainability research to have a common method of identifying these lands because the term has a wide range of definitions and interpretations (e.g. Shortall 2013). In this analysis, the original definition of underutilised agricultural land was created for the Australian context, and the identification relied on a land capability study previously prepared for the region. However, establishing a common and/or alternative method in future research would be valuable for regions especially where previous land suitability/capability studies are not available. Secondly, the land use change scenarios used for this analysis may be updated as more reliable information becomes available. There was a high level of uncertainty in terms of developing 'realistic' bioenergy-driven land use change scenarios for the case study region, without a clear national/regional bioenergy/biofuel policy and plan, economic analysis and field based environmental data concerning these crops. Updating will enable estimate results that are aligned with the actual bioenergy land use plan of the region, and this will be valuable for decision makers. As a result, the scenarios may be more varied and complex than those identified as land use change pathways in this research. For example, this research excluded scenarios using lands suitable for cropping due to the potential conflicts with land use for food production. However, land use in agroforestry systems, for example, could have better environmental outcomes by fixing multiple natural resource issues caused by cropping activities, and those were suggested in the experiences of eucalypt production in other states in Australia (i.e. SA and WA). Those possible land use changes will also have to be evaluated in future research.

Lastly, the scope of the environmental impacts considered in this evaluation was limited, being specifically targeted to the direct land use impacts at the regional scale. As a result, the complex dynamics of bioenergy-driven land use change and their impacts at larger spatial scales, such as indirect land use change (iLUC), were not considered in this evaluation framework. iLUC is a critical research area in the bioenergy sustainability field, despite its high complexity and uncertainty, and the difficulty of reaching consensus among the scientific community (Prins et al. 2010; Di Lucia, Ahlgren & Ericsson 2012; Wicke et al. 2012). For example, the land use change pathway/scenario from 'underutilised' pasture to future bioenergy crop production suggested in this evaluation may involve displacement of grazing to previously ungrazed areas with associated environmental impacts. In reality, indirect deforestation in the Amazon and the Cerrado in Brazil has been attributed to the recent expansion of sugarcane and soybean on abandoned or degraded pasture in the southern states (e.g. Barona et al. 2010; Lapola et al. 2010; Loarie et al. 2011). Large-scale clearing experienced in Brazil (or extreme scenarios as envisaged in Pathway 3) is less likely to occur in the Australian context (although future policy directions may change), but it may occur in other countries. The dynamics of this land use change pathway/scenario should be further investigated using alternative methods to avoid adverse consequences including iLUC and to ensure better environmental outcomes from these scenarios.

6.5 Summary

An environmental sustainability evaluation framework was applied to three bioenergy land use change pathways or six scenarios in the Burnett River catchment, Queensland, Australia. Environmental outcomes, as identified by eight indicators, were quantified and compared with a baseline scenario (2005/06 land use) and with current thresholds values for water quality and biodiversity for the case study region. The scenarios were based on native woody perennial bioenergy crops with high potential for bioenergy production. These included Pongamia and two eucalypt species - Spotted gum and Chinchilla white gum. The land use change scenarios embraced production of those crops on existing 'underutilised' open grazing areas (pastures) (Pathway1), 'underutilised' forested grazing areas (Pathway 2), or all 'underutilised agricultural land' (Pathway 3) for high and low management intensity options. These scenarios were developed in the context of subtropical regions of Australia, and will certainly differ from scenarios appropriate for other countries and regions.

The results indicated that the production of bioenergy crops on 'underutilised agricultural land' could be environmentally positive or negative. Pathway 1 was the only land use change pathway that could possibly benefit hydrology, water quality and vegetation habitat qualities in the case study region. These benefits would be expected only when: (i) open grazing areas (pastures) were used; (ii) native woody perennial bioenergy crops were planted; and (iii) the new plantations were under low intensity management. Scenarios based on Pathways 2 and 3 did not suggest any favourable outcomes in terms of environmental sustainability. Two factors that directly affected the environmental outcomes were: (i) crop extent; and (ii) management intensity. The first factor was correlated with the area of native vegetation and habitat loss, and the second factor was associated closely with water quality outcomes. Such findings flagged that current policy—simply limiting bioenergy crop production to underutilised agricultural land—will not necessarily result in improved environmental sustainability. Thus future policy should provide more detailed prescriptions concerning the use of underutilised agricultural land for future bioenergy crop production. The results also flagged the importance of careful planning and management strategies, since a well-planned and integrated bioenergy industry can have major benefits for a region in terms of environmental sustainability, but it requires a number of conditions to be satisfied.

Chapter 7: CONCLUSION

This chapter highlights the key research findings and contributions in relation to the research questions, aim and objectives (Chapter 1). The limitations are briefly addressed and key areas for future research are discussed.

7.1 Research questions, aims and objectives

Two questions that this research addressed were:

1. How, and to what extent will conversion of land from existing land uses to bioenergy agriculture affect environmental qualities at the regional spatial scale?
2. Compared to current land uses, what are the environmental impacts of using 'underutilised agricultural lands' to produce bioenergy crops, and how and under what conditions can the conversion of these lands for crop production deliver better environmental sustainability outcomes?

This research aimed to evaluate whether land use change scenarios that involve bioenergy crop production on underutilised agricultural land can enhance regional-scale environmental outcomes when compared with current land uses, and to provide recommendations and suggestions for future land use options. The research questions and aim were directly translated into four research objectives.

7.2 Research findings

The key findings and/or outputs are presented in relation to the four research objectives.

Objective 1 (incorporated in Chapter 2): *Review the environmental and land use pressures resulting from the global increase in bioenergy production.*

The review identified that the environmental effects from land use changes differed among countries and regions, and were dependent on a range of factors (e.g. geophysical, political, institutional and legal). However, land use change pathways and site management practices were key factors that affected environmental sustainability outcomes. The review synthesised the attributes of bioenergy-driven land use changes and their consequences occurring on a global scale. The recent increase in demand for biofuel (mostly in the U.S.A. and EU) was found to have impacted on ‘land- and resource-abundant’ developing regions, such as Brazil, and the future increase in the global demand for biofuel was identified as an ongoing threat to these countries. The political and institutional dilemmas for these developing regions revolved around their strong emphasis on economic development policies/private interests over environmental policies, and their limited law enforcement capacity. This led to expansion of agricultural land (e.g. cattle pasture, cropland) and consequent deforestation. To prevent further impacts associated with bioenergy-driven land use changes, the review identified the imperative for:

1. prioritising ‘waste and residue’ feedstock and associated technology to avoid demand for agricultural/forestry land use;
2. developing sustainable land use options for bioenergy crop production, including the conversion of ‘underutilised agricultural land’ to non-food bioenergy crops;
3. developing/utilising agreed international policy mechanisms and instruments, e.g. international climate change policy such as REDD under the UNFCCC, that can also potentially mitigate impacts of bioenergy-driven land use change; and
4. strengthening sustainability requirements and certification schemes, despite challenges and uncertainties around current developments in certification criteria.

The findings from the review, especially concerning the environmental effects, provided a context for the development of an environmental evaluation framework for bioenergy-driven land use changes (Objective 2).

Objective 2 (incorporated in Chapter 3): *Develop a framework for evaluating the environmental consequences of bioenergy-driven land use change at the regional scale.*

The main outcome of this objective was a spatially explicit framework to evaluate changes in regional-scale environmental qualities associated with several land use change scenarios. The framework was designed for application to diverse regions to meet the need for more effective and sustainable land use decision making in relation to future bioenergy crops. From the information gathered in the review, the regional-scale environmental consequences arising from bioenergy-driven land use changes were identified to include air and water pollution, regional climate change, soil erosion, change in hydrological regimes, water extraction, introduction of exotic species, and change in vegetation and habitats for native fauna species. However, the evaluation framework in this research was scoped to focus on water quantity and quality, and terrestrial biodiversity, as these were identified as the key environmental qualities positively or negatively impacted by bioenergy-driven land use changes.

Eight indicators were identified, including: four for water quantity and quality—runoff volume, sediment (TSS), nitrogen (TN) and phosphorous (TP); and four for terrestrial biodiversity—total area (CA), size (LPI) and number (NP) of native vegetation patches, and biodiversity conservation value indicators. This was followed by selection and design of the methods to quantify those indicators. As a result, methods using *Source*, a spatial hydrological modelling platform, and *Fragstats*, a spatial pattern analysis tool for evaluating native vegetation communities, were packaged into the evaluation framework to quantify the first seven indicators. A new method was developed to quantify 'actual habitat amount' using biodiversity conservation values corresponding to each land use class (BCV_{LU}), based on the European literature and expert opinion.

Objective 3 (incorporated in Chapter 5): *Test the effectiveness of the environmental evaluation framework by applying it to a selected region that has experienced land use changes due to crop production.*

The evaluation framework was applied to spatial land use data (1992/93, 2001/02 and 2005/06) representing past land use change in the Burnett River catchment in Queensland, Australia. The results indicated that the evaluated environmental qualities of the catchment degraded between 1992/93 and 2005/06, particularly during the first decade (i.e. 1992/93-2001/02) due to vegetation removal for agricultural expansion (especially grazing and dryland cropping). Native vegetation removal for agricultural expansion in the catchment resulted in higher sediment and nutrients loads, particularly TN, between 1992/93 and 2001/02. After 2001/02, a decrease in the vegetation loss rate and in 'dryland cropping' contributed to an improvement in water quality indicators in 2005/06. By comparing the modelled outputs with trends gained from an examination of the literature, existing data and interviews with local experts (Chapter 4), the results from the application of the evaluation framework to this region, indicated close alignment with the reported trends in land use, vegetation, water quality and biodiversity over this period. As the evaluation framework effectively captured the main characteristics of the region's environmental challenges and its changes during the period, it has the ability to provide reasonable predictive capacity in relation to future environmental impacts from land use change within the case study catchment.

The strengths of the evaluation framework were its effectiveness in providing a good overview of the environmental changes associated with land use changes of the case study region, and its flexibility, lack of complexity and logical structure. The framework's flexibility enables it to be customised for future applications addressing particular local circumstances. On the other hand, the application of the framework may be limited to some extent by the availability of appropriate parameters for the models utilised in the *Source* platform. However, intensive efforts to enable wider application of *Source* to catchments outside of Australia are currently in progress, and are expected to help solve several issues associated with the future application of the framework.

Objective 4 (incorporated in Chapter 6): *Develop land use change scenarios that incorporated the use of 'underutilised agricultural land' in a case study region, and apply the evaluation framework to evaluate the regional scale environmental consequences of the scenario in relation to current land use.*

The evaluation framework was applied to six land use scenarios (based on three land use change pathways) in the Burnett River catchment. Environmental outcomes, as identified by eight indicators were quantified and compared with a baseline scenario (2005/06 land use) and threshold values currently used in the case study region. The scenarios were based on native woody perennial bioenergy crops with high potential for bioenergy production. These included Pongamia and/or eucalypt species, such as Spotted gum and Chinchilla white gum. The land use change scenarios embraced production of those crops on existing 'underutilised' grazing open areas (modified pastures) (Pathway 1), 'underutilised' grazing forested areas (Pathway 2), or all 'underutilised agricultural land' (Pathway 3) for both high and low management intensity options. These scenarios were developed in the context of subtropical regions in Australia, and would need to be re-designed to enable evaluations in other global regions as they may differ between countries and regions depending on local conditions.

The results indicated that the use of 'underutilised agricultural land' for bioenergy crop production could be environmentally positive or negative. The only land use change pathway that would potentially benefit regional scale environmental qualities in the case study region was Pathway 1. These benefits would be expected only when:

(i) open grazing areas (pastures) were used;
(ii) native woody perennial bioenergy crops were used; and
(iii) the new plantations were under low intensity management.

The other scenarios based on Pathways 2 and 3 did not suggest any favourable outcomes in terms of environmental sustainability. Two factors that directly affected the environmental outcomes were: (i) land areas/proportions of land converted; and (ii) management intensity. The first factor was correlated with the area of native vegetation and habitat loss, and the second factor was associated closely with water quality outcomes.

Bioenergy crops are proposed as an energy-efficient alternative to fossil fuels. These findings indicate the need for careful policy planning in their introduction if they are to provide better environmental outcomes. Simply encouraging bioenergy crop production on underutilised agricultural land without careful selection of land and crop management strategies may result in poor environmental outcomes. The findings indicate that future policy should provide more detailed prescriptions concerning the use of these lands for future bioenergy crop production. The results also flag the importance of careful planning and management strategies, since a well-planned and integrated bioenergy industry can have major benefits for a region in terms of environmental sustainability. However, a number of conditions need to be satisfied. These include the selection of the most sustainable land use change pathways and scenarios, stringent control of native vegetation clearing, and identification and implementation of optimum management strategies on site (e.g. harvesting cycle and method, and application of fertilisers).

7.3 Contributions of the research to knowledge gaps

Contribution 1: *Providing evaluations and recommendations for more environmentally sustainable land use change options using 'underutilised agricultural land' for future bioenergy crop production.*

The research findings contributed to knowledge in bioenergy sustainability research in both the international and Australian contexts. This research is the first attempt to quantify the potential environmental effects of bioenergy crop production on 'underutilised agricultural land' in a catchment/regional context using a spatially explicit modelling approach. While the potential benefits and/or impacts of using underutilised agricultural land for bioenergy crop production have been discussed in a number of bioenergy studies (e.g. Campbell et al. 2008; Odeh, Tan & Ancev 2011), very limited attention has been given to the impacts of using these lands for bioenergy production. Very few studies have attempted to model, quantify, and compare the environmental outcomes with current practice or other land use change scenarios. In this context, this research has made a substantial contribution by providing and testing an evaluation methodology or framework, generating quantified results for scenarios in a case study region, and providing recommendations for future bioenergy land use options.

The research findings have answered two research questions. The first question regarding information on the environmental consequences of bioenergy-driven land use change was achieved by reviewing existing studies on bioenergy-driven land use changes (Objective 1). and then identifying the key environmental consequences at the regional-scale, which was the focus of this evaluation framework (Objective 2). The recent global trends in environmental and land use policy and decision-making have indicated a greater reliance on decentralised arrangements at the local and regional scales. Thus this framework was designed to provide an evaluation methodology for assessing land use change and environmental impacts and thus to support policy and/or decision makers at the regional level in relation to future bioenergy land use options. To answer the second research question, the methodology or framework was established (Objective 2) and tested against the existing data (Objective 3). Using the methodology, this research quantified the environmental outcomes of six land use change scenarios, which incorporated the production of bioenergy crops on underutilised agricultural land (Objective 4), and then identified the circumstances under which the use of underutilised agricultural land for bioenergy crop production can result in better environmental sustainability outcomes (Objective 4).

The findings from the evaluation suggested that woody perennial bioenergy crop production on underutilised agricultural lands could reduce environmental impacts compared to existing land use. However, the research indicated that to achieve these successful outcomes, careful planning and management strategies were required. These included: (i) selection of the most suitable crops and site location (i.e. the land use change pathway most suited to a region); (ii) stringent control of native vegetation clearing and consideration of local ecological characteristics (e.g. appropriate distance from watercourses and conservation areas, spatial configuration of native vegetation, and ecology of local fauna); and (iii) identification and implementation of the optimum management intensity. This decision making process should also involve careful consideration of 'trade-offs'. The findings suggested that the 'balanced' scenario (i.e. a scenario that can provide a radar chart with a smaller area) should be favoured in future implementation. Thus this research recommended that simply limiting bioenergy crop production to underutilised agricultural land would not necessarily result in enhanced environmental sustainability, and future policy should include more detailed prescriptions about how to make effective use of underutilised agricultural land for bioenergy crop production. This research identified these prescriptions from the case study evaluation (Objective 4). These are perhaps its most significant findings, and have added important knowledge to current bioenergy research and policy. The prescriptions may, however, vary according to the land use change scenario, which means they may vary regionally.

The findings from this research are also beneficial to the bioenergy research community in Australia. To date, a few notable spatially explicit modelling studies have been conducted on potential bioenergy crop production in the Australian context. Odeh, Tan and Ancev (2011) investigated the potential suitability and economic viability of biodiesel crops (Pongamia and Indian Mustard [*Brassica juncea*]) using marginal agricultural lands at the national scale. The CSIRO (Bryan, Ward & Hobbs 2008; Bryan, King & Wang 2010) undertook research in the lower Murray region of southern Australia to quantify the economic viability, carbon emissions, salinity and wind erosion impacts from broad scale production of mallee eucalypt species. Nevertheless, their research aim, scope, and geographical area were different from those of this research. The focus of these studies was the economic viability of crop production (although the CSIRO assessment examined a few natural resource management impacts). They did not consider the impacts of water quantity and quality, and terrestrial biodiversity, which were the main focus of this research.

Moreover, compared to South Australia and Western Australia, a bioenergy industry based on eucalypts and other perennial woody species has not been developed yet in subtropical Australia. So far, little has been known about the production systems, site management, and environmental impacts of large scale production of eucalypt species and Pongamia that were considered in this research. There is still a high level of uncertainty in commercial scale production of these crops, but the results from this research can provide an important basis for further discussion and research into the development of an Australian bioenergy policy and industry. Due to the high level of uncertainty, the land use change scenarios in the case study region were developed mainly based on biophysical constraints, yet they attempted to take into account possible production models for the future (i.e. integration with grazing landscapes) suggested by Australian researchers (e.g. O'Connell et al. 2009; Farine et al. 2012; Murphy et al. 2012) based on the socio-economic and environmental challenges the Australia's rural sector is facing. In this way, this research has been built on the issues and gaps addressed in the existing research on these bioenergy crops and bioenergy sustainability, and has played a significant role in complementing existing knowledge. As a result, this research has advanced the existing knowledge base in renewable energy, agriculture, forestry, natural resource management, and land use policies in Australia, and this will be valuable for bioenergy researchers as well as relevant policy makers in the Australian and Queensland governments. In particular, one of the most important contributions to knowledge is about the implications and regional environmental outcomes of broad-scale production of these crops in the subtropical Australian context.

In addition, the findings and implications of this research are highly relevant and valuable not only in the Australian context, but also to other global regions outside of Australia where cattle grazing is predominant in the agricultural landscape. For example, South American countries, such as Brazil and Argentina have been facing serious land use and environmental pressures from crop expansion involving conversion of cattle pasture (Chapter 2). For these regions, the findings could serve as a reference case for more detailed evaluations of land use change impacts in order to develop appropriate land use policies to tackle these environmental challenges.

Contribution 2: *Developing and testing a spatially explicit framework for evaluating the consequences of bioenergy-driven land use changes.*

An important output and contribution of this research was the development of an evaluation framework. It provides a methodology applicable to various regions for land use change scenario evaluation, and forms an important basis for future decision making concerning sustainable land use options for bioenergy crop production. In particular, this evaluation framework aimed its application to regions experiencing high land use and environmental stress from bioenergy crop expansion, where an urgent solution is required to prevent further deforestation and degradation. Such regions often lack financial and technical resources, such as high level expertise. They are often unable to undertake extensive field trials, and may lack detailed environmental quality data that other assessment methodologies may require. This evaluation framework was designed to provide results with minimum inputs, as the emphasis of this evaluation framework was to enable future application to such regions.

In this regard, the advantages of this evaluation were found to be: (i) supported by solid scientific logic (spatial hydrology and spatial ecology); (ii) a flexible and less complex structure; (iii) fewer data inputs (in comparison with other methodologies); (iv) quantified results that enable easy interpretation; and (v) effective presentation/communication using maps and radar charts. A useful and new concept (i.e. the evaluation framework) was developed as it packages methods commonly used in spatial hydrology and spatial ecology for integrative interpretation of the results. In particular, quantifying biodiversity impacts was a shortcoming in frameworks suggested in past studies in the area of environmental impacts of bioenergy crop production, and this is the first study employing indicators and a method commonly used in spatial ecology (i.e. *Fragstats*). Moreover, the GIS-based modelling was integrated with visual presentation such as maps and radar charts. This is useful to interpret results and inform preferred land use scenarios/plans/options that can facilitate complex policy development and planning process involving many stakeholders. The need for an evaluation framework such as this presented here may be growing, as land use change related to bioenergy crops and related decision making is increasing at the regional scale worldwide.

In addition, this methodology is also applicable, with modification, to evaluate various type of land use change, including any type of crop expansion regardless of its original aim. For instance, the modified version of this evaluation framework can be applied to GBR catchments including the Burnett River catchment to facilitate future land use and natural resource management decisions in the region. The GBR lagoon and its ecosystem remain under extremely high land use and environmental pressures from various activities conducted in the coastal catchments. In the modified version, biodiversity and other relevant indicators that can be linked to spatial information will be added to the water indicators obtained from ongoing water quality modelling and monitoring studies under the Paddock to Reef program. The results will inform Australian and State Government departments involved in a range of GBR protection policies, and relevant organisations and NRM regional bodies, as it will enable the evaluation and reporting of changes in environmental outcomes in a more comprehensive and integrated way.

7.4 Research limitations

The limitations of this research relate mainly to two areas: (i) research scope; and (ii) the current high level of uncertainty in bioenergy policy and related lack of data.

Limitation 1: Research scope

The research scope was clearly defined (Chapter 1), with a focus on regional-scale environmental issues arising from direct land use change. This research identified the importance of understanding the environmental effects of bioenergy-driven land use changes, and then identified that limited attention had been given to regional-scale environmental issues in past debates when this research commenced in 2009. Since then, however, parallel studies have emerged and begun to address the necessity for a full range of indicators and comprehensive assessment frameworks for bioenergy sustainability that also take into account socio-economic impacts at the regional-scale. This was a response to increased requirements for sustainability in bioenergy production and the recent development of certification schemes. Although the research scope was essential, bioenergy crop production on 'underutilised agricultural land' should be evaluated continuously in future research. To enable more thorough evaluation and decision making on future bioenergy land use, the evaluation framework can be further developed to include a greater range of socio-economic and environmental sustainability indicators and methods that address the regional scale issues as long as the indicators can be linked to spatial data. Those elements are crucial for the success of commercial scale bioenergy crop production. The strong advantage of this evaluation framework is its flexible structure, enabling seamless expansion and tailoring for particular situations in future case study applications. The future development of the evaluation framework can be undertaken in line with sustainability indicators, such as those developed by international accredited certification organisations.

Another limitation related to the research scope was the spatial scale. Due to issues of complexity, science has been struggling to model and estimate the dynamics of bioenergy-driven land use change and environmental issues on a global scale, as seen in the cases of indirect land use change (iLUC) (Prins et al. 2010; Di Lucia, Ahlgren & Ericsson 2012; Wicke et al. 2012). For example, 'underutilised' open grazing land (modified pasture) was identified in this research as the most sustainable land use change option for future bioenergy crop production in the Australian context. However, this land use change pathway/scenario may involve the displacement of grazing activities (i.e. iLUC effects) and associated environmental impacts on a larger spatial scale as a result of trade-offs with food production, as reported in Brazil (Chapter 2). This evaluation framework was used at a regional scale and was not designed to capture those impacts. The land use change dynamics caused by conversion of 'underutilised' modified pastures provide opportunities for further research. In parallel, inclusion and evaluation of impacts from land use change dynamics at a larger than regional scale is the next challenge for future development of this evaluation framework.

Limitation 2: High level of uncertainty in renewable energy/bioenergy policy direction and lack of availability of data

The last limitation concerns changes in the political climate related to renewable energy in Australia, and the resulting large uncertainties in bioenergy crop production and processing at both national and state levels. This situation may have affected the overall results of this research significantly through the lack of clear direction in bioenergy policy, and the lack of availability of various scientific data on bioenergy crops. A biofuels target of 350 ML established by the former Howard Government (under the 'Biofuels for Cleaner Transport' announcement of October 2001) was surpassed, but unlike other developed countries, since then Australia appears to have abandoned bioenergy policy or even renewable energy policy at the national level. The current Abbot Government announced in February 2014 that the 20% Renewable Energy Target (RET) by 2020 is under review (Australian Government 2014). Considering the current scepticism towards climate change issues and renewable energy at the national level the outcomes of this review are similarly unfavourable to future development in the Australia's bioenergy industry. At the state level, New South Wales (NSW) is the only state that adopted and increased a biofuel mandate, however.

The lack of direction due to inconsistent and unclear bioenergy policy placed limitations on the ability of this research to develop 'realistic' land use change scenarios for the case study region. Due to high levels of uncertainty in production (e.g. national and regional demand, costs) and processing of these crops (e.g. technology, costs), the land use change scenarios in this research were developed based on biophysical suitability. Although they were effective in assisting in better answering the research questions, and highlighting the potential environmental effects and implications from land use change pathways and scenarios, the land use change scenarios used for this analysis may be updated as more reliable information becomes available (if this happens in future). This updating will enable estimates that are aligned with the actual bioenergy land use plans for the region, and this will be valuable for decision makers of the region.

The uncertainty in bioenergy policy at the national level is one of the reasons, which has also caused slow progress in bioenergy research in Australia generally compared to other parts of the developed world. This includes the current lack of scientific data on large-scale production of Pongamia and eucalypt species in subtropical Queensland (Chapter 6). Due to the lack of such regional-specific actual data on these crops, the estimates of biomass production and environmental evaluation depended largely on assumptions based on results reported in the literature in other parts of the world, and expert judgements. As a result, this may have limited the overall accuracy of the estimation of the environmental outcomes of the scenarios. However, research including field trials is currently in progress for these crops across Australia, including in the case study catchment in subtropical Queensland. In coming years more accurate data will be available from the trials. By collaborating with researchers working on bioenergy crops, future applications should be conducted reflecting the data and the results obtained from field sites. This could significantly improve the quality of the assumptions for the hydrological and water quality simulation and for the biodiversity conservation value, which would benefit the overall reliability of the evaluation.

7.5 Priorities for future research

Priorities for future research are identified mainly related to two areas: (i) further evaluation of sustainability outcomes of land use change scenarios using ‘underutilised agricultural land’; and (ii) refinement and development in the evaluation methodology or framework.

In this research, the scenarios that incorporated underutilised agricultural land were developed and evaluated in the context of subtropical Queensland, Australian, with relevance to the regional spatial scale. In light of a wide variety of definitions of underutilised agricultural lands and associated land use change scenarios across the world, the environmental sustainability of underutilised agricultural land scenarios needs to be evaluated further using alternative land use change scenarios relevant to other geographic regions. The findings from these future evaluations will ensure improved outcomes for future bioenergy crop production on these lands, and can provide suggestions for future policy direction.

For the above reason, further improvement, refinement and development of the evaluation framework through future applications is a high priority for future research. In this research, the evaluation framework was developed to ensure that it was suitable for widespread application, including use in regions in developing countries that urgently need evidence-based recommendations on which to make decisions concerning land use for future bioenergy crop production. Although the application in this research was limited to a case study region in the subtropical Queensland, the experience identified several benefits and challenges of the evaluation framework that need to be taken into account in future applications. These included the currently limited scope of the evaluation framework (not including other socio-economic and environmental sustainability indicators, iLUC), the limited input data availability for the key models used within the *Source* platform and the method for calculating biodiversity conservation values and actual habitat amount. In this context, a number of future applications of the evaluation framework to varied land use change scenarios will help to increase the flexibility depending on varied regional-scale sustainability issues and situations, and also improve the overall reliability, accuracy and stability of this methodology as overcoming the current challenges.

References

ABARES 2011a, *Catchment scale examples of land use change reporting*, Australian Collaborative Land Use and Management Program (ACLUMP), viewed 20 September 2012, <http://adl.brs.gov.au/landuse/index.cfm?fa=main.catchmentExamples>.

—— 2011b, *Energy in Australia 2011*, Canberra.

—— 2011c, *Guidelines for land use mapping in Australia: principles, procedures and definitions, fourth edition*, Australian Bureau of Agricultural and Resource Economics and Sciences, Canberra.

ABARES & BRS 2006, *Land Use of Australia, Version 3 2001/2002*, Australian Bureau of Agricultural and Resource Economics, Bureau of Rural Sciences.

Abdullah, SA & Hezri, AA 2008, 'From forest landscape to agricultural landscape in the developing tropical country of Malaysia: Pattern, process, and their significance on policy', *Environmental Management*, vol. 42, no. 5, pp. 907-17.

Abdullah, SA & Nakagoshi, N 2007, 'Forest fragmentation and its correlation to human land use change in the state of Selangor, peninsular Malaysia', *Forest Ecology and Management*, vol. 241, no. 1-3, pp. 39-48.

ABS 2010a, *National Regional Profile: North Burnett (R) - Monto (statistical local area)*, Australian Bureau of Statistics, viewed 31 July 2012, <http://www.abs.gov.au/AUSSTATS/abs@nrp.nsf/Previousproducts/315105764Population/People12005-2009?opendocument&tabname=Summary&prodno=315105764&issue=2005-2009>.

—— 2010b, *National Regional Profile: North Burnett (R) - Mundubbera (Statistical Local Area)* Australian Bureau of Statistics, viewed 26 September 2012, <http://www.abs.gov.au/AUSSTATS/abs@nrp.nsf/Previousproducts/315105765Population/People12005-2009?opendocument&tabname=Summary&prodno=315105765&issue=2005-2009>.

—— 2010c, *National Regional Profile: South Burnett (R) - Kingaroy (statistical local area)*, 31 July, Australian Bureau of Statistics, 31 October, 2011, <http://www.abs.gov.au/AUSSTATS/abs@nrp.nsf/Previousproducts/315106632Population/People12005-2009?opendocument&tabname=Summary&prodno=315106632&issue=2005-2009>.

Ackom, E, Mabee, W & Saddler, J 2010, *Backgrounder: Major environmental criteria of biofuel sustainability: IEA Bioenergy Task 39 Report T39-PR4*.

Ajanovic, A 2011, 'Biofuels versus food production: Does biofuels production increase food prices?', *Energy*, vol. 36, no. 4.

Al-Riffai, P, Dimaranan, B & Laborde, D 2010, *Global trade and environmental impact study of the EU biofuels mandate: Final report*, International Food Policy Institute (IFPRI), European Commission.

Alkemade, R, van Oorschot, M, Miles, L, Nellemann, C, Bakkenes, M & ten Brink, B 2009, 'GLOBIO3: A framework to investigate options for reducing global terrestrial biodiversity loss', *Ecosystems*, vol. 12, no. 3, pp. 374-90.

Allan, JD 2004, 'Landscapes and riverscapes: The influence of land use on stream ecosystems', *Annual Review of Ecology Evolution and Systematics*, vol. 35, pp. 257-84.

Ardiansyah, F & Barano, T 2012, 'Pushing forward to better land use', *Jakarta Post 27 May 2012*, 27 May, 2012.

Arnold, JG, Srinivasan, R, Muttiah, S & Williams, JR 1998, ‘Large-area hydrologic modeling and assessment: Part I. Model development’. *Journal of the American Water Resources Association*, vol. 34, no. 1, pp.73-89.

Andrén, H 1994, 'Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat - A review', *Oikos*, vol. 71, no. 3, pp. 355-66.

Australian Government 2003, *Australia's 15 National Biodiversity Hotspots*, Department of Sustainability, Environment, Water, Population and Communities, viewed 21 August 2013, <http://www.environment.gov.au/biodiversity/hotspots/national-hotspots.html>.

—— 2011, Biodiversity summary for NRM regions, species list for Burnett Mary Queensland. Department of Sustainability, Environment, Water, Population and Communities,

<http://www.environment.gov.au/heritage/anhat/summaries/qld/pubs/species-qld-burnett-mary.pdf>.

—— 2013, *Brigalow (Acacia harpophylla dominant and co-dominant)* Department of Sustainability, Environment, Water, Population and Communities,

<http://www.environment.gov.au/cgi-bin/sprat/public/publicshowcommunity.pl?id=28>.

Banse, M, van Meijl, H, Tabeau, A & Woltjer, G 2008, 'Will EU biofuel policies affect global agricultural markets?', *European Review of Agricultural Economics*, vol. 35, no. 2, pp. 117-41.

Banse, M, van Meijl, H, Tabeau, A, Woltjer, G, Hellmann, F & Verburg, PH 2011, 'Impact of EU biofuel policies on world agricultural production and land use', *Biomass and Bioenergy*, vol. 35, no. 6, pp. 2385-90.

Banse, M, van Meijl, H & Woltjer, G 2008, 'The impact of first and second generation biofuels on global agricultural production, trade and and use', paper presented to *The11th Annual GTAP Conference*, Helsinki, Finland, June 12-14, 2008.

Barona, E, Ramankutty, N, Hyman, G & Coomes, OT 2010, 'The role of pasture and soybean in deforestation of the Brazilian Amazon', *Environmental Research Letters*, vol. 5, no. 2, p. 9.

Bauen, A, Chudziak, C, Vad, K & Watson, P 2010, *A causal descriptive approach to modelling the GHG emissions associated with the indirect land use impacts of biofuels. Final report. A study for the UK Department for Transport*, E4tech, London.

Baum, S, Bolte, A & Weih, M 2012, 'Short rotation coppice (SRC) plantations provide additional habitats for vascular plant species in agricultural mosaic landscapes', *BioEnergy Research*, vol. 5, no. 3, pp. 573-83.

Bellamy, PE, Croxton, PJ, Heard, MS, Hinsley, SA, Hulmes, L, Hulmes, S, Nuttall, P, Pywell, RF & Rothery, P 2009, 'The impact of growing miscanthus for biomass on farmland bird populations', *Biomass and Bioenergy*, vol. 33, no. 2, pp. 191-9.

Berndes, G, Bird, N & Cowie, A 2010, *Bioenergy, land use change and climate change mitigation*, IEA Bioenergy.

Bhattacharya, SC, Salam, PA, Pham, HL & Ravindranath, NH 2003, 'Sustainable biomass production for energy in selected Asian countries', *Biomass and Bioenergy*, vol. 25, no. 5, pp. 471-82.

Biodiesel Magazine 2010, 'Global biodiesel production and market report', *Biodiesel Magazine*, September 2010 issue.

Biopact 2006, *Palm biofuel survives low crude oil prices - official*, viewed 29 December 2010,

<http://news.mongabay.com/bioenergy/2006/09/palm-biofuel-survives-low-crude-oil.html>.

Blanco Fonseca, M, Burrell, A, Gay, H, Henseler, M, Kavallari, A, M'Barek, R, Domínguez, IP & Tonini, A 2010, *Impacts of the EU biofuel target on agricultural markets and land use: a comparative modelling assessment: Final report (June 2010)*, European Commission, Joint Research Centre, Institute for Prospective Technological Studies, Luxembourg.

BMRG 2008, *Healthy Habitats newsletter November/December 2008*, The Burnett Mary Regional Group, Bundaberg.

Bonyhady, T 2002, *The colonial earth*, Melbourne University Press, Carlton, Vic.

Borah, DK & Berra, M 2004, 'Watershed-scale hydrologic and nonpoint-source pollution models: Review of applications', *Transactions of the ASABE*, vol. 47, no. 3, pp.789-803.

Borras, SM, McMichael, P & Scoones, I 2010, 'The politics of biofuels, land and agrarian change: editors' introduction', *Journal of Peasant Studies*, vol. 37, no. 4, pp. 575-92.

Bosch, JM & Hewlett, JD 1982, 'A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration', *Journal of Hydrology*, vol. 55, no. 1-4, pp. 3-23.

Bouget, C, Lassauce, A & Jonsell, M 2012, 'Effects of fuelwood harvesting on biodiversity - a review focused on the situation in Europe', *Canadian Journal of Forest Research*, vol. 42, no. 8, pp. 1421-32.

Bowen, ME, McAlpine, CA, House, APN & Smith, GC 2007, 'Regrowth forests on abandoned agricultural land: A review of their habitat values for recovering forest fauna', *Biological Conservation*, vol. 140, no. 3-4, pp. 273-96.

Brannstrom, C, Jepson, W, Filippi, AM, Redo, D, Xu, Z & Ganesh, S 2008, 'Land change in the Brazilian Savanna (Cerrado), 1986-2002: Comparative analysis and implications for land-use policy', *Land Use Policy*, vol. 25, no. 4, pp. 579-95.

Brodie, J & Mitchell, A 2005, 'Nutrients in Australian tropical rivers: changes with agricultural development and implications for receiving environments', *Marine and Freshwater Research*, vol. 56, no. 3, pp. 279-302.

Bryan, BA, King, D & Wang, E 2010, 'Potential of woody biomass production for motivating widespread natural resource management under climate change', *Land Use Policy*, vol. 27, no. 3, pp. 713-25.

Bryan, BA, Ward, J & Hobbs, T 2008, 'An assessment of the economic and environmental potential of biomass production in an agricultural region', *Land Use Policy*, vol. 25, no. 4, pp. 533-49.

Bryngelsson, DK & Lindgren, K 2013, 'Why large-scale bioenergy production on marginal land is unfeasible: A conceptual partial equilibrium analysis', *Energy Policy*, vol. 55, pp. 454-66.

Bureau of Meteorology 2008, Mean monthly and mean annual maximum, minimum & mean temperature data (base climatological data sets),

<http://www.bom.gov.au/climate/averages/climatology/gridded-data info/metadata/md_ave_temp_1961-90.shtml>.

—— 2009, Mean monthly and mean annual rainfall data (base climatological data sets), <http://www.bom.gov.au/climate/averages/climatology/gridded-data-info/metadata/md_ave_rain_1961-90.shtml>.

Bureau of Rural Science 2010, *User guide and caveats for the Land Use of Australia, Version 4, 2005-06*, by Bureau of Rural Science.

Burnett Catchment Care Association Inc. 2012a, *Healthy Habitats case study: Feral pig baiting was undertaken south-west of Mundubbera*, Monto.

—— 2012b, *Healthy Habitats case study: Ted May's properties on the Mulgildie plateau, near Monto*, Mundubbera.

—— 2012c, *Healthy Habitats case study: The property located north of Monto in Queensland's North Burnett region*, Monto.

Burnett Mary Regional Group 2013, *2013 Flood Event*, viewed 12 February 2013, <http://www.bmrg.org.au/>.

—— n.d., *Burnett-Baffle Water Quality Improvement Plan*.

Calfornia Air Resource Board 2012, *Low Carbon Fuel Standard Program*, viewed 10 January 2012, <http://www.arb.ca.gov/fuels/lcfs/lcfs.htm>.

Campbell, JE, Lobell, DB, Genova, RC & Field, CB 2008, 'The global potential of bioenergy on abandoned agriculture lands', *Environmental Science & Technology*, vol. 42, no. 15, pp. 5791-4.

Carpenter, SR, Caraco, NF, Correll, DL, Howarth, RW, Sharpley, AN & Smith, VH 1998, 'Nonpoint pollution of surface waters with phosphorus and nitrogen', *Ecological Applications*, vol. 8, no. 3, pp. 559-68.

Centre for Ecology & Hydrology 2013, *Accessing land cover mapping data*, viewed 18 March 2013, <http://www.ceh.ac.uk/AccessingLCMData.html>.

Chambers, D & Holtum, JAM 2010, *Feasibility of Agave as a feedstock for biofuel production in Australia*, Rural Industries Research and Development Corporation.

Chiew, F & Scanlon, P 2002, *Estimation of pollutant concentrations for EMSS modelling of the South-East Queensland region, technical report*, Cooperative Research Centre for Catchment Hydrology, Canberra.

Chiew, F & Siriwardena, L 2005, 'Estimation of SIMHYD parameter values for application in ungauged catchments', paper presented to *MODSIM 2005 International Congress on Modelling and Simulation*, Modelling and Simulation Society of Australia and New Zealand, Melbourne, December 2005.

CIAT & CCAFS 2012, *GCM downscaled GCM data portal*, the International Center for Tropical Agriculture (CIAT), the CGIAR Research Program on Climate Change, Agriculture and Food Security (CCAFS), viewed 27 March 2013, <http://www.ccafs-climate.org/>.

Clean Energy Council 2008, *Australian bioenergy roadmap - setting the direction for biomass in stationary energy to 2020 and beyond*.

Collard, S, Le Brocque, A & Zammit, C 2009, 'Bird assemblages in fragmented agricultural landscapes: The role of small brigalow remnants and adjoining land uses', *Biodiversity and Conservation*, vol. 18, no. 6, pp. 1649-70.

Commonwealth of Australia 2013, *Australian national greenhouse accounts, Quarterly update of Australia's national greenhouse gas inventory, December quarter 2012*, The Department of Industry, Innovation, Climate Change, Science, Research and Tertiary Education.

Companhia de Desenvolvimento dos Vales do São Francisco 2002, *Integrated management of land based activities in the São Francisco river basin project: determination of land-use in the lower São Francisco river basin, executive summary of the final report*, Agência Nacional de Águas (ANA), Global Environment Facility (GEF), United Nations Environment Program (UNEP), Organization of American States (OAS) Brasilia.

Costa, MH, Yanagi, SNM, Souza, P, Ribeiro, A & Rocha, EJP 2007, 'Climate change in Amazonia caused by soybean cropland expansion, as compared to caused by pastureland expansion', *Geophysical Research Letters*, vol. 34, no. 7.

Crutzen, PJ, Mosier, AR, Smith, KA & Winiwarter, W 2007, 'N_2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels', *Atmospheric Chemistry and Physics Discussion*, vol. 7, no. 4, pp. 11191-205.

Dale, VH, Kline, KL, Wiens, J & Fargione, J 2010, *Biofuels: implications for land use and biodiversity*, Ecological Society of America Washington, DC.

Danielsen, F, Beukema, H, Burgess, ND, Parish, F, Bruhl, CA, Donald, PF, Murdiyarso, D, Phalan, B, Reijnders, L, Struebig, M & Fitzherbert, EB 2009, 'Biofuel plantations on forested lands: Double jeopardy for biodiversity and climate', *Conservation Biology*, vol. 23, no. 2, pp. 348-58.

de Nie, D, Sayer, J & McCormick, N 2009, 'Indirect effects of bioenergy: effects on landscapes and livelihoods', in *Bioenergy - the impact of indirect land use change: Summary and conclusions from the IEA Bioenergy ExCo63 Workshop*, Rotterdam, the Netherlands.

de Vries, SC, van de Ven, GWJ, van Ittersum, MK & Giller, KE 2010, 'Resource use efficiency and environmental performance of nine major biofuel crops, processed by first-generation conversion techniques', *Biomass and Bioenergy*, vol. 34, no. 5, pp. 588-601.

DEEDI 2010, *Weed risk assessment: Pongamia (Millettia pinnata syn. Pongamia pinnata)*, Queensland Department of Employment, Economic Development and Innovation

DEHP 2014, Remnant vegetation in Queensland, Queensland Department of Environment and Heritage Protection, viewed 30 May 2014,

<http://www.ehp.qld.gov.au/ecosystems/remnant-vegetation/>.

DERM 2009, *Regional Ecosystem Mapping, Version 6.0b*, Queensland Department of Environment and Resource Management, Brisbane.

—— 2010, *Land cover change in the Burnett-Mary Regional Group Natural Resource Management region 2008-09*, Queensland Department of Environment and Resource Management.

—— 2011, *Foliage Projective Cover (FPC) data 2001*, Queensland Department of Environment and Resource Management, Brisbane.

Di Lucia, L, Ahlgren, S & Ericsson, K 2012, 'The dilemma of indirect land-use changes in EU biofuel policy—An empirical study of policy-making in the context of scientific uncertainty', *Environmental Science & Policy*, vol. 16, pp. 9-19.

Diesendorf, M 2007, *Greenhouse solutions with sustainable energy,* University of New South Wales Press, Sydney.

Dimitriou, L, Busch, G, Jacobs, S, Schmidt-Walter, P & Lamersdorf, N 2009, 'A review of the impacts of Short Rotation Coppice cultivation on water issues', *Landbauforschung Volkenrode*, vol. 59, no. 3, pp. 197-206.

DNRM 2003, *Land cover change in Queensland 1991-2001, A Statewide Landcover and Trees Study Report (SLATS), Jan 2003.*, Queensland Department of Natural Resources and Mines, Indooroopilly.

Donnollan, TE 1998, *Soils and irrigated land suitability of the Bundaberg area, south east Queensland* Queensland Department of Natural Resources, Brisbane.

Donnollan, TE & Searle, RD 1999, *Land resources of the Burnett region, Queensland. Part 3: North Burnett*, Queensland Department of Natural Resources, Brisbane.

Doornbosch, R & Steenblik, R 2007, *OECD: Biofuels: Is the cure worse than the disease?*, OECD.

Dornburg, V, Faaij, A, Verweij, P, Langeveld, H, van de Ven, G, Wester, F, van Keulen, H, van Diepen, K, Meeusen, M, Banse, M, Ros, J, van Vuuren, D, van den Born, GJ, van Oorschot, M, Smout, F, van Vliet, J, Aiking, H, Londo, M, Mozaffarian, H & Smekens, K 2008, *Biomass assessment: Assessment of global biomass potentials and their links to food, water, biodiversity, energy demand and economy*, Universiteit Utrecht, Wageningen Universiteit, Vrije Universiteit Amsterdam, Netherlands Environmental Assessment Agency, ECN, UCE, Bilthoven.

Dornburg, V, Van Vuuren, D, Van De Ven, G, Langeveld, H, Meeusen, M, Banse, M, Van Oorschot, M, Ros, J, Jan Van Den Born, G, Aiking, H, Londo, M, Mozaffarian, H, Verweij, P, Lysen, E & Faaij, A 2010, 'Bioenergy revisited: Key factors in global potentials of bioenergy', *Energy and Environmental Science*, vol. 3, no. 3, pp. 258-67.

DSITIA 2012a, *Foliage Projective Cover (FPC) data1999*, Queensland Department of Science, Information Technology, Innovation and the Arts, Brisbane.

—— 2012b, *Foliage Projective Cover (FPC) data 2005*, Queensland Department of Science Information Technology Innovation and the Arts, Brisbane.

—— 2012c, *Land cover change in Queensland 2009-10: Statewide Landcover and Tree Study (SLATS) report*, Queensland Department of Science, Information Technology, Innovation and the Arts, Brisbane.

—— 2012d, *Land use Summary 1999–2009: Burnett–Mary NRM Region.*, Queensland Department of Science, Information Technology, Innovation and the Arts, Brisbane.

—— 2012e, *Sediment, nutrient and pesticide loads: Great Barrier Reef catchment loads monitoring 2009-2010*, Queensland Department of Science, Information Technology, Innovation and the Arts, Brisbane.

—— 2012f, *SILO climate data*, Queensland Department of Science, Information Technology, Innovation and the Arts, 8 May, 2012, <http://www.longpaddock.qld.gov.au/silo/>.

—— 2012g, *Updated 2009 land use mapping in the priority GBR catchments*, Queensland Land Use Mapping Program (QLUMP), Queensland Department of Science, Information Technology, Innovation and the Arts, Brisbane.

EBB 2010, *Statistics - The EU biodiesel industry*, European Biodiesel Board (EBB), viewed 16 November 2010, <http://www.ebb-eu.org/stats.php#>.

EEA 2007, *Estimating the environmentally compatible bioenergy potential from agriculture*, European Environmental Agency, Copenhagen.

—— 2013, *Greenhouse gas emission trends (CSI 010) - Assessment published May 2013*, viewed June 6 2013, <http://www.eea.europa.eu/data-and-maps/indicators/greenhouse-gas-emission-trends/greenhouse-gas-emission-trends-assessment-5>.

Efroymson, RA, Dale, VH, Kline, KL, McBride, AC, Bielicki, JM, Smith, RL, Parish, ES, Schweizer, PE & Shaw, DM 2013, 'Environmental indicators of biofuel sustainability: What about context?', *Environmental Management*, vol. 51, no. 2, pp. 291-306.

ePure 2010, *Production data*, European renewable ethanol, viewed January 5 2011, <http://www.epure.org/statistics.php?id=4>.

ERA-NET Bioenergy 2010, *RATING-SRC*, viewed 5 April 2013, <http://www.ratingsrc.eu/node/1>.

European Commission 2003, *Directive 2003/30/EC on the promotion of the use of biofuels or other renewable fuels for transport*, by European Commission.

European Commission 2009, *Directive 2009/28/EC on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directive 2001/77/EC and 2003/30/EC*

European Commission Joint Research Centre 2003, *Global Land Cover 2000 database*, European Commission, viewed 2013 15 March,

<http://bioval.jrc.ec.europa.eu/products/glc2000/glc2000.php>.

European Environment Agency 2013, *Data and maps*, viewed 15 March 2013, <http://www.eea.europa.eu/data-and-maps>.

European Union 2010, *Communication from the Commission on the practical implementation of the EU biofuels and bioliquids sustainability scheme and on counting rules for biofuels (2010/C 160/02)*, 19 June 2010, <http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:C:2010:160:0008:0016:EN:PDF>.

Eva, HD, De Miranda, EE, Di Bella, CM, Gond, V, Huber, O, Sgrenzaroli, M, Jones, S, Coutinho, A, Dorado, A, Guimarães, M, Elvidge, C, Achard, F, Belward, AS, Bartholomé, E, Baraldi, A, De Grandi, G, Vogt, P, Fritz, S & Hartley, A 2002, *A vegetation map of South America*, European Commission Joint Research Centre, Luxembourg.

Berndes, G, Bird, N & Cowie, A 2010, *Bioenergy, land use change and climate change mitigation*, IEA Bioenergy, <http://www.ieabioenergy.com/LibItem.aspx?id=6770>.

eWater CRC 2010, '*Source Catchments' scientific reference guide*, eWater Cooperative Research Centre, Canberra.

—— 2011, *Introduction to catchment modelling using 'Source Catchments': Training manual* eWater Cooperative Research Centre, Canberra.

—— 2012a, *Case studies*, viewed 10 June 2013, <http://www.ewater.com.au/casestudies/>.

—— 2012b, *River Murray case study*, viewed 1 July 2013, <http://www.ewater.com.au/casestudies/rivers-case-studies/the-river-murray/>.

—— 2012c, Source *modelling community*, viewed 19 March 2013, <http://www.ewater.com.au/products/ewater-source/source-modelling-community/>.

—— 2013, *Australia and India partner to manage water for scarcity*, viewed 6 September 2013, <http://www.ewater.com.au/news/media/?news=249>.

Ewers, RM & Didham, RK 2006, 'Confounding factors in the detection of species responses to habitat fragmentation', *Biological Reviews*, vol. 81, no. 1, pp. 117-42.

Eyre, TJ 2006, 'Regional habitat selection of large gliding possums at forest stand and landscape scales in southern Queensland, Australia - I. Greater glider (Petauroides volans)', *Forest Ecology and Management*, vol. 235, no. 1-3, pp. 270-82.

Eyre, TJ, Maron, M, Mathieson, MT & Haseler, M 2009, 'Impacts of grazing, selective logging and hyper-aggressors on diurnal bird fauna in intact forest landscapes of the Brigalow Belt, Queensland', *Austral Ecology*, vol. 34, no. 6, pp. 705-16.

Fabricius, KE 2005, 'Effects of terrestrial runoff on the ecology of corals and coral reefs: review and synthesis', *Marine Pollution Bulletin*, vol. 50, no. 2, pp. 125-46.

Fahrig, L 1997, 'Relative effects of habitat loss and fragmentation on population extinction', *Journal of Wildlife Management*, vol. 61, no. 3, pp. 603-10.

—— 2001, 'How much habitat is enough?', *Biological Conservation*, vol. 100, no. 1, pp. 65-74.

—— 2003, 'Effects of habitat fragmentation on biodiversity', *Annual Review of Ecology Evolution and Systematics*, vol. 34, pp. 487-515.

FAO 2006, *Global forest resources assessment 2005: progress towards sustainable forest management*, FAO forestry paper, Food and Agriculture Organization of the United Nations, Rome.

—— 2008, *The state of food and agriculture 2008. Biofuels: prospects, risks and opportunities*, Food and Agriculture Organization, Rome.

—— 2010, *Global forest resources assessment 2010, main report*, Food and Agriculture Organization of the United Nations, Rome.

FAOSTAT 2011, *FAOSTAT database*, Food and Agriculture Organization of the United Nations (FAO), <http://faostat.fao.org/>.

Fargione, J, Hill, J, Tilman, D, Polasky, S & Hawthorne, P 2008, 'Land clearing and the biofuel carbon debt', *Science*, vol. 319, no. 5867, pp. 1235-8.

Farine, DR, O'Connell, DA, Raison, RJ, May, BM, O'Connor, MH, Crawford, DF, Herr, A, Taylor, JA, Jovanovic, T, Campbell, PK, Dunlop, MIA, Rodriguez, LC, Poole, ML, Braid, AL & Kriticos, D 2012, 'An assessment of biomass for bioelectricity and biofuel, and for greenhouse gas emission reduction in Australia', *Global Change Biology Bioenergy*, vol. 4, no. 2, pp. 148-75.

Fearnside, PM 2001a, 'Land-tenure issues as factors in environmental destruction in Brazilian Amazonia: The case of southern Pará', *World Development*, vol. 29, no. 8, pp. 1361-72.

—— 2001b, 'Soybean cultivation as a threat to the environment in Brazil', *Environmental Conservation*, vol. 28, no. 1, pp. 23-38.

—— 2005, 'Deforestation in Brazilian Amazonia: History, rates, and consequences', *Conservation Biology*, vol. 19, no. 3, pp. 680-8.

Fearnside, PM, Righi, CA, Graca, P, Keizer, EWH, Cerri, CC, Nogueira, EM & Barbosa, RI 2009, 'Biomass and greenhouse-gas emissions from land-use change in Brazil's Amazonian "arc of deforestation": The states of Mato Grosso and Rondonia', *Forest Ecology and Management*, vol. 258, no. 9, pp. 1968-78.

Fentie, B in press, *Modelling reductions of pollutant loads due to improved management practices in the Great Barrier Reef catchments – Mary-Burnett NRM region, volume 7, technical report March 2013* Queensland Department of Natural Resources and Mines.

Field, CB, Campbell, JE & Lobell, DB 2008, 'Biomass energy: the scale of the potential resource', *Trends in Ecology & Evolution*, vol. 23, no. 2, pp. 65-72.

Fiorese, G & Guariso, G 2010, 'A GIS-based approach to evaluate biomass potential from energy crops at regional scale', *Environmental Modelling & Software*, vol. 25, no. 6, pp. 702-11.

Fischer, G, Prieler, S, van Velthuizen, H, Berndes, G, Faaij, A, Londo, M & de Wit, M 2010, 'Biofuel production potentials in Europe: Sustainable use of cultivated land and pastures, part II: Land use scenarios', *Biomass & Bioenergy*, vol. 34, no. 2, pp. 173-87.

Fritsche, UR, Hennenberg, K & Hünecke, K 2010, *The "iLUC Factor" as a means to hedge risks of GHG emissions from indirect land use change—working paper*, Öko-Institut, Darmstadt.

Fritsche, UR, Hennenberg, KJ, Hermann, A, Hünecke, K, Herrera, R, Fehrenbach, H, Roth, E, Hennecke, A & Giegrich, J 2010, *Development of strategies and sustainability standards for the certification of biomass for international trade: Summarizing final report*, German Federal Environment Agency (Umweltbundesamt), Dessau-Roßlau.

Fritsche, UR, Sims, REH & Monti, A 2010, 'Direct and indirect land-use competition issues for energy crops and their sustainable production - an overview', *Biofuels, Bioproducts and Biorefining*, vol. 4, no. 6, pp. 692-704.

Gawel, E & Ludwig, G 2011, 'The iLUC dilemma: How to deal with indirect land use changes when governing energy crops?', *Land Use Policy*, vol. 28, no. 4, pp. 846-56.

Geoscience Australia & ABARE 2010, *Australian energy resource assessment*, Commonwealth of Australia, Canberra, Australia.

Ghazoul, J, Butler, RA, Mateo-Vega, J & Koh, LP 2010, 'REDD: a reckoning of environment and development implications', *Trends in Ecology & Evolution*, vol. 25, no. 7, pp. 396-402.

Gibbs, HK, Johnston, M, Foley, JA, Holloway, T, Monfreda, C, Ramankutty, N & Zaks, D 2008, 'Carbon payback times for crop-based biofuel expansion in the tropics: the effects of changing yield and technology', *Environmental Research Letters*, vol. 3, no. 3, pp. 1-10.

Gibbs, HK, Ruesch, AS, Achard, F, Clayton, MK, Holmgren, P, Ramankutty, N & Foley, JA 2010, 'Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s', *Proceedings of the National Academy of Sciences of the United States of America*, vol. 107, no. 38, pp. 16732-7.

Gilbert, M & Brodie, J 2001, *Population and major land use in the Great Barrier Reef catchment area: Spatial and temporal trends*, Great Barrier Reef Marine Park Authority

Githui, F, Selle, B & Thayalakumaran, T 2012, 'Recharge estimation using remotely sensed evapotranspiration in an irrigated catchment in southeast Australia', *Hydrological Processes*, vol. 26, no. 9, pp. 1379-89.

Goldemberg, J, Coelho, ST & Guardabassi, P 2008, 'The sustainability of ethanol production from sugarcane', *Energy Policy*, vol. 36, no. 6, pp. 2086-97.

Gopalakrishnan, G, Negri, MC, Wang, M, Wu, M, Snyder, SW & Lafreniere, L 2009, 'Biofuels, land, and water: A systems approach to sustainability', *Environmental Science & Technology*, vol. 43, no. 15, pp. 6094-100.

Gourley, CJP & Weaver, DM 2012, 'Nutrient surpluses in Australian grazing systems: Management practices, policy approaches, and difficult choices to improve water quality', *Crop and Pasture Science*, vol. 63, no. 9, pp. 805-18.

Government of Western Australia 2013, Amendments to native vegetation clearing regulations, viewed 24 April 2014, <http://www.der.wa.gov.au/8-about-us/news/87-amendments-to-native-vegetation-clearing-regulations>.

Graham, RL, Downing, M & Walsh, ME 1996, 'A framework to assess regional environmental impacts of dedicated energy crop production', *Environmental Management*, vol. 20, no. 4, pp. 475-85.

Grau, HR, Gasparri, NI & Aide, TM 2008, 'Balancing food production and nature conservation in the neotropical dry forests of northern Argentina', *Global Change Biology*, vol. 14, no. 5, pp. 985-97.

Great Barrier Reef Marine Park Authority 2009, *Burnett River catchment*, viewed 26 September 2012, <http://www.gbrmpa.gov.au/outlook-for-the-reef>.

Green, RJ & Catterall, CP 1998, 'The effects of forest clearing and regeneration on the fauna of Wivenhoe Park, south-east Queensland', *Wildlife Research*, vol. 25, no. 6, pp. 677-90.

Hammerschlag, R 2006, 'Ethanol's energy return on investment: A survey of the literature 1990–present', *Environmental Science & Technology*, vol. 40, no. 6, pp. 1744-50.

Hammersley, M & Gomm, R 2000, 'Introduction', in R Gomm, M Hammersley & P Foster (eds), *Case study method*, SAGE Publications, London.

Hargis, C, Bissonette, J & David, J 1998, 'The behavior of landscape metrics commonly used in the study of habitat fragmentation', *Landscape Ecology*, vol. 13, no. 3, pp. 167-86.

Harper, RJ, Sochacki, SJ, Smettem, KRJ, Robinson, N, Silberstein, RP, Clarke, CJ, McGrath, JF, Crombie, DS & Hampton, CE 2009, *Catchment scale evaluation of trees, water and salt*, Rural Industries Research and Development Corporation (RIRDC).

Hartman, JC, Nippert, JB, Orozco, RA & Springer, CJ 2011, 'Potential ecological impacts of switchgrass (*Panicum virgatum L.*) biofuel cultivation in the central Great Plains, USA', *Biomass and Bioenergy*, vol. 35, no. 8, pp. 3415-21.

Haynes, D, Brodie, J, Waterhouse, J, Bainbridge, Z, Bass, D & Hart, B 2007, 'Assessment of the water quality and ecosystem health of the Great Barrier Reef (Australia): conceptual models', *Environmental Management*, vol. 40, pp. 993-1003.

Hiederer, R, Ramos, F, Capitani, C, Koeble, R, Blujdea, V, Gomez, O, Mulligan, D & Marelli, L 2010, *Biofuels: a new methodology to estimate GHG emissions from global land use change, A methodology involving spatial allocation of agricultural land demand and estimation of CO2 and N2O emissions*, European Commission, Joint Research Centre.

Hijmans, RJ, Cameron, S & Parra, J 2005, *WorldClim—global climate data, free climate data for ecological modeling and GIS*, <http://www.worldclim.org/>.

Hellmann, F & Verburg, PH 2010, 'Impact assessment of the European biofuel directive on land use and biodiversity', *Journal of Environmental Management*, vol. 91, no. 6, pp. 1389-96.

—— 2011, 'Spatially explicit modelling of biofuel crops in Europe', *Biomass and Bioenergy*, vol. 35, no. 6, pp. 2411-24.

Hill, J 2007, 'Environmental costs and benefits of transportation biofuel production from food- and lignocellulose-based energy crops. A review', *Agronomy for Sustainable Development*, vol. 27, no. 1, pp. 1-12.

Hill, J, Nelson, E, Tilman, D, Polasky, S & Tiffany, D 2006, 'Environmental, economic, and energetic costs and benefits of biodiesel and ethanol biofuels', *Proceedings of the National Academy of Sciences of the United States of America*, vol. 103, no. 30, pp. 11206-10.

Hirota, M, Oyama, MD & Nobre, C 2011, 'Concurrent climate impacts of tropical South America land-cover change', *Atmospheric Science Letters*, vol. 12, no. 3, pp. 261-7.

HM Government 2009, *The UK Reneable Energy Strategy, Incentive*, London.

Hoh, R 2010, *Malaysia biofuels annual report 2010, USDA Foreign Agriculture Service GAIN Report* U.S. Department of Agriculture.

Hoogwijk, M, Faaij, A, Eickhout, B, de Vries, B & Turkenburg, W 2005, 'Potential of biomass energy out to 2100, for four IPCCSRES land-use scenarios', *Biomass & Bioenergy*, vol. 29, no. 4, pp. 225-57.

IBGE 2003, *Mapa mural de cobertura e uso da terra 1996 (Land use map 1996)*, 1: 5,000,000, Instituto Brasileiro de Geografia e Estatística, Rio de Janeiro.

—— 2010a, *Mapa de Vegetação do Brasil 2006 (Vegetation map of Brazil 2006)*, 1:5,000,000, Instituto Brasileiro de Geografia e Estatística, Rio de Janeiro.

—— 2010b, *Mapa mural de cobertura e uso da terra 2010 (Land use map 2010)*, 1: 5,000,000, Instituto Brasileiro de Geografia e Estatística, Rio de Janeiro.

IEA 2004, *Biofuels for transport: An international percepective*, International Energy Agency, Paris.

—— 2011, *Technology roadmap: Biofuels for transport*, International Energy Agency, Paris.

IEA Bioenergy 2009, 'Bioenergy—The impact of indirect land use change: summary and conclusions from the IEA Bioenergy ExCo63 Workshop', in Rotterdam, the Netherlands.

Im, S, Brannan, KM, Mostaghimi, S & Kim, SM 2007, 'Comparison of HSPF and SWAT models performance for runoff and sediment yield prediction', *Journal of Environmental Science and Health, Part A: Toxic/Hazardous Substances and Environmental Engineering*, vol. 42, pp. 1561-70.

INPE 2012, *Taxas anuais do desmatamento - 1988 até 2011 (Annual deforestation rate - 1988 to 2011)*, Instituto Nacional de Pesquisa Especiais (INPE), viewed July 10 2012, <http://www.obt.inpe.br/prodes/prodes_1988_2011.htm>.

Iowa Department of Natural Resources 2013, *Natural resources geographic information systems library*, viewed 15 March 2013, <http://www.igsb.uiowa.edu/nrgislibx/>.

IPCC 2006, *2006 IPCC Guidelines for national greenhouse gas inventories volume 4: Agriculture, forestry and other land use,* Intergovernmental Panel on Climate Change (IPCC), Hayama, Kanagawa.

Kanowski, J, Catterall, CP & Wardell-Johnson, GW 2005, 'Consequences of broadscale timber plantations for biodiversity in cleared rainforest landscapes of tropical and subtropical Australia', *Forest Ecology and Management*, vol. 208, no. 1-3, pp. 359-72.

Kanowski, JJ, Reis, TM, Catterall, CP & Piper, SD 2006, 'Factors affecting the use of reforested sites by reptiles in cleared rainforest landscapes in tropical and subtropical Australia', *Restoration Ecology*, vol. 14, no. 1, pp. 67-76.

Kazakoff, SH, Gresshoff, PM & Scott, PT 2011, 'Pongamia pinnata, a sustainable feedstock for biodiesel production (Chapter 12)', in N G.Halford & A Karp (eds), *Energy Crops*, Royal Society for Chemistry, Cambridge, UK, pp. 233-58.

Kent, DJ 2002, *Land resources of the Burnett region. Part 2: Central Burnett*, Queensland Department of Natural Resources and Mines, Brisbane.

Keyzer, M, Merbis, M & Voortman, R 2008, 'The biofuel controversy', *Economist-Netherlands*, vol. 156, no. 4, pp. 507-27.

Kim, H, Kim, S & Dale, BE 2009, 'Biofuels, land use change, and greenhouse gas emissions: some unexplored variables', *Environmental Science & Technology*, vol. 43, no. 3, pp. 961-7.

Kim, S & Dale, BE 2008, 'Effects of nitrogen fertilizer application on greenhouse gas emissions and economics of corn production', *Environmental Science & Technology*, vol. 42, no. 16, pp. 6028-33.

Klein-Marcuschamer, D, Turner, C, Allen, M, Gray, P, Dietzgen, RG, Gresshoff, PM, Hankamer, B, Heimann, K, Scott, PT, Stephens, E, Speight, R & Nielsen, LK 2013, 'Technoeconomic analysis of renewable aviation fuel from microalgae, Pongamia pinnata, and sugarcane', *Biofuels, Bioproducts and Biorefining*, vol. 7, no. 4, pp. 416-28.

Kløverpris, J, Wenzel, H, Banse, M, Milà i Canals, L & Reenberg, A 2008, 'Conference and workshop on modelling global land use implications in the environmental assessment of biofuels', *International Journal of Life Cycle Assessment*, vol. 13, no. 3, pp. 178-83.

Kløverpris, JH, Baltzer, K & Nielsen, PH 2010, 'Life cycle inventory modelling of land use induced by crop consumption', *International Journal of Life Cycle Assessment*, vol. 15, no. 1, pp. 90-103.

Koh, LP & Ghazoul, J 2010, 'Spatially explicit scenario analysis for reconciling agricultural expansion, forest protection, and carbon conservation in Indonesia', *Proceedings of the National Academy of Sciences of the United States of America*, vol. 107, no. 24, pp. 11140-4.

Koh, LP, Miettinen, J, Liew, SC & Ghazoul, J 2011, 'Remotely sensed evidence of tropical peatland conversion to oil palm', *Proceedings of the National Academy of Sciences of the United States of America*, vol. 108, no. 12, pp. 5127-32.

Koh, LP & Wilcove, DS 2008, 'Is oil palm agriculture really destroying tropical biodiversity?', *Conservation Letters*, vol. 1, no. 2, pp. 60-4.

Krautgartner, R, Henard, M-C, Lieberz, S, Boshnakova, M, Flach, B, Wilson, J, Wideback, A, Bettini, O, Guerrero, M & Bendz, K 2011, *EU-27, Oilseeds and products annual—modest rebound in EU-27 oilseeds production*, USDA Foreign Agricultural Service.

Kroon, FJ, Kuhnert, PM, Henderson, BL, Wilkinson, SN, Kinsey-Henderson, A, Abbott, B, Brodie, JE & Turner, RDR in press, 'River loads of suspended solids, nitrogen, phosphorus and herbicides delivered to the Great Barrier Reef lagoon', *Marine Pollution Bulletin*.

Laborde, D 2011, *Assessing the land use change consequences of European biofuels policies*, International Food Policy Research Institute (IFPRI).

Lambin, EF, Geist, H & Rindfuss, RR 2006, 'Introduction: Local Processes with global impacts (Chapter 1)', in EF Lambin & H Geist (eds), *Land-use and land-cover change: Local processes and global impacts*, Springer, Berlin.

Lane, MB, Taylor, B & Robinson, C 2009, 'Introduction: contested country - regional natural resource management in Australia', in *Contested country: local and regional natural resources management in Australia*, CSIRO Publishing, p. 261.

Langeveld, H, Quist-Wessel, F, Dimitriou, I, Aronsson, P, Baum, C, Schulz, U, Bolte, A, Baum, S, Köhn, J, Weih, M, Gruss, H, Leinweber, P, Lamersdorf, N, Schmidt-Walter, P & Berndes, G 2012, 'Assessing environmental impacts of Short Rotation Coppice (SRC) expansion: Model definition and preliminary results', *BioEnergy Research*, vol. 5, no. 3, pp. 621-35.

Langpap, C & Wu, J 2011, 'Potential environmental impacts of increased reliance on corn-based bioenergy', *Environmental and Resource Economics*, vol. 49, no. 2, pp. 147-71.

Lapola, DM, Priess, JA & Bondeau, A 2009, 'Modeling the land requirements and potential productivity of sugarcane and jatropha in Brazil and India using the LPJmL dynamic global vegetation model', *Biomass & Bioenergy*, vol. 33, no. 8, pp. 1087-95.

Lapola, DM, Schaldach, R, Alcamo, J, Bondeau, A, Koch, J, Koelking, C & Priess, JA 2010, 'Indirect land-use changes can overcome carbon savings from biofuels in Brazil', *Proceedings of the National Academy of Sciences of the United States of America*, vol. 107, no. 8, pp. 3388-93.

Lattimore, B, Smith, CT, Titus, B, Stupak, I & Egnell, G 2013, 'Woodfuel harvesting: A review of environmental risks, criteria and indicators, and certification standards for environmental sustainability', *Journal of Sustainable Forestry*, vol. 32, no. 1-2, pp. 58-88.

Le, PVV, Kumar, P & Drewry, DT 2011, 'Implications for the hydrologic cycle under climate change due to the expansion of bioenergy crops in the Midwestern United States', *Proceedings of the National Academy of Sciences of the United States of America*, vol. 108, no. 37, pp. 15085-90.

Lee, DJ, Huth, JR, Osborne, DO & Hogg, BW 2010, 'Selecting hardwood taxa for wood and fibre production in Queensland's subtropics', *Australian Forestry*, vol. 73, no. 2, pp. 106-14.

Lee, DJ, Brawner, JT, Smith TE, Hogg, BW, Meder, R & DO Osborne 2011, *Productivity of plantation forest tree species in northern-eastern Australia: A report from the Forest Adaptation and Sequestration Alliance*. Australian Government, Department of Agriculture, Fisheries and Forestry. Canberra.

Loarie, SR, Lobell, DB, Asner, GP, Mu, Q & Field, CB 2011, 'Direct impacts on local climate of sugar-cane expansion in Brazil', *Nature Climate Change*, 17 April 2011.

Love, B & Nejadhashemi, AP 2011, 'Environmental impact analysis of biofuel crops expansion in the Saginaw river watershed', *Journal of Biobased Materials and Bioenergy*, vol. 5, no. 1, pp. 30-54.

Love, BJ & Nejadhashemi, AP 2011, 'Water quality impact assessment of large-scale biofuel crops expansion in agricultural regions of Michigan', *Biomass and Bioenergy*, vol. 35, no. 5, pp. 2200-16.

Malczewski, J 2004, 'GIS-based land-use suitability analysis: a critical overview', *Progress in Planning*, vol. 62, no. 1, pp. 3-65.

Margono, BA, Turubanova, S, Zhuravleva, I, Potapov, P, Tyukavina, A, Baccini, A, Goetz, S & Hansen, MC 2012, 'Mapping and monitoring deforestation and forest degradation in Sumatra (Indonesia) using Landsat time series data sets from 1990 to 2010', *Environmental Research Letters*, vol. 7.

Maron, M & Kennedy, S 2007, 'Roads, fire and aggressive competitors: Determinants of bird distribution in subtropical production forests', *Forest Ecology and Management*, vol. 240, no. 1-3, pp. 24-31.

Martin, F 2011, 'Agroecological Zoning for Biofuels: The Brazilian experience', in *IEA Bioenergy: Quantifying and managing land use effects of bioenergy, 19-20 September 2011*, Campinas, Brazil.

Martin, TG & Catterall, CP 2001, 'Do fragmented coastal heathlands have habitat value to birds in eastern Australia?', *Wildlife Research*, vol. 28, no. 1, pp. 17-31.

Martin, TG, McIntyre, S, Catterall, CP & Possingham, HP 2006, 'Is landscape context important for riparian conservation? Birds in grassy woodland', *Biological Conservation*, vol. 127, no. 2, pp. 201-14.

Martinelli, LA & Filoso, S 2008, 'Expansion of sugarcane ethanol production in Brazil: Environmental and social challenges', *Ecological Applications*, vol. 18, no. 4, pp. 885-98.

Matondi, PB, Havnevik, K & Beyene, A 2011, *Biofuels, land grabbing and food security in Africa,* Zed Books, London, New York.

Matthews, T 1997a, *Landscapes of change: A history of the South Burnett*, vol.1, Paticipating members of the South Burnett local government association, Queensland.

—— 1997b, *Landscapes of change: A history of the South Burnett*, vol.2, Perticipating members of the South Burnett local government association., Queensland.

Mayaux, P, Bartholomé, E, Massart, M, Cutsem, CV, Cabral, A, Nonguierma, A, Diallo, O, Pretorius, C, Thompson, M, Cherlet, M, Pekel, J-F, Defourny, P, Vasconcelos, M, Gregorio, AD, S.Fritz, Grandi, GD, C..Elvidge, P.Vogt & Belward, A 2003, *A land cover map of Africa*, European Commission Joint Research Centre, Luxembourg.

McAlpine, CA, Etter, A, Fearnside, PM, Seabrook, L & Laurance, WF 2009, 'Increasing world consumption of beef as a driver of regional and global change: A call for policy action based on evidence from Queensland (Australia), Colombia and Brazil', *Global Environmental Change*, vol. 19, no. 1, pp. 21-33.

McBride, AC, Dale, VH, Baskaran, LM, Downing, ME, Eaton, LM, Efroymson, RA, Garten Jr, CT, Kline, KL, Jager, HI, Mulholland, PJ, Parish, ES, Schweizer, PE & Storey, JM 2011, 'Indicators to support environmental sustainability of bioenergy systems', *Ecological Indicators*, vol. 11, no. 5, pp. 1277-89.

McCarroll, SM & Brough, DM 2000, *Soil attributes and agricultural suitability of the Burnett River riparian lands, Gayndah-Perry River*, Queensland Department of Natural Resources Brisbane.

McGarigal, K, Cushman, S & Ene, E 2012, *FRAGSTATS v4: Spatial pattern analysis program for categorical and continuous maps*, Computer software program produced by the authors at the University of Massachusetts, Amherst,

<http://www.umass.edu/landeco/research/fragstats/fragstats.html>.

McGarigal, K & Marks, BJ 1995, *FRAGSTATS. Spatial pattern analysis program for quantifying landscape structure*, Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon.

McGarigal, K, S. A. , Cushman, M, Neel, C & Ene., E 2002, *FRAGSTATS: Spatial pattern analysis program for categorical maps*, Computer software program produced by the authors at the University of Massachusetts, viewed 28 May 2010,

<http://www.umass.edu/landeco/research/fragstats/fragstats.html>.

McKergow, LA, Prosser, IP, Hughes, AO & Brodie, J 2005, 'Sources of sediment to the Great Barrier Reef World Heritage Area', *Marine Pollution Bulletin*, vol. 51, no. 1-4, pp. 200-11.

Mehaffey, M, Smith, E & van Remortel, R 2012, 'Midwest U.S. landscape change to 2020 driven by biofuel mandates', Ecological Applications, vol. 22, no. 1, pp. 8-19.

Menichetti, E & Otto, M 2009, 'Energy balance and greenhouse gas emissions of biofuels from a life-cycle perspective', in RW Howarth & S Bringezu (eds), *Biofuels: Environmental consequences and interactions with changing land use. Proceedings of the scientific committee on problems of the environment (SCOPE) International Biofuels Project Rapid Assessment, 22-25 September 2008*, Gummersbach Germany, pp. 81-109.

Miller, SA, Landis, AE & Theis, TL 2007, 'Environmental trade-offs of biobased production', *Environmental Science & Technology*, vol. August 1, 2007, pp. 5176-82.

Mitchell, D 2008, *A note on rising food prices*, The World Bank.

Mitchell, TD & Osborn, TJ 2005, 'ClimGen: a flexible tool for generating monthly climate data sets and scenarios', paper presented to *Tyndall Centre for Climate Change Research Working Paper*.

Miyake, S, Renouf, M, Peterson, A, McAlpine, C & Smith, C 2012, 'Land-use and environmental pressures resulting from current and future bioenergy crop expansion: A review', *Journal of Rural Studies*, vol. 28, no. 4 pp. 650-8

MMA 2006, *Classificação Supervisionada a partir de imagens Landsat 2000, Mapeamento Temático de Uso da Terra no Submédio São Francisco (Supervised classification from 2000 Landsat imagery, Land use map in São Francisco River catchment)*, 1:100,000 Ministério do Meio Ambiente (MMA).

MoF 2010, 'Digital forest land use, land cover and administrative boundary maps ', Ministry of Forestry of Indonesia, <http://webgis.dephut.go.id/ditplanjs/index.html>

Morton, DC, DeFries, RS, Shimabukuro, YE, Anderson, LO, Arai, E, Espirito-Santo, FdB, Freitas, R & Morisette, J 2006, 'Cropland expansion changes deforestation dynamics in the southern Brazilian Amazon', *Proceedings of the National Academy of Sciences of the United States of America*, vol. 103, no. September 26, 2006.

Moss, A, Scheltinga, D & Tilden, J 2008, *State of the estuarine environment report for the Burnett Mary NRM region 2008*, Burnett Mary Regional Group, Bundaberg.

Mueller, CC 2003, *Expansion and modernization of agriculture in the Cerrado – the case of soybeans in Brazil's Center-West*, Department of Economics, University of Brazilia.

Murphy, HT, O'Connell, DA, Seaton, G, Raison, RJ, Rodriguez, LC, Braid, AL, Kriticos, DJ, Jovanovic, T, Abadi, A, Betar, M, Brodie, H, Lamont, M, McKay, M, Muirhead, G, Plummer, J, Arpiwi, NL, Ruddle, B, Saxena, S, Scott, PT, Stucley, C, Thistlethwaite, B, Wheaton, B, Wylie, P & Gresshoff, PM 2012, 'A common view of the opportunities, challenges, and research actions for Pongamia in Australia', *BioEnergy Research*, vol. 5, no. 3, pp. 778-800.

Nantha, HS & Tisdell, C 2009, 'The orangutan-oil palm conflict: economic constraints and opportunities for conservation', *Biodiversity and Conservation*, vol. 18, no. 2, pp. 487-502.

NASA Jet Propulsion Laboratory 2000, *Shuttle Radar Topography Mission (SRTM)*, viewed 18 March 2013, <http://www2.jpl.nasa.gov/srtm/>.

Naylor, RL, Liska, AJ, Burke, MB, Falcon, WP, Gaskell, JC, Rozelle, SD & Cassman, KG 2007, 'The ripple effect: Biofuels, food security, and the environment', *Environment*, vol. 49, no. 9, pp. 30-43.

Nearing, MA, Norton, LD & Zhang, X 2001, 'Chapter 2: Soil erosion and sedimentation', in WF Ritter & A Shirmohammadi (eds), *Agricultural nonpoint source pollution: Watershed management and hydrology*, Lewis publishers, Boca Raton.

Nepstad, DC, Stickler, CM, Soares, B & Merry, F 2008, 'Interactions among Amazon land use, forests and climate: Prospects for a near-term forest tipping point', *Philosophical Transactions of the Royal Society B-Biological Sciences*, vol. 363, no. 1498, pp. 1737-46.

Ng, TL, Eheart, JW, Cai, X & Miguez, F 2010, 'Modeling miscanthus in the Soil and Water Assessment Tool (SWAT) to simulate its water quality effects as a bioenergy crop', *Environmental Science and Technology*, vol. 44, no. 18, pp. 7138-44.

O'Connell, D., May, B., Farine, D., Raison, J., Herr, A., O'Connor, M., Campbell, P., Taylor, J., Dunlop, M., Poole, M., Crawford, D. 2009. 'Substitution of fossil fuels with bioenergy from renewable biomass resources', In: Eady, S., Grundy, M., Battaglia, M., Keating, B. (Eds.), *An analysis of greenhouse gas mitigation and carbon biosequestration opportunities from rural land use,* August 2009. CSIRO, St Lucia, Queensland.

O'Connor, GF 1948, *The Central and Upper Burnett River District of Queensland: centenary souvenir, 1848-1948 : embracing the districts of Gayndah, Munduberra, Eidsvold and Monto*, Brisbane.

Odeh, I & Tan, D 2007, 'Expanding biofuel production in Australia: opportunities beyond the horison', *Farm Policy Journal*, vol. 4, no. 2 May Quarter 2007, pp. 29-39.

Odeh, IOA, Tan, DKY & Ancev, T 2011, 'Potential suitability and viability of selected biodiesel crops in Australian marginal agricultural lands under current and future climates', *BioEnergy Research*, vol. 4, no. 3, pp. 165-79.

OECD & FAO 2012, *Biofuel, OECD-FAO Agricultural Outlook 2011-2020*, July 7, 2012, <http://www.agri-outlook.org/document/0/0,3746,en_36774715_36775671_47877696_1_1_1_1,00.html>.

OECD & International Transport Forum 2008, *ITF round tables no. 138 biofuels: Linking support to performance*, OECD Publications, Paris.

OESR 2012, *Population and dwelling profile: Bundaberg Regional Council*, <http://www.oesr.qld.gov.au/products/profiles/pop-housing-profiles-lga/pop-housing-profile-bundaberg.pdf>.

Osher, LJ, Matson, PA & Amundson, R 2003, 'Effect of land use change on soil carbon in Hawaii', *Biogeochemistry*, vol. 65, no.2, pp.213-232.

Oyama, MD & Nobre, CA 2004, 'Climatic consequences of a large-scale desertification in northeast Brazil: A GCM simulation study', *Journal of Climate*, vol. 17, no. 16, pp. 3203-13.

Pacheco, P 2009, 'Agrarian reform in the Brazilian Amazon: Its implications for land distribution and deforestation', *World Development*, vol. 37, no. 8, pp. 1337-47.

Panichelli, L, Dauriat, A & Gnansounou, E 2009, 'Life cycle assessment of soybean-based biodiesel in Argentina for export', *International Journal of Life Cycle Assessment*, vol. 14, no. 2, pp. 144-59.

Payne, WA 2010, 'Are biofuels antithetic to long-term sustainability of soil and water resources? (Chapter 1)', *Advances in Agronomy*, vol. 105, pp. 1-46.

Perlack, RD, Wright, LL, Turhollow, AF, Graham, RL, Stokes, BJ & Erbach, DC 2005, *Biomass as feedstock for a bioenergy and bioproducts industry: The technical feasibility of a billion-ton annual supply*, U.S. Department of Energy.

Petersen, J-E 2008, 'Energy production with agricultural biomass: environmental implications and analytical challenges', *European Review of Agricultural Economics. October 8, 2008.*

Pimentel, D & Patzek, TW 2005, 'Ethanol production using corn, switchgrass, and wood; biodiesel production using soybean and sunflower', *Natural Resources Research*, vol. 14, no. 1, pp. 65-76.

Pineiro, G, Jobbagy, EG, Baker, J, Murray, BC & Jackson, RB 2009, 'Set-asides can be better climate investment than corn ethanol', *Ecological Applications*, vol. 19, no. 2, pp. 277-82.

Pousa, GPAG, Santos, ALF & Suarez, PAZ 2007, 'History and policy of biodiesel in Brazil', *Energy Policy*, vol. 35, no. 11, pp. 5393-8.

Powlson, DS, Riche, AB & Shield, I 2005, 'Biofuels and other approaches for decreasing fossil fuel emissions from agriculture', *Annals of Applied Biology*, vol. 146, no. 2, pp. 193-201.

Prins, AG, Stehfest, E, Overmars, K & Ros, J 2010, *Are models suitable for determining ILUC factors*, Netherlands Environmental Assessment Agency, Bilthoven, The Netherlands.

Puckett, LJ 1995, 'Identifying the major sources of nutrient water pollution', *Environmental Science & Technology*, vol. 29, no. 9, pp. 408A-14A.

Pulsford, JS 1996, *Historical nutrient usage in coastal Queensland river catchments adjacent to the Great Barrier Reef Marine Park.*, Great Barrier Reef Marine Park Authority, Townsville.

Qpzm 2013, *Gayndah population (QLD)*,
<http://localstats.qpzm.com.au/population/qld/central-coast/bundaberg-wide-bay/gayndah>

Queensland CRA RFA Steering Committee 1998, *Commercial plantation land capability analysis of Southeast Queensland*.

Queensland Department of Science Information Technology Innovation and the Arts 2012, *Foliage Projective Cover (FPC) data 2005*, Brisbane.

Queensland Department of the Premier and Cabinet 2009a, *Paddock to reef program: Integrated monitoring, modelling and reporting*, The State of Queensland, Brsibane.

—— 2009b, *Reef Water Quality Protection Plan 2009—for the Great Barrier Reef World Heritage Area and adjacent catchments*, The State of Queensland, Brisbane.

Queensland Government 2009a, *Paddock to Reef Program: Integrated monitoring, modelling and reporting: Reef water quality plan*, The State of Queensland Department of the Premier and Cabinet, Brisbane.

—— 2009b, *Vegetation Management and Other Legislation Amendment Bill 2009*, <http://www.legislation.qld.gov.au/Bills/53PDF/2009/VMOLAB09.pdf>.

—— 2013, *Vegetation Management Framework Amendament Act 2013*,

<https://www.legislation.qld.gov.au/LEGISLTN/ACTS/2013/13AC024.pdf>.

Queensland Government 2014, Biofuels and bioenergy projects in Queensland viewed 1 March 2014, <http://www.business.qld.gov.au/industry/energy/renewable-energy/renewable-energy-research-projects-in-queensland/biofuels-and-bioenergy-projects-in-queensland>.

Quirin, M, Gärtner, S, Pehnt, M & Reindhardt, G 2004, *CO_2 mitigation through biofuels in the transport sector*, Institute for Energy and Environmental Research Heidelberg (IFEU), Heidelberg, Germany.

Radford, JQ, Bennett, AF & Cheers, GJ 2005, 'Landscape-level thresholds of habitat cover for woodland-dependent birds', *Biological Conservation*, vol. 124, no. 3, pp. 317-37.

Rajagopal, D 2007, 'Rethinking current strategies for biofuel production in India', paper presented to *Linkages between energy and water management for agriculture in developing countries*, ICRISAT campus, Hyderabad, India, 29-31 January, 2007,

<http://www.jatropha.pro/PDF%20bestanden/Rethinking%20current%20strategies%20for%20Biofuel%20production%20in%20India.pdf>.

REDD 2010, REDD (Reducing Emissions from Deforestation and Forest Degradation) + partnership, *Oslo Forest Climate Conference 2010* (May 2010), Oslo, Norway, <http://www.oslocfc2010.no/>.

REDD in Indonesia 2010, *Indonesia moves on forest clearing moratorium*, viewed 8 November 2010, <http://redd-indonesia.org/en/news-room/detail-news/read/indonesia-moves-on-forest-clearing-moratorium/>.

Reijnders, L 2009, 'Acute view transport biofuels: Can they help limiting climate change without an upward impact on food prices?', *Journal Fur Verbraucherschutz Und Lebensmittelsicherheit-Journal of Consumer Protection and Food Safety*, vol. 4, no. 1, pp. 75-8.

Reijnders, L & Huijbregts, M 2009, *Biofuels for road transport - A seed to wheel perspective*, Springer, London.

Renouf, MA, Wegener, JK & Nielsen, LK 2008, 'An environmental life cycle assessment comparing Australian sugarcane with US corn and UK sugar beet as producers of sugars for fermentation', *Biomass and Bioenergy*, vol. 32, no. 12, pp. 1144-55.

RFA 2010, *Statistics*, Renewable Fuels Association, viewed 21 October 2010, <http://www.ethanolrfa.org/pages/statistics>.

RIRDC 2013, *Research project details: developing a new renewable fuel and food industry in Australia: Sweet sorghum*, viewed 11 April 2013, <http://www.rirdc.gov.au/research-project-details/custr10_BBE/PRJ-005254>.

Roberts, MG, Male, TD & Toombs, TP 2007, *Potential impacts of biofuels expansion on natural resources: A case study of the Ogallala Aquifer region*, Environmental Defence.

Robertson, BA, Doran, PJ, Loomis, LR, Robertson, JR & Schemske, DW 2011, 'Perennial biomass feedstocks enhance avian diversity', *Global Change Biology Bioenergy*, vol. 3, no. 3, pp. 235-46.

Robertson, GP, Dale, VH, Doering, OC, Hamburg, SP, Melillo, JM, Wander, MM, Parton, WJ, Adler, PR, Barney, JN, Cruse, RM, Duke, CS, Fearnside, PM, Follett, RF, Gibbs, HK, Goldemberg, J, Mladenoff, DJ, Ojima, D, Palmer, MW, Sharpley, A, Wallace, L, Weathers, KC, Wiens, JA & Wilhelm, WW 2008, 'Agriculture - Sustainable biofuels Redux', *Science*, vol. 322, no. 5898, pp. 49-50.

Rosser, J, Swartz, GL, Dawson, NM & Briggs, HS 1974, *A land capability classification for agricultural purposes*, Queensland Department of Primary Industries, Division of Land Utilisation.

Rounsevell, MDA & Reay, DS 2009, 'Land use and climate change in the UK', *Land Use Policy*, vol. 26, no. Supplement 1, pp. S160-S9.

Rounsevell, MDA, Reginster, I, Araujo, MB, Carter, TR, Dendoncker, N, Ewert, F, House, JI, Kankaanpaa, S, Leemans, R, Metzger, MJ, Schmit, C, Smith, P & Tuck, G 2006, 'A coherent set of future land use change scenarios for Europe', *Agriculture Ecosystems & Environment*, vol. 114, no. 1, pp. 57-68.

Rowe, RL, Hanley, ME, Goulson, D, Clarke, DJ, Doncaster, CP & Taylor, G 2011, 'Potential benefits of commercial willow Short Rotation Coppice (SRC) for farm-scale plant and invertebrate communities in the agri-environment', *Biomass and Bioenergy*, vol. 35, no. 1, pp. 325-36.

Rowe, RL, Street, NR & Taylor, G 2009, 'Identifying potential environmental impacts of large-scale deployment of dedicated bioenergy crops in the UK', *Renewable & Sustainable Energy Reviews*, vol. 13, no. 1, pp. 271-90.

Sampaio, G, Nobre, C, Costa, MH, Satyamurty, P, Soares, BS & Cardoso, M 2007, 'Regional climate change over eastern Amazonia caused by pasture and soybean cropland expansion', *Geophysical Research Letters*, vol. 34, no. 17, p. 7.

Sanderson, MA & Adler, PR 2008, 'Perennial forages as second generation bioenergy crops', *International Journal of Molecular Sciences*, vol. 9, no. 5, pp. 768-88.

Sawyer, D 2008, 'Climate change, biofuels and eco-social impacts in the Brazilian Amazon and Cerrado', *Philosophical Transactions of the Royal Society B-Biological Sciences*, vol. 363, no. 1498, pp. 1747-52.

Schilling, KE, Jha, MK, Zhang, YK, Gassman, PW & Wolter, CF 2008, 'Impact of land use and land cover change on the water balance of a large agricultural watershed: Historical effects and future directions', *Water Resources Research*, vol. 44, pp. -.

Schubert, R, Schellnhuber, HJ, Buchmann, N, Epiney, A, Grießhammer, R, Kulessa, M, Messner, D, Rahmstorf, S & Schmid, J 2010, *Future bioenergy and sustainable land use*, Earthscan, London ; Sterling, VA.

Scott, P, Pregelj, L, Chen, N, Hadler, J, Djordjevic, M & Gresshoff, P 2008, 'Pongamia pinnata: An untapped resource for the biofuels industry of the future', *BioEnergy Research*, vol. 1, no. 1, pp. 2-11.

Seabrook, L, McAlpine, C & Fensham, R 2006, 'Cattle, crops and clearing: Regional drivers of landscape change in the Brigalow Belt, Queensland, Australia, 1840-2004', *Landscape and Urban Planning*, vol. 78, no. 4, pp. 373-85.

Searchinger, T, Heimlich, R, Houghton, RA, Dong, FX, Elobeid, A, Fabiosa, J, Tokgoz, S, Hayes, D & Yu, TH 2008, 'Use of US croplands for biofuels increases greenhouse gases through emissions from land-use change', *Science*, vol. 319, no. 5867, pp. 1238-40.

Secchi, S, Kurkalova, L, Gassman, PW & Hart, C 2011, 'Land use change in a biofuels hotspot: The case of Iowa, USA', *Biomass and Bioenergy*, vol. 35, no. 6, pp. 2391-400.

Semere, T & Slater, FM 2007, 'Ground flora, small mammal and bird species diversity in miscanthus (Miscanthus x giganteus) and reed canary-grass (Phalaris arundinacea) fields', *Biomass & Bioenergy*, vol. 31, no. 1, pp. 20-9.

Sharma, S, Negi, M, Sinha, P, Kumar, K & Tripathi, SB 2011, 'Assessment of genetic diversity of biodiesel spieces Pongamia pinnata assessing using AFLP and three Endonuclease-AFLP', *Plant Molecular Biology Reporter*, vol.29, pp.12-18.

Sharpley, AN 2002, 'Introduction: Agriculture as a potential source of water pollution', in PM Haygarth & SC Jarvis (eds), *Agriculture, hydrology and water quality*, CABI Publishing, Wallingford, UK; New York.

Shepherd, M, Bartle, J, Lee, DJ, Brawner, J, Bush, D, Turnbull, P, Macdonell, P, Brown, TR, Simmons, B & Henry, R 2011a, 'Eucalypts as a biofuel feedstock', *Biofuels*, vol. 2, no. 6, pp. 639-57.

Shepherd, M, Lee, D, Turnbull, P, Brown, T, Bush, D, Brawner, J, Bartle, J, Macdonell, P & Henry, R 2011b, *Eucalypts for biofuel production in northern Australia: Identifying species from current and future testing programs: A report for the Forest Industries Climate Change Research Fund (FICCRF) project "Assessment of forest species for bioenergy production in northern Australia"*, Rural Industries Research and Development Corporation.

Short, FT & Wyllie-Echeverria, S 1996, 'Natural and human-induced disturbance of seagrasses', *Environmental Conservation*, vol. 23, no. 1, pp. 17-27.

Shortall, OK 2013, '"Marginal land" for energy crops: Exploring definitions and embedded assumptions', *Energy Policy*, vol. 62, pp. 19-27.

Sims, R, Taylor, M, Saddler, J & Mabee, W 2008, *From 1st- to 2nd-generation biofuel technologies: an overview of current industry and RD&D activities* International Energy Agency and OECD, Paris.

Singh, A & Olsen, SI 2011, 'A critical review of biochemical conversion, sustainability and life cycle assessment of algal biofuels', *Applied Energy*, vol. 88, no. 10, pp. 3548-55.

Sioli, H 1985, 'The effects of deforestation in Amazonia', *The Geographical Journal*, vol. 151, no. 2, pp. 197-203.

Smith, GC & Agnew, G 2002, 'The value of 'bat boxes' for attracting hollow-dependent fauna to farm forestry plantations in southeast Queensland', *Ecological Management & Restoration*, vol. 3, no. 1, pp. 37-46.

Soille, P & Vogt, P 2009, 'Morphological segmentation of binary patterns', *Pattern Recognition Letters*, vol. 30, no. 4, pp. 456-9.

Sparovek, G, Barretto, A, Berndes, G, Martins, S & Maule, R 2009, 'Environmental, land-use and economic implications of Brazilian sugarcane expansion 1996-2006', *Mitigation and Adaptation Strategies for Global Change*, vol. 14, no. 3, pp. 285-98.

Sparovek, G, Berndes, G, Klug, ILF & Barretto, A 2010, 'Brazilian agriculture and environmental legislation: Status and future challenges', *Environmental Science & Technology*, vol. 44, no. 16, pp. 6046-53.

Stake, RE 1995, *The art of case study research*, Sage, Thousand Oaks, Calif ; London.

State of Calfornia Office of the Governor 2007, *Executive Order S-01-07*, by State of Calfornia Office of the Governor.

State of the Environment Committee 2011, *Australia state of the environment 2011: an independent report presented to the Australian Government Minister for Sustainability, Environment, Water, Population and Communities*, Department of Sustainability, Environment, Water, Population and Communities, Canberra.

Stoms, DM, Davis, FW, Jenner, MW, Nogeire, TM & Kaffka, SR 2012, 'Modeling wildlife and other trade-offs with biofuel crop production', *GCB Bioenergy*, vol. 4, no. 3, pp. 330-41.

Sudha, P, Somashekhar, HI, Rao, S & Ravindranath, NH 2003, 'Sustainable biomass production for energy in India', *Biomass and Bioenergy*, vol. 25, no. 5, pp. 501-15.

Sun, H & Cornish, PS 2005, 'Estimating shallow groundwater recharge in the headwaters of the Liverpool Plains using SWAT', *Hydrological Processes*, vol. 19, no. 3, pp. 795-807.

—— 2006, 'A catchment-based approach to recharge estimation in the Liverpool plains, NSW, Australia', *Australian Journal of Agricultural Research*, vol. 57, no. 3, pp. 309-20.

SWAT Development Team 2013, *SWAT, Soil & Water Assessment Tool*, viewed 8 July 2013, <http://swat.tamu.edu/>.

Tellis, W 1997, 'Application of a case study methodology ', *The Qualitative Report*, vol. 3, no. 3.

The Soybean and Corn Advisor 2010, *7% of Brazil's soybeans used for biodiesel production (June 7, 2010)*, viewed 28 June 2011, <http://www.soybeansandcorn.com/news/Jun7_10-7-of-Brazils-Soybeans-Used-for-Biodiesel-Production>.

Thoenes, P 2006, 'Biofuels and commodity markets—palm oil focus', paper presented to *The impact of biofuels on commodity markets* conference, Brussels, 24-25 October 2006, <http://www.fao.org/es/esc/common/ecg/629/en/full_paper_English.pdf>.

Tienwong, K, Dasananda, S & Navanugraha, C 2009, 'Integration of land evaluation and the analytical hierarchical process method for energy crops in Kanchanaburi, Thailand', *Scienceasia*, vol. 35, no. 2, pp. 170-7.

Tilman, D, Fargione, J, Wolff, B, D'Antonio, C, Dobson, A, Howarth, R, Schindler, D, Schlesinger, WH, Simberloff, D & Swackhamer, D 2001, 'Forecasting agriculturally driven global environmental change', *Science*, vol. 292, no. 5515, pp. 281-4.

Tilman, D, May, RM, Lehman, CL & Nowak, MA 1994, 'Habitat destruction and the extinction debt', *Nature*, vol. 371, no. 6492, pp. 65-6.

Tokgoz, S, Elobeid, A, Fabiosa, J, Hayes, DJ, Babcock, BA, Yu, T-H, Dong, F, Hart, CE & Beghin, JC 2007, *Emerging biofuels: Outlook of effects on U.S. grain, oilseed, and livestock markets*, Center for Agricultural and Rural Development (CARD).

Tolbert, VR, Todd, DE, Mann, LK, Jawdy, CM, Mays, DA, Malik, R, Bandaranayake, W, Houston, A, Tyler, D & Pettry, DE 2002, 'Changes in soil quality and below-ground carbon storage with conversion of traditional agricultural crop lands to bioenergy crop production', *Environmental Pollution*, vol. 116, pp. S97-S106.

Tomei, J, Semino, S, Paul, H, Joensen, L, Monti, M & Jelsoe, E 2010, 'Soy production and certification: the case of Argentinean soy-based biodiesel', *Mitigation and Adaptation Strategies for Global Change*, vol. 15, no. 4, pp. 371-94.

Trisurat, Y, Alkemade, R & Verburg, PH 2010, 'Projecting land-use change and its consequences for biodiversity in northern Thailand', *Environmental Management*, vol. 45, no. 3, pp. 626-39.

Tsao, CC, Campbell, JE, Mena-Carrasco, M, Spak, SN, Carmichael, GR & Chen, Y 2012, 'Increased estimates of air-pollution emissions from Brazilian sugar-cane ethanol', *Nature Climate Change*, vol. 2, no. 1, pp. 53-7.

Twine, TE, Kucharik, CJ & Foley, JA 2004, 'Effects of land cover change on the energy and water balance of the Mississippi River basin', *Journal of Hydrometeorology*, vol. 5, no. 4, pp. 640-55.

Tyner, WE 2013, 'Biofuels and food prices: Separating wheat from chaff', *Global Food Security,* vol. 2, no. 2, pp. 126-30.

Tyner, WE, Taheripour, F, Zhuang, Q, Birur, D & Baldos, U 2010, *Land use changes and consequent CO^2 emissions due to US corn ethanol production: A comprehensive analysis—final report*, Department of Agricultural Economics, Purdue University

U.S. Environmental Protection Agency 1997, *Compendium of tools for watershed assessment and TMDL development*, Office of Water, USEPA, Wasington, DC.

U.S. EPA 2009, *EPA proposes new regulations for the National Renewable Fuel Standard Program for 2010 and beyond (May 2009)*, U.S. Environmental Protection Agency, viewed September 17 2010, <http://www.epa.gov/OMS/renewablefuels/420f09023.htm>.

—— 2010, *Renewable Fuel Standard (RFS)*, U.S. Environmental Protection Agency, viewed January 3 2011, <http://www.epa.gov/otaq/fuels/renewablefuels/index.htm>.

—— 2013, *National greenhouse gas emissions data*, United States Environmental Protection Agency, viewed 6 June 2013,

<http://www.epa.gov/climatechange/ghgemissions/usinventoryreport.html>.

Ugarte, DGD, He, LX, Jensen, KL & English, BC 2010, 'Expanded ethanol production: Implications for agriculture, water demand, and water quality', *Biomass & Bioenergy*, vol. 34, no. 11, pp. 1586-96.

UMass Landscape Ecology Lab n.d., *Fragstats help*, viewed 7 August 2012, <http://www.umass.edu/landeco/research/fragstats/documents/fragstats.help.4.0.pdf>.

UNICA 2008a, *Brazilian ethanol exports*, Brazilian Sugarcane Industry Association, viewed 21 October 2010,
<http://english.unica.com.br/dadosCotacao/estatistica/>.

—— 2008b, *Brazilian ethanol production*, Brazilian Sugarcane Industry Association, viewed 28 October 2010,

<http://english.unica.com.br/dadosCotacao/estatistica/>.

USDA 2010, *Corn for grain 2009 - production by county and location of ethanol plants*, U.S. Department of Agriculture.

van der Hilst, F, Dornburg, V, Sanders, JPM, Elbersen, B, Graves, A, Turkenburg, WC, Elbersen, HW, van Dam, JMC & Faaij, APC 2010, 'Potential, spatial distribution and economic performance of regional biomass chains: The North of the Netherlands as example', *Agricultural Systems*, vol. 103, no. 7, pp. 403-17.

van Dam, J, Faaij, APC, Hilbert, J, Petruzzi, H & Turkenburg, WC 2009, 'Large-scale bioenergy production from soybeans and switchgrass in Argentina Part B: Environmental and socio-economic impacts on a regional level', *Renewable & Sustainable Energy Reviews*, vol. 13, no. 8, pp. 1679-709.

van Dam, J, Junginger, M & Faaij, APC 2010, 'From the global efforts on certification of bioenergy towards an integrated approach based on sustainable land use planning', *Renewable and Sustainable Energy Reviews*, vol. 14, no. 9, pp. 2445-72.

van der Hilst, F, Lesschen, JP, van Dam, JMC, Riksen, M, Verweij, PA, Sanders, JPM & Faaij, APC 2012a, 'Spatial variation of environmental impacts of regional biomass chains', *Renewable & Sustainable Energy Reviews*, vol. 16, no. 4, pp. 2053-69.

van der Hilst, F, Verstegen, JA, Karssenberg, D & Faaij, APC 2012b, 'Spatiotemporal land use modelling to assess land availability for energy crops - illustrated for Mozambique', *GCB Bioenergy*, vol. 4, no. 6, pp. 859-74.

van der Horst, D & Vermeylen, S 2011, 'Spatial scale and social impacts of biofuel production', *Biomass and Bioenergy*, vol. 35, no. 6, pp. 2435-43.

van Rooij, W 2008, *Using GLOBIO3 and CLUE methodology to calculate current and future status of biodiversity, Case study area: Zambia*, MNP Bilthoven, the Netherlands.

Vandersee, BE & Kent, DJ 1983, *Land resources of the Burnett region, Queensland. Part 1: South Burnett*, Queensland Department of Primary Industries, Brisbane.

Vanloocke, A, Bernacchi, CJ & Twine, TE 2010, 'The impacts of Miscanthus x giganteus production on the Midwest US hydrologic cycle', *Global Change Biology Bioenergy*, vol. 2, no. 4, pp. 180-91.

Varkkey, H 2012, 'Patronage politics as a driver of economic regionalisation: The Indonesian oil palm sector and transboundary haze', *Asia Pacific Viewpoint*, vol. 53, no. 3, pp. 314-29.

—— 2013, 'Patronage politics, plantation fires and transboundary haze', *Environmental Hazards*, vol. 12, no. 3-4, pp. 200-17.

Verstegen, JA, Karssenberg, D, van der Hilst, F & Faaij, A 2012, 'Spatio-temporal uncertainty in Spatial Decision Support Systems: A case study of changing land availability for bioenergy crops in Mozambique', *Computers, Environment and Urban Systems*, vol. 36, no. 1.

Vogt, P 2010, *MSPA guide (version 1.3)*, Institute for Environment and Sustainability (IES), European Commission, Joint Research Centre (JRC).

von Blottnitz, H & Curran, MA 2007, 'A review of assessments conducted on bio-ethanol as a transportation fuel from a net energy, greenhouse gas, and environmental life cycle perspective', *Journal of Cleaner Production*, vol. 15, no. 7, pp. 607-19.

Wachenfeld, DR, Oliver, JK & Morrissey, JI 1998, *State of the Great Barrier Reef World Heritage Area*, Great Barrier Reef Marine Park Authority, viewed 7 February 2013, <http://kurrawa.gbrmpa.gov.au/corp_site/info_services/publications/sotr/1998/>.

Walker, DH, Vella, KJ & Kotzman, M 2005, *Regional planning and the sugar industry*, CSIRO, CRC for Sustainable Sugar Production, Canberra.

Walsh, ME, De La Torre Ugarte, DG, Shapouri, H & Slinsky, SP 2003, 'Bioenergy crop production in the United States: Potential quantities, land use changes, and economic impacts on the agricultural sector', *Environmental and Resource Economics*, vol. 24, no. 4, pp. 313-33.

Waters, D & Carroll, C 2013, *Modelling of catchment pollutant load-reductions due to improved management practices in the Great Barrier Reef: Tier 2 technical report*, Queensland Department of Natural Resources and Mines.

Watson, BM, Srikanthan, R, Selvalingam, S & Ghafouri, M 2005, 'Evaluation of three daily rainfall generation models for SWAT', *Transactions of the Asae*, vol. 48, no. 5, pp. 1697-711.

Wegerner, MK 1997, 'Economic research projects related to land use in sugarcane growing area', in AKL Johnson, IB Robinson & MK Wegener (eds), *Coastal Queensland and the sugar industry—land use problems and opprtunities*, Gold Coast, pp. 26-32.

Wicke, B 2011, *Bioenergy production on degraded and marginal land: assessing its potential, economic performance, and environmental impacts for different settings and geographical scales* (PhD. thesis), PhD thesis, Utrecht University.

Wicke, B, Smeets, E, Watson, H & Faaij, A 2011, 'The current bioenergy production potential of semi-arid and arid regions in sub-Saharan Africa', *Biomass & Bioenergy*, vol. 35, no. 7, pp. 2773-86.

Wicke, B, Verweij, P, van Meijl, H, van Vuuren, DP & Faaij, APC 2012, 'Indirect land use change: review of existing models and strategies for mitigation', *Biofuels*, vol. 3, no. 1, pp. 87-100.

Wide Bay Burnett Environment and Natural Resources Working Group 2012, *Wide Bay Burnett environment and natural resource management plan 2012-2031*, Burnett Mary Regional Group for Natural Resource Management Ltd., Bundaberg.

Wiegmann, K, Hennenberg, KJ & Fritsche, UR 2008, 'Degraded land and sustainable bioenergy feedstock production—Issue paper', paper presented to the *Joint international workshop on high nature value criteria and potential for sustainable use of degraded lands,* Paris, 30 June to 1 July , 2008

<http://www.unep.fr/energy/activities/mapping/pdf/degraded.pdf>.

Wilcove, DS & Koh, LP 2010, 'Addressing the threats to biodiversity from oil-palm agriculture', *Biodiversity and Conservation*, vol. 19, no. 4, pp. 999-1007.

Williams, MR & Melack, JM 1997, 'Solute export from forested and partially deforested catchments in the central Amazon', *Biogeochemistry*, vol. 38, no. 1, pp. 67-102.

Williams, PRD, Inman, D, Aden, A & Heath, GA 2009, 'Environmental and sustainability factors associated with next-generation biofuels in the US: What do we really know?', *Environmental Science & Technology*, vol. 43, no. 13, pp. 4763-75.

Wilson, BA, Neldner, VJ & Accad, A 2002, 'The extent and status of remnant vegetation in Queensland and its implications for statewide vegetation management and legislation', *Rangeland Journal*, vol. 24, no. 1, pp. 6-35.

Wolfe, ML 2001, 'Chapter 1: Hydrology', in WF Ritter & A Shirmohammadi (eds), *Agricultural nonpoint source pollution*, Lewis publishers, Boca Raton.

Worldwatch Institute 2006, *Biofuels for transportation: Global potential and implications for sustainable agriculture and energy in the 21st century—extended summary* -, Prepared by the Worldwatch Institute for the German Federal Ministry of Food, Agriculture and Consumer Protection (BMELV), in cooperation with the Agency for Technical Cooperation (GTZ) and the Agency of Renewable Resourcs (FNR), Washington, D.C.

—— 2007, *Biofuels for transport: global potential and implications for sustainable energy and agriculture*, Earthscan, Sterling, VA

Wu, Y & Liu, S 2012, 'Impacts of biofuels production alternatives on water quantity and quality in the Iowa River basin', *Biomass and Bioenergy*, vol. 36, pp. 182-91.

Yang, YS & Wang, L 2010, 'A review of modelling tools for implementation of the EU water framework directive in handling diffuse water pollution', *Water Resources Management*, vol. 24, no. 9, pp. 1819-43.

Yeh, S & Witcover, J 2010, *Indirect land-use change from biofuels: Recent developments in modeling and policy landscapes*, International Food and Agricultural Trade Policy Council 2010

Yin, RK 2009, *Case study research : design and methods*, 4th edn, Applied social research methods, Sage, Los Angeles, Calif. ; London.

Zah, R, Böni, H, Gauch, M, Hischier, R, Lehmann, M & Wäger, P 2007, *Life cycle assessment of energy products: Environmental assessment of biofuels—executive summary—* Bern.

Zhang, X, Srinivasan, R & van Liew, M 2008, ‘Multi-site calibration of the SWAT model for hydrologic modelling’, *Transactions of the ASABE*, vol. 51, no.6, pp. 2039-49

Zhang, X, Izaurralde, RC, Manowitz, D, West, TO, Post, WM, Thomson, AM, Bandaru, VP, Nichols, J & Williams, JR 2010, 'An integrative modeling framework to evaluate the productivity and sustainability of biofuel crop production systems', *GCB Bioenergy*, vol. 2, no. 5, pp. 258-77.

国土交通省国土政策局国土整備課 2013, *国土数値情報　土地利用細分メッシュデータ*　viewed 15 March 2013, <http://nlftp.mlit.go.jp/ksj/jpgis/datalist/KsjTmplt-L03-b.html>.

Appendices

Page

Journal of Rural Studies 28 (2012) 650–658

Contents lists available at SciVerse ScienceDirect

Journal of Rural Studies

journal homepage: www.elsevier.com/locate/jrurstud

Review

Land-use and environmental pressures resulting from current and future bioenergy crop expansion: A review

Saori Miyake [a,*,1], Marguerite Renouf [a], Ann Peterson [a], Clive McAlpine [a], Carl Smith [b]

[a] The University of Queensland, School of Geography, Planning and Environmental Management, St. Lucia, 4072 Brisbane, Australia
[b] The University of Queensland, School of Agriculture and Food Sciences, St Lucia, 4072 Brisbane, Australia

Keywords:
Bioenergy
Biofuel
Land-use change
Deforestation
Abandoned agricultural land
Marginal and degraded land

A B S T R A C T

Recent energy and climate policies, particularly in the developed world, have increased demand for bioenergy[2] as an alternative, which has led to both direct and indirect land-use changes and an array of environmental and socio-economic concerns. A comprehensive understanding of the land-use dynamics of bioenergy crop production is essential for the development of sustainable bioenergy and land-use policies. In this paper, we review the patterns and dynamics of land-use change associated with bioenergy crops (hereafter referred to as 'bioenergy-driven land-use change'). The review focuses on four regions as the most prominent locations in which these patterns and changes occur: Brazil; Indonesia and Malaysia; the United States of America (U.S.A.); and the European Union (EU). The review confirms that bioenergy-driven land-use change has affected and will impact most severely on the 'land- and resource-abundant' developing regions, such as Brazil, where economic development takes priority over sustainable land-use policies, and the enforcement capability is limited. Opportunities for more effective policy are available through the development of international climate change policy (e.g. REDD under the UNFCCC), and certification criteria for sustainable bioenergy products (e.g. EU RED). However, bioenergy produced from no and/or less land-using feedstocks (e.g. wastes and residues), and their associated technologies must be given higher priority to minimise bioenergy-driven land-use change and its negative impacts.

1. Introduction

Since the mid-2000s, a growing number of governments have introduced bioenergy policies. They have stressed the benefits of bioenergy (especially biofuels) for climate change mitigation, improved energy security and rural development (European Commission, 2009; IEA, 2004; U.S. EPA, 2010). However, a rapid increase in demand for bioenergy production has led to various environmental and socio-economic concerns. It has been linked to the 'global food crisis' and sparked the food versus fuel debate (Mitchell, 2008; Matondi et al., 2011), which revolves around the current use of major food crops for biofuels feedstock (de Vries et al., 2010). Environmental concerns have also included deforestation and increased greenhouse gas (GHG) emissions, soil and water degradation, and biodiversity loss (Reijnders, 2009). Further, social polarisation (between large land holders and smallholder/landless farmers), displacement of communities, and the disregard for local land rights have been reported in developing countries (Worldwatch Institute, 2007). The production of dedicated bioenergy crops will most likely continue to place significant demands on land resources worldwide, even though high-yield plant species, such as non-food oil crops (e.g. pongamia, jatropha curcas) and lignocellulosic crops (e.g. switchgrass, short rotation woody crops) will be introduced in the medium- to long-term.

To meet global demand for biofuels, the International Energy Agency (IEA, 2011) estimated that 65 million hectares (ha) of land will be required by 2030, and 105 million ha by 2050. Several studies have suggested a link between bioenergy policies, demand for cropland, and adverse land-use changes (hereafter referred to 'bioenergy-driven land-use change'), especially deforestation in the 'South' (e.g. Searchinger et al., 2008). However, there is currently a lack of understanding of the sustainability of such land-use changes. Thus it is timely to review the existing research on bioenergy-driven land-use changes to inform sustainable bioenergy and land-use policies.

* Corresponding author. Tel.: +61 7 3365 4370; fax: +61 7 3365 6899.
E-mail address: s.miyake@uq.edu.au (S. Miyake).
[1] Permanent address: School of Geography, Planning and Environmental Management, The University of Queensland, St Lucia, 4072 Brisbane, Australia.
[2] Bioenergy generally encompasses a wide range of energy products, but in this paper it refers to electricity and biofuels (mainly ethanol and biodiesel used for transport fuel) generated from agricultural and forest crops, plants and their residues.

0743-0167/$ – see front matter © 2012 Elsevier Ltd. All rights reserved.
http://dx.doi.org/10.1016/j.jrurstud.2012.09.002

This paper reviews the dynamics of bioenergy-driven land-use change, with a focus on four geographic regions considered to be the most prominent locations for bioenergy-driven land-use changes - Brazil, Indonesia and Malaysia, the United States of America (U.S.A.), and the European Union (EU). For each region, the dynamics of bioenergy-driven land-use change were synthesised into land-use change pathways. Opportunities for land-use options and policy instruments that can reduce the impact of future bioenergy crop expansion are also identified.

2. Materials and methods

Our review included peer-reviewed publications and grey literature that report direct and indirect land-use changes associated with bionergy production. Direct land-use change (dLUC) is defined as the change in land-use on a site used for bioenergy crop production, while indirect land-use change (iLUC) comprises the unintended effects that occur elsewhere as a consequence of the displacement of existing crops or other land-uses, often in one or several countries outside the original country (Berndes et al., 2010; IEA Bioenergy, 2009). Bioenergy crop type and land-use prior to bioenergy crop conversion were used in the literature to describe the bioenergy-driven land-use change pathways within the four geographic regions covered in our review (Fig. 1). Information on land-use change drivers and time-scales were also recorded and used for this analysis.

3. Results

3.1. Brazil

Brazil has successfully established the world's second largest bioenergy production base (24.5 billion litres in 2008) and largest export market (5.1 billion litres in 2008) for ethanol fuel under its sugarcane ethanol fuel programme (Martinelli and Filoso, 2008; RFA, 2010; UNICA, 2010a, b). Sugarcane is one of the cheapest, most efficient and productive crops for ethanol production, with favourable energy balances and GHG emission potential (Renouf et al., 2008; von Blottnitz and Curran, 2007). The area of harvested sugarcane increased from 1.4 to 8.6 million ha between 1961 and 2009 (FAOSTAT, 2011), with production concentrated in the south-central region (UNICA, 2010b). The total planted area increased by 53% between 2004 and 2009 due to the increased demand for fuel ethanol (FAOSTAT, 2011). Under the Brazilian Biodiesel Program (PNPB), Brazil also has a legislated biodiesel target. Soybean oil is currently the major biodiesel feedstock (Pousa et al., 2007). The area of soybean production increased from 0.2 million ha in 1961 to 21.7 million ha in 2009 (FAOSTAT, 2011). This was driven by high global prices for soybean since 1990 (Fearnside, 2001), although only 7% of total soybean production was used to produce biodiesel in 2010 (The Soybean and Corn Advisor, 2010).

In Brazil, soybean production, cattle ranching, and more recently global demand for sugarcane ethanol, are the major drivers of the

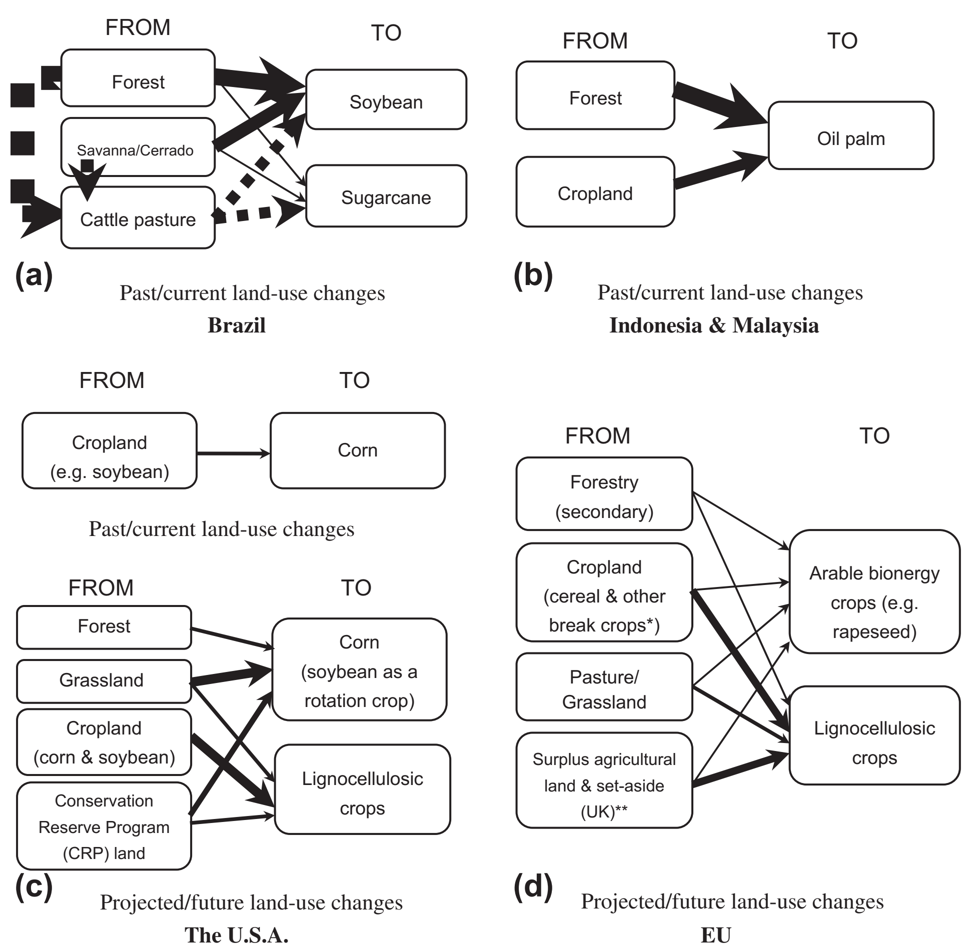


Fig. 1. Bioenergy-driven land-use change pathways in four geographic regions. Arrow width is proportional to the number of documented land-use changes in the reviewed literature, which includes both direct and indirect land-use changes. *Other break crops include linseed, lupins, drypeas and soybeans (Bauen et al., 2010). **Set-aside (UK): The set-aside measure under the EU's common agricultural policy (CAP) was abolished in 2008 after the global food price crisis, but the UK revived the concept as a voluntarily approach in 2009 due to the strong benefits for soil and water quality, and wildlife conservation (Campaign for the Farmed Environment, 2010).

conversion of native forests and savannas to agriculture (Barona et al., 2010; Brannstrom et al., 2008; Fearnside, 2005; McAlpine et al., 2009; Morton et al., 2006; Sawyer, 2008). The expansion of soybean and cattle pasture has resulted in large-scale deforestation of the Amazon rainforest (700,000 km^2 deforested) (Barona et al., 2010; Fearnside, 2001; Fearnside et al., 2009; Gibbs et al., 2010; Morton et al., 2006; Sawyer, 2008) and the Cerrado savanna (800,000–1,600,000 km^2 deforested) (Brannstrom et al., 2008; Fearnside, 2001, Fearnside et al., 2009; Mueller, 2003; Sawyer, 2008) (Fig. 1(a)). The geographic expansion of sugarcane is less than that for soybean production, because the Amazon and most of the Cerrado are not suited to sugarcane production (Goldemberg et al., 2008; Sparovek et al., 2009). However, many studies argued that recent expansion of sugarcane production areas and part of soybean production areas has occurred through the conversion of abandoned or degraded lands previously cleared for cattle pasture in the southern states,[3] and this has resulted in indirect deforestation by pushing displaced cattle ranching further into the frontier regions (Fig. 1 (a)) (Barona et al., 2010; Goldemberg et al., 2008; Lapola et al., 2010; Loarie et al., 2011; Nepstad et al., 2008; Sawyer, 2008). High international market prices for bioenergy crops have resulted in a rapid rise in agricultural land prices in the southern states (de Nie et al., 2009), which will continue to push cattle ranchers into frontier regions as they seek larger and cheaper tracts of land for grazing (Barona et al., 2010; McAlpine et al., 2009; Nepstad et al., 2008; Sawyer, 2008). A simulation based on Brazil's biofuels targets for 2020 estimates that sugarcane ethanol and soybean biodiesel will be responsible for 41% and 59% of indirect deforestation in Brazil respectively, the main mechanism for disforestation being displacement of cattle ranching by sugarcane in the south-eastern states and by soybean in the south-western states (Lapola et al., 2010).[3]

Brazil has a favourable climate for agricultural crop production, abundant land and water resources, low labour costs, and favourable government policies (Martinelli and Filoso, 2008; Naylor et al., 2007; Sawyer, 2008). Hence, the scarcity of suitable land for crop expansion in other countries has increased pressure to expand bioenergy crop production in Brazil (Nepstad et al., 2008). Brazil will continue to meet a large proportion of the future global demand for bioenergy, especially demand coming from the U.S.A. (Searchinger et al., 2008; Tyner et al., 2010) and from the EU (Al-Riffai et al., 2010; Banse et al., 2008; Blanco Fonseca et al., 2010; Hiederer et al., 2010; Laborde, 2011).

The Brazilian government has identified agricultural expansion and large-scale deforestation as a major challenge, and deforestation rates within Brazil as a result have slowed in recent years (INPE, 2012). However, the amended Forest Law [Law no. 4771/65] is regarded as being ineffective in restricting further clearing (Sparovek et al., 2010) due to inadequate legal enforcement and conflicting policy responses between environmental and other agencies that pursue economic development, agricultural interests, and land reform (McAlpine et al., 2009; Pacheco, 2009). Ambitious goals of the Brazilian government and the bioenergy industry to expand the production and export of biofuels still remain. A substantial land area has been identified for future expansion of sugarcane (64.7 million ha) and oil palm (more than 30 million ha) under the Brazilian government's Agroecological Zoning program (Martin, 2011).[4]

3.2. Indonesia and Malaysia

Palm oil is a valued ingredient in a number of food and cosmetic products. This has resulted in the rapid expansion of palm oil plantations in tropical developing regions, especially Southeast Asia. Palm oil has been the cheapest source of vegetable oil on the global market, and its higher yields and more favourable GHG and energy balance, compared to temperate oilseed crops, make it economically attractive as a biodiesel feedstock (Naylor et al., 2007; Thoenes, 2006; Worldwatch Institute, 2007). Indonesia and Malaysia are the largest producers and exporters of palm oil, accounting for 84% of the world palm oil production in 2008 (16.9 and 17.7 million tonnes, respectively; FAOSTAT, 2011). Both countries provide ideal agro-climatic conditions for oil palm production, low establishment and running costs for plantations, and hence high profitability (Nantha and Tisdell, 2009). The harvested area for oil palm has increased steadily in Malaysia since the 1970s (Abdullah and Hezri, 2008), while an exponential increase has occurred in Indonesia since the 1990s (FAOSTAT, 2011). Between 2000 and 2007, the harvested area of palm oil in the region increased by 63% (125% alone in Indonesia) from 5.1 to 8.3 million ha, which correlates with a doubling of Europe's palm oil imports (FAOSTAT, 2011). In addition to their internal biodiesel programs, both countries have announced the allocation of six million tonnes of palm oil for export, to meet the global demand for biodiesel, mainly from EU countries, the U.S.A. and other Asian countries (Biopact, 2006; Hoh, 2010; Thoenes, 2006).

Consistent with an earlier study by Koh and Wilcove (2008), our review identified that more than half the recent oil palm plantation expansion in Indonesia and Malaysia occurred at the expense of forests, with the remainder displacing existing cropland (e.g. rubber plantations in Malaysia, which were originally converted from primary rainforests before the 1970s) (Abdullah and Nakagoshi, 2007) (Fig. 1(b)). Thus palm oil plantations have been a major driver of deforestation in the region, with resulting increases in carbon emissions, habitat loss, and biodiversity decline including the endangered orang-utan (*Pongo abelii* and *Pongo pygmaeus*) (Danielsen et al., 2009; Koh et al., 2011; Koh and Wilcove, 2008; Nantha and Tisdell, 2009), and land disputes including the lack of clarity around land ownership and the displacement of indigenous people (Naylor et al., 2007).

The high economic return for oil palm has attracted large private sector investment, but also regional government support through economic development policies (Abdullah and Hezri, 2008; Nantha and Tisdell, 2009; Thoenes, 2006), which are frequently prioritised over environmental policies (Abdullah and Hezri, 2008). This was demonstrated by the granting of large-scale development permits for the conversion of primary rainforests to oil palm plantations in Indonesia until recently (Nantha and Tisdell, 2009). Future projections of land-use change in the region indicated that oil palm plantation was unlikely to expand further into existing cropland due to the decreasing availability, and instead would mainly occur through the conversion of primary forests (Bauen et al., 2010). However, international pressures may limit this trend. For example, the Indonesian government entered a partnership with the Reducing Emissions from Deforestation and Forest Degradation (REDD) program in May 2010, with an immediate two-year moratorium to stop issuing new permits for clearing primary forest and peatland (REDD, 2010; REDD in Indonesia, 2010).

3.3. The United States of America (U.S.A.)

Top-down biofuel policies and mandates under the National Renewable Fuel Standard (RFS) created under the Energy Policy Act of 2005 (U.S. EPA, 2009) were viewed as the primary cause of recent

[3] The south-eastern estates of Brazil refer to those in São Paulo, Minas Gerais, and Paraná, while the south-western states refer to those in Mato Grosso, Mato Grosso do Sul, and Goiás.

[4] The Sugarcane Agroecological Zoning prohibits sugarcane cultivation in the Amazon, in native ecosystems, and in areas with high conservation values, while oil palm zoning focuses on the recovery of degraded land within the Amazon basin and aims to provide socio-economic benefit to smallholder farmers (Martin, 2011).

bioenergy-driven land-use changes in the U.S.A. (Tyner et al., 2010). From the substantial increase in corn-based ethanol production since the mid-2000s, subsidies and programs under the RFS led to the U.S.A. overtaking Brazil as the largest fuel ethanol producer in the world (34 billion litres in 2008) (RFA, 2010). The area of corn harvested increased from 28.6 to 35 million ha between 2006 and 2007 alone (FAOSTAT, 2011).

The rapid corn-based ethanol expansion has led to a global debate about food security (Mitchell, 2008), the relative energy and GHG benefits of corn-based ethanol (Hammerschlag, 2006; Miller et al., 2007; Pimentel and Patzek, 2005; von Blottnitz and Curran, 2007), and its iLUC impacts (Searchinger et al., 2008; Tyner et al., 2010). The state of California was the first to respond by enacting a Low-Carbon Fuel Standard in 2007 (LCFS) (State of California Office of the Governor, 2007). Its eligibility criteria include life cycle GHG emissions from iLUC outside the country (California Air Resource Board, 2012). At the national level, the RFS is being revised (RFS2), under the Energy Independence and Security Act (EISA) of 2007, to expand the total biofuels mandate to 136 billion litres (36 billion gallons) by 2022, incorporating advanced biofuels,[5] cellulosic biofuel, and biodiesel. It also requires new GHG accounting methods taking into account the iLUC emissions (U.S. EPA, 2010).

Main land-use change pathway for bioenergy crops in the U.S.A. has been the result of increased corn production in the Corn Belt region (USDA, 2010), replacing existing soybean cropland (Mitchell, 2008; Schilling et al., 2008) (Fig. 1(c)). Around 22 million ha of cropland will be available for bioenergy crop production by 2050, although this may be insufficient to meet demand under the current national targets (Perlack et al., 2005). The total area under corn production is predicted to reach around 38 million ha by 2008–2016 (Tokgoz et al., 2007), replacing soybean cropland and lands under the Conservation Reserve Program (CRP)[6] (Secchi et al., 2011). Future production under the RFS2 targets also will be met through cellulosic biofuel from lignocellulosic crops (i.e. switchgrass and/or miscanthus) (Le et al., 2011; Ng et al., 2010; Schilling et al., 2008; Ugarte et al., 2010; Vanloocke et al., 2010), possibly grown on CRP lands (Fig. 1(c)) (Love and Nejadhashemi, 2011; Payne, 2010; Schilling et al., 2008; Wu and Liu, 2012). The use of CRP lands for large-scale lignocellulosic crop production has been proposed in the medium- to long-term (Graham et al., 1996; Hartman et al., 2011; Naylor et al., 2007; Payne, 2010; Walsh et al., 2003). However, this proposal has been controversial in terms of carbon emissions (Pineiro et al., 2009), natural resource management, and wildlife conservation (Payne, 2010; Roberts et al., 2007).

More importantly, the U.S. biofuel program has and will continue to cause land-use change outside the country. ILUC studies suggested that expansion of U.S. corn ethanol production could trigger large-scale conversion of native forest and grasslands to bioenergy crops worldwide, especially in Brazil, as a result of the displacement of existing crops, such as soybeans (Nepstad et al., 2008; Searchinger et al., 2008; Tyner et al., 2010).

3.4. European Union (EU)

The EU first introduced biofuel targets in 2003 [COM 2003/30/EC] (European Commission, 2003), under which various policy instruments have been used to increase bioenergy use, including fuel tax exemptions, mandates, import tariffs, and financial support for industry development (Blanco Fonseca et al., 2010). As a consequence, fuel ethanol produced from grains and sugar beet increased sevenfold between 2004 and 2009 to 3.9 million litres per annum (ePure, 2010), and biodiesel production from rapeseed increased sixfold between 2003 and 2009 to 7.9 billion litres per annum (EBB, 2010). The EU is a world leader in biodiesel production, with 65% of the global biodiesel output in 2009 (Biodiesel Magazine, 2010). There has been strong public support for the more costly, domestically produced feedstocks (Thoenes, 2006). As a result, the harvested area of rapeseed in the EU27[6,7] increased by 53% from 4.2 to 6.5 million ha between 2002 and 2009 (FAOSTAT, 2011), and diversion of domestically produced rapeseed oil from food to biodiesel occurred in line with the increased palm oil import from Southeast Asia (Krautgartner et al., 2011; Thoenes, 2006). Biodiesel use accounted for nearly two-thirds of the total EU27 produced rapeseed oil in 2011 (Krautgartner et al., 2011).

In 2009, the EU replaced existing bioenergy targets with the Renewable Energy Directive (RED) [2009/28], which sets targets of 20% renewable energy overall and 10% renewable transport energy by 2020 (European Commission, 2009). The RED introduces environmental sustainability criteria for production processes and a minimum rate of direct GHG emission savings for biofuels consumed in the EU, including GHG emissions from both dLUC and iLUC within and outside of the EU (European Commission, 2009). This has resulted in extensive research efforts for quantifying GHG emissions from bioenergy-driven land-use changes, using life cycle assessment (LCA) and the iLUC factor approach (Fritsche et al., 2010a,b).

To date, bioenergy-driven land-use change within the EU has been limited. However, the reviewed literature indicated future bioenergy demand will certainly influence land-use in the EU (Banse et al., 2011) (Fig. 1(d)). In EU15,[7] the demand for cropland and pasture for food production was expected to decrease, and then 'surplus' agricultural land would become available for future bioenergy crop production (Fischer et al., 2010; Rounsevell and Reay, 2009; Rounsevell et al., 2006). A major focus is on bioenergy production from lignocellulosic crops in the mid to longer term (Bellamy et al., 2009; Powlson et al., 2005; Rowe et al., 2009), with an increased future allocation of set-aside areas[8] for large-scale production of lignocellulosic crops in the UK (Powlson et al., 2005; Rowe et al., 2009, 2011) and in Europe (Fiorese and Guariso, 2010; Rowe et al., 2009).

Since land is limited in the EU, growing biofuel demand has started to impact on land resources outside the region. Besides the increasing vegetable oil imports, European companies have claimed over 5 million ha of land in the 'South', namely South America, Southeast Asia, and Africa for biofuel production (Borras et al., 2010; Matondi et al., 2011). Countries such as Spain have started importing soybean-based biodiesel from Argentina (Biodiesel Magazine, 2010), and this has raised environmental and social sustainability concerns (Grau et al., 2008; Panichelli et al., 2009; Tomei et al., 2010). To meet the biofuel target, EU countries

[5] 'Advanced fuel' refers to renewable fuel, other than ethanol derived from corn starch that has at least 50% less than baseline life cycle GHG emissions [The Energy Independence and Security Act of 2007].

[6] Conservation Reserve Program (CRP) is a voluntary set-aside program established by the U.S. Department of Agriculture (USDA) to remove highly erodible and environmentally sensitive land from agricultural production.

[7] The EU15 refers to EU member states before the enlargement in 2004: Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxembourg, the Netherlands, Portugal, Spain, Sweden and the United Kingdom. The EU27 includes EU15, and Bulgaria, the Czech Republic, Estonia, Cyprus, Latvia, Lithuania, Hungary, Malta, Poland, Romania, Slovenia, and Slovakia.

[8] Until 2008, set-aside was required under the EU's Common Agricultural Policy (CAP) to regulate food production, develop non-food crops, mitigate environmental impacts and support small farmers. The EU abolished the scheme in 2008, but the UK revived the concept as a voluntarily approach in 2009 in expectation of the strong environmental benefits.

will depend more on imported feedstock and processed biofuels, especially fuel ethanol, from countries where agricultural expansion is possible—Brazil, Argentina, Ukraine and other Commonwealth of Independent States (CIS) countries, the U.S.A. and Canada (Al-Riffai et al., 2010; Banse et al., 2008; Blanco Fonseca et al., 2010; Hiederer et al., 2010; Laborde, 2011). This may involve further conversion of primary forests, savannas, and grasslands to bioenergy crops in these countries. For example, in Brazil, 58% of cropland extension is projected to occur on savanna grassland and 15% on primary forest by 2020, due to the implementation of the EU biofuels mandate (Al-Riffai et al., 2010).

3.5. *Synthesis of land-use change pathways*

The bioenergy-driven land-use changes of the four regions are synthesised into four pathways (Fig. 2).

Pathway 1 involves direct clearing of primary forests, savannas, and native grasslands to make way for bioenergy crop expansion. This pathway was the most common land-use change pathway for developing regions, such as South America, Southeast Asia, and Africa. Pathway 2 involves conversion of cattle pasture resulting indirectly from bioenergy crop expansion, as described for Brazil. The displacement of cattle ranching may lead to indirect deforestation in other locations (IEA Bioenergy, 2009). Pathway 3 represents the conversion of existing cropland to bioenergy crop production, and was primarily documented in the U.S.A. and the EU. This pathway could also trigger iLUCs in other locations through the displacement of existing crops. There is also a risk of indirect deforestation due to the displacement of existing agricultural lands in developing regions with abundant land resources, ideal agro-climatic conditions, and strong development pressures, such as Brazil.

Pathway 4 involves the conversion of marginal, degraded, or abandoned agricultural land to bioenergy crop production, especially for non-food and lignocellulosic crop production. These agricultural lands are not in production, or not suitable for food production (referred to here as 'underutilised agricultural lands'), which include lands under CRP in the U.S.A. and set-aside areas in Europe. The availability and potential use of these agricultural lands have been increasingly recognised as having potential for future bioenergy production in order to minimise various land-use change impacts (Campbell et al., 2008; Field et al., 2008; Wiegmann et al., 2008). However, the sustainability of the use of these lands for bioenergy crop production is uncertain and requires further research (Wicke, 2011).

4. Discussion

The results from our review should be considered in the wider context of global and regional land-use and bioenergy policies.

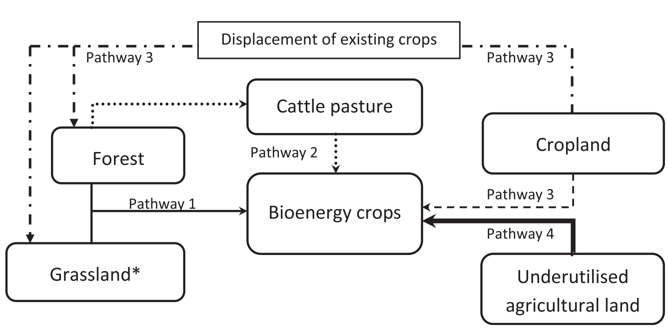


Fig. 2. General pathways for bioenergy-driven land-use change across all regions. *Grassland includes natural grassland, rangeland and savanna.

We identify the following opportunities for more effective policy development in relation to bioenergy crop production, and its expansion.

4.1. *Opportunity 1: give high priority to no and/or less land-using bioenergy feedstock*

Bioenergy produced from waste and residues has substantial environmental advantages over bioenergy produced from dedicated energy crops (Ackom et al., 2010). It can largely avoid the sustainability issues associated with bioenergy-driven land-use changes such as: competition for land with food production; the release of carbon from the soil and biomass; impacts on water, soil, and biodiversity (Campbell et al., 2008; Field et al., 2008; Hill et al., 2006; Wiegmann et al., 2008); and more importantly, the risk of iLUC (Fargione et al., 2008). The consensus view in the literature is that bioenergy policy should continuously give priority to the use of such feedstocks that require no and/or less land resources and their associated technologies, such as lignicellulosic conversion technologies and biofuels from high-yield algae (Fritsche et al., 2010c; Yeh and Witcover, 2010). Costs are still a challenge for commercialisation of lignocellulosic conversion technologies (Sims et al., 2008). Algae is a promising feedstock due to its much higher yield and lower land demands compared to terrestrial crops. However, a significant technical breakthrough is still required to achieve its commercialisation (Singh and Olsen, 2011). Despite these limitations, bioenergy production from no and/or less land-using feedstocks must continue to be given priority over dedicated bioenergy crops in the long term. This can be achieved via various policy instruments such as financial support for research and development of related industries and supply chains.

4.2. *Opportunity 2: develop sustainable land-use options for bioenergy crop production*

A significant portion of future bioenergy demand can be met in the long term by the previously mentioned no and/or less land-using feedstocks and other renewable energy technologies. However, land is still required for dedicated bioenergy crops to meet the short- to mid-term demand for bioenergy. This review emphasises that careful consideration must be given to the nature of the land-use change pathways to ensure that their effects are minimised. Sustainable land-use options for bioenergy crop production may involve two solutions: agricultural land-use intensification; and the use of underutilised agricultural land.

Intensification of production on existing agricultural lands is certainly a solution that will help minimise further agriculture expansion of bioenergy crops. It can be achieved through the introduction of high-yield and land-efficient crops improvements to the productivity of existing crops through the application of appropriate agricultural management practices, maximum use of by-products and co-products, and the introduction of multiple crop rotations (Wicke et al., 2012). For example, increasing grazing density has been proposed in Brazil to minimise deforestation associated with agricultural expansion and the associated carbon emissions (Lapola et al., 2010).

A second solution is the use of underutilised agricultural lands for non-food and lignocellulosic bioenergy crops (Pathway 4 in Fig. 2). However, the sustainability of the use of these lands is controversial and uncertain for several reasons. Firstly, the availability and the potential of these lands may be much smaller than initially estimated (Fritsche et al., 2010c). Secondly, the environmental impacts from the use of these lands still requires further research, as they often require significant inputs of water and nutrients to maintain productivity (Fritsche et al., 2010c; Robertson et al., 2008; Wicke, 2011), and may have high conservation and biodiversity values, particularly if abandoned for long periods (Bowen et al., 2007; Reijnders and Huijbregts, 2009; Robertson et al., 2008). The socio-economic and social outcomes are also questionable. In India, Africa, and other developing regions, marginal land is an important part of the livelihood of smallholder farmers and the rural poor (Matondi et al., 2011; Rajagopal, 2007; Van der Horst and Vermeylen, 2011). For example, livestock production on these lands is important for the rural economy in Africa (Matondi et al., 2011). Thus the socio-economic consequences of their use must be evaluated carefully. The sustainability of using underutilised agricultural land for bioenergy production is an important emerging area of research (Wicke et al., 2012), the results of which will assist policy makers in understanding the potential impacts of its use.

4.3. Opportunity 3: develop agreed international policy mechanisms and instruments for sustainable land-use options for bioenergy crop production

A sustainable land-use policy for bioenergy production must be implemented through effective land-use planning intervention. Bioenergy-driven land-use changes often have been described as capitalist relationships between 'North' and 'South' in the political and social science literature (e.g. Borras et al., 2010). This review confirmed that in developed countries, land-use planning is mostly well regulated, and there has been much less evidence of large-scale conversion of natural vegetation to bioenergy crops. There are also land constraints in many developed countries (e.g. EU), and this has resulted in large-scale bioenergy crop production and investments in countries where agricultural expansion is still possible. The challenges are more acute in developing countries, where political, institutional, and enforcement capabilities are limited, land-use legislation and planning is ineffective, and economic development and private interests often take priority over environmental and sustainable land-use policies.

Effective strategies for avoiding the negative effects of large-scale bioenergy crop expansion are required, including economic mechanisms and institutional improvements through international political action and cooperation, especially in countries where there is a lack of clear environmental and land-use policies, a legal framework, and enforcement capability at the national level. Existing international climate policy, such as the emission accounting system under the Kyoto Protocol (IPCC, 2006) has encouraged developed countries to import bioenergy products from developing countries, triggering the conversion of native vegetation to bioenergy crop production in developing countries (Schubert et al., 2010). However, emerging climate policies have introduced economic mechanisms, such as REDD under the UNFCCC. REDD aims to provide incentives to protect forests with high biodiversity value and high carbon stock, and is expected to influence future land-use policy and planning worldwide. The mechanism is still being developed and REDD has attracted various concerns and criticisms, including the imposition of long-term constraints on land-use in certain areas, because it may affect local communities and cause displacement of deforestation to areas where REDD schemes are not active (Ghazoul et al., 2010). Its effectiveness also has been questioned because of its market-oriented nature (Nantha and Tisdell, 2009) and dependence on various conditions such as additionality, leakage or permanence (Gawel and Ludwig, 2011). However, the REDD mechanism is generally regarded as a positive step towards minimising the negative environmental consequences of future bioenergy-driven land-use change in developing countries (Gibbs et al., 2010; Nepstad et al., 2008).

4.4. Opportunity 4: Strengthen sustainability requirements and certification schemes

Rapid developments are occurring in international markets, requiring agricultural producers to comply with sustainability requirements and certification criteria in order to participate in international commodity markets. The EU's Renewable Energy Directive (RED) has adopted certification criteria for biofuels, which include a prohibition on the use of those biofules produced from biomass grown on land converted from forests, wetlands, or other high-carbon stock areas (e.g. peatland), and highly biodiverse areas (European Union, 2010). Thus biofuels used in the EU have to comply with certification criteria including iLUC. In the U.S.A., there are standards for biofuel sustainability in both the public and private sectors (e.g. Council for Sustainable Biomass Production (CSBP)), and this trend is likely to expand into other international markets. There are also international initiatives towards sustainable crop production across various stakeholders and their voluntarily certification schemes, such as the Roundtable on Sustainable Palm Oil (RSPO), the Round Table on Responsible Soy (RTRS), Bonsucro (Better Sugarcane Initiatives), the Roundtable on Sustainable Biofuels (RSB), and Global Bioenergy Partnership (GBEP). As the application of sustainability certification criteria is in its infancy, there are still significant uncertainties surrounding its effectiveness. The main challenges of certification criteria relate to weak application in emerging markets, implementation time and cost, inconsistency in the definition of terms (e.g. the distinction between primary and secondary forests) (Wilcove and Koh, 2010), and the uncertainty surrounding their ability to ensure compliance in producing countries (Tomei et al., 2010). However, they have the potential to influence not only future land-use policy but also its implementation and enforcement in bioenergy crop producing countries in coming decades. The environmental and social impacts of these certification schemes need to be comprehensively evaluated in the coming years.

Acknowledgement

The author acknowledges the funding support provided by an Australian Postgraduate Award (APA), and the inputs and comments provided by L. Seabrook and M. Wegener at the University of Queensland, and the valuable comments from the Editor and the two anonymous reviewers.

References

Abdullah, S.A., Hezri, A.A., 2008. From forest landscape to agricultural landscape in the developing tropical country of Malaysia: pattern, process, and their significance on policy. Environmental Management 42, 907–917.

Abdullah, S.A., Nakagoshi, N., 2007. Forest fragmentation and its correlation to human land use change in the state of Selangor, peninsular Malaysia. Forest Ecology and Management 241, 39–48.

Ackom, E., Mabee, W., Saddler, J., 2010. Backgrounder: Major Environmental Criteria of Biofuel Sustainability: IEA Bioenergy Task 39 Report T39-PR4.

Al-Riffai, P., Dimaranan, B., Laborde, D., 2010. Global Trade and Environmental Impact Study of the EU Biofuels Mandate: Final Report. International Food Policy Institute (IFPRI), European Commission.

Banse, M., van Meijl, H., Woltjer, G., 2008. The Impact of First and Second Generation Biofuels on Global Agricultural Production, Trade and Land Use. The11th Annual GTAP Conference. Helsinki, Finland. June 12–14, 2008.

Banse, M., van Meijl, H., Tabeau, A., Woltjer, G., Hellmann, F., Verburg, P.H., 2011. Impact of EU biofuel policies on world agricultural production and land use. Biomass and Bioenergy 35, 2385–2390.

Barona, E., Ramankutty, N., Hyman, G., Coomes, O.T., 2010. The role of pasture and soybean in deforestation of the Brazilian Amazon. Environmental Research Letters 5, 9.

Bauen, A., Chudziak, C., Vad, K., Watson, P., 2010. A Casual Descriptive Approach to Modelling the GHG Emissions Associated with the Indirect Land Use Impacts of Biofuels. Final Report. A Study for the UK Department for Transport. E4tech, London.

Bellamy, P.E., Croxton, P.J., Heard, M.S., Hinsley, S.A., Hulmes, L., Hulmes, S., Nuttall, P., Pywell, R.F., Rothery, P., 2009. The impact of growing miscanthus for biomass on farmland bird populations. Biomass and Bioenergy 33, 191–199.

Berndes, G., Bird, N., Cowie, A., 2010. Bioenergy, Land Use Change and Climate Change Mitigation. IEA Bioenergy.

Biodiesel Magazine, 2010. Global Biodiesel Production and Market Report, Biodiesel Magazine. Available from: http://www.biodieselmagazine.com/articles/4447/global-biodiesel-production-and-market-report. Accessed 16 November 2010.

Biopact, 2006. Palm Biofuel Survives Low Crude Oil Prices – Official. 2010. Available from: http://news.mongabay.com/bioenergy/2006/09/palm-biofuel-survives-low-crude-oil.html. Accessed 29 December 2010.

Blanco Fonseca, M., Burrell, A., Gay, H., Henseler, M., Kavallari, A., M'Barek, R., Domínguez, I.P., Tonini, A., 2010. Impacts of the EU Biofuel Target on Agricultural Markets and Land Use: A Comparative Modelling Assessment: Final Report (June 2010). European Commission, Joint Research Centre, Institute for Prospective Technological Studies, Luxembourg.

Borras, S.M., McMichael, P., Scoones, I., 2010. The politics of biofuels, land and agrarian change: editors' introduction. Journal of Peasant Studies 37, 575–592.

Bowen, M.E., McAlpine, C.A., House, A.P.N., Smith, G.C., 2007. Regrowth forests on abandoned agricultural land: a review of their habitat values for recovering forest fauna. Biological Conservation 140, 273–296.

Brannstrom, C., Jepson, W., Filippi, A.M., Redo, D., Xu, Z., Ganesh, S., 2008. Land change in the Brazilian Savanna (Cerrado), 1986–2002: comparative analysis and implications for land-use policy. Land Use Policy 25, 579–595.

California Air Resource Board, 2012. Low Carbon Fuel Standard Program. Available from: http://www.arb.ca.gov/fuels/lcfs/lcfs.htm. Accessed 10 January 2012.

Campbell, J.E., Lobell, D.B., Genova, R.C., Field, C.B., 2008. The global potential of bioenergy on abandoned agriculture lands. Environmental Science & Technology 42, 5791–5794.

Danielsen, F., Beukema, H., Burgess, N.D., Parish, F., Bruhl, C.A., Donald, P.F., Murdiyarso, D., Phalan, B., Reijnders, L., Struebig, M., Fitzherbert, E.B., 2009. Biofuel plantations on forested lands: double jeopardy for biodiversity and climate. Conservation Biology 23, 348–358.

de Nie, D., Sayer, J., McCormick, N., 2009. Indirect Effects of Bioenergy: Effects on Landscapes and Livelihoods, Bioenergy, the Impact of Indirect Land Use Change: Summary and Conclusions from the IEA Bioenergy ExCo63 Workshop. International Union for Conservation of Nature (IUCN), Rotterdam, the Netherlands.

de Vries, S.C., van de Ven, G.W.J., van Ittersum, M.K., Giller, K.E., 2010. Resource use efficiency and environmental performance of nine major biofuel crops, processed by first-generation conversion techniques. Biomass and Bioenergy 34, 588–601.

EBB (European Biodiesel Board), 2010. Statistics: The EU Biodiesel Industry. Available from: http://www.ebb-eu.org/stats.php#. Accessed 16 November 2010.

ePure, 2010. Production Data. 2011. Available from: http://www.epure.org/statistics/info/Productiondata. Accessed January 5 2011.

European Commission, 2003. Directive 2003/30/EC on the Promotion of the Use of Biofuels or Other Renewable Fuels for Transport. The European Parliament and the Council of the European Union.

European Commission, 2009. Directive 2009/28/EC on the Promotion of the Use of Energy from Renewable Sources and Amending and Subsequently Repealing Directive 2001/77/EC and 2003/30/EC. The European Parliament and the Council of the European Union.

European Union, 2010. Communication from the Commission on the Practical Implementation of the EU Biofuels and Bioliquids Sustainability Scheme and on Counting Rules for Biofuels (2010/C 160/02).

FAOSTAT, 2011. FAOSTAT Database. Food and Agriculture Organization of the United Nations (FAO). Available from: http://faostat.fao.org/.

Fargione, J., Hill, J., Tilman, D., Polasky, S., Hawthorne, P., 2008. Land clearing and the biofuel carbon debt. Science 319, 1235–1238.

Fearnside, P.M., 2001. Soybean cultivation as a threat to the environment in Brazil. Environmental Conservation 28, 23–38.

Fearnside, P.M., 2005. Deforestation in Brazilian Amazonia: history, rates, and consequences. Conservation Biology 19, 680–688.

Fearnside, P.M., Righi, C.A., Graca, P., Keizer, E.W.H., Cerri, C.C., Nogueira, E.M., Barbosa, R.I., 2009. Biomass and greenhouse-gas emissions from land-use change in Brazil's Amazonian "arc of deforestation": the states of Mato Grosso and Rondonia. Forest Ecology and Management 258, 1968–1978.

Field, C.B., Campbell, J.E., Lobell, D.B., 2008. Biomass energy: the scale of the potential resource. Trends in Ecology & Evolution 23, 65–72.

Fiorese, G., Guariso, G., 2010. A GIS-based approach to evaluate biomass potential from energy crops at regional scale. Environmental Modelling & Software 25, 702–711.

Fischer, G., Prieler, S., van Velthuizen, H., Berndes, G., Faaij, A., Londo, M., de Wit, M., 2010. Biofuel production potentials in Europe: sustainable use of cultivated land and pastures, Part II: land use scenarios. Biomass & Bioenergy 34, 173–187.

Fritsche, U.R., Hennenberg, K., Hünecke, K., 2010a. The "iLUC Factor" as a Means to Hedge Risks of GHG Emissions from Indirect Land Use Change: Working Paper. Öko-Institut, Darmstadt.

Fritsche, U.R., Hennenberg, K.J., Hermann, A., Hünecke, K., Herrera, R., Fehrenbach, H., Roth, E., Hennecke, A., Giegrich, J., 2010b. Development of Strategies and Sustainability Standards for the Certification of Biomass for International Trade: Summarizing Final Report. German Federal Environment Agency (Umweltbundesamt), Dessau-Roßlau.

Fritsche, U.R., Sims, R.E.H., Monti, A., 2010c. Direct and indirect land-use competition issues for energy crops and their sustainable production: an overview. Biofuels, Bioproducts and Biorefining 4, 692–704.

Gawel, E., Ludwig, G., 2011. The iLUC dilemma: how to deal with indirect land use changes when governing energy crops? Land Use Policy 28, 846–856.

Ghazoul, J., Butler, R.A., Mateo-Vega, J., Koh, L.P., 2010. REDD: a reckoning of environment and development implications. Trends in Ecology & Evolution 25, 396–402.

Gibbs, H.K., Ruesch, A.S., Achard, F., Clayton, M.K., Holmgren, P., Ramankutty, N., Foley, J.A., 2010. Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. Proceedings of the National Academy of Sciences of the United States of America 107, 16732–16737.

Goldemberg, J., Coelho, S.T., Guardabassi, P., 2008. The sustainability of ethanol production from sugarcane. Energy Policy 36, 2086–2097.

Graham, R.L., Downing, M., Walsh, M.E., 1996. A framework to assess regional environmental impacts of dedicated energy crop production. Environmental Management 20, 475–485.

Grau, H.R., Gasparri, N.I., Aide, T.M., 2008. Balancing food production and nature conservation in the Neotropical dry forests of northern Argentina. Global Change Biology 14, 985–997.

Hammerschlag, R., 2006. Ethanol's energy return on investment: a survey of the literature 1990–present. Environmental Science & Technology 40, 1744–1750.

Hartman, J.C., Nippert, J.B., Orozco, R.A., Springer, C.J., 2011. Potential ecological impacts of switchgrass (*Panicum virgatum* L.) biofuel cultivation in the Central Great Plains, USA. Biomass and Bioenergy 35, 3415–3421.

Hiederer, R., Ramos, F., Capitani, C., Koeble, R., Blujdea, V., Gomez, O., Mulligan, D., Marelli, L., 2010. Biofuels: A New Methodology to Estimate GHG Emissions from Global Land Use Change, a Methodology Involving Spatial Allocation of Agricultural Land Demand and Estimation of CO2 and N2O Emissions. European Commission, Joint Research Centre.

Hill, J., Nelson, E., Tilman, D., Polasky, S., Tiffany, D., 2006. Environmental, economic, and energetic costs and benefits of biodiesel and ethanol biofuels. Proceedings of the National Academy of Sciences of the United States of America 103, 11206–11210.

Hoh, R., 2010. Malaysia Biofuels Annual Report 2010. USDA Foreign Agriculture Service GAIN Report. U.S. Department of Agriculture.

IEA, 2011. Technology Roadmap: Biofuels for Transport. International Energy Agency, Paris.

IEA (International Energy Agency), 2004. Biofuels for Transport: An International Percepective. International Energy Agency, Paris.

IEA Bioenergy, 2009. Bioenergy: The Impact of Indirect Land Use Change: Summary and Conclusions from the IEA Bioenergy ExCo63 Workshop, Rotterdam, the Netherlands.

INPE, 2012. Taxas anuais do desmatamento – 1988 até 2011 (Annual deforestation rate – 1988 to 2011). PRODES database. Instituto Nacional de Pesquisa Especiais (INPE). Available from: http://www.obt.inpe.br/prodes/prodes_1988_2011.htm.

IPCC (Intergovernmental Panel on Climate Change), 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories vol. 4. Agriculture, Forestry and Other Land Use Intergovernmental Panel on Climate Change (IPCC), Hayama, Kanagawa.

Koh, L.P., Wilcove, D.S., 2008. Is oil palm agriculture really destroying tropical biodiversity? Conservation Letters 1, 60–64.

Koh, L.P., Miettinen, J., Liew, S.C., Ghazoul, J., 2011. Remotely sensed evidence of tropical peatland conversion to oil palm. Proceedings of the National Academy of Sciences of the United States of America 108, 5127–5132.

Krautgartner, R., Henard, M.-C., Lieberz, S., Boshnakova, M., Flach, B., Wilson, J., Wideback, A., Bettini, O., Guerrero, M., Bendz, K., 2011. EU-27, Oilseeds and Products Annual, Modest Rebound in EU-27 Oilseeds Production. USDA Foreign Agricultural Service.

Laborde, D., 2011. Assessing the Land Use Change Consequences of European Biofuels Policies. International Food Policy Research Institute (IFPRI).

Lapola, D.M., Schaldach, R., Alcamo, J., Bondeau, A., Koch, J., Koelking, C., Priess, J.A., 2010. Indirect land-use changes can overcome carbon savings from biofuels in Brazil. Proceedings of the National Academy of Sciences of the United States of America 107, 3388–3393.

Le, P.V.V., Kumar, P., Drewry, D.T., 2011. Implications for the hydrologic cycle under climate change due to the expansion of bioenergy crops in the Midwestern United States. Proceedings of the National Academy of Sciences of the United States of America 108, 15085–15090.

Loarie, S.R., Lobell, D.B., Asner, G.P., Mu, Q., Field, C.B., 2011. Direct Impacts on Local Climate of Sugar-cane Expansion in Brazil, Nature Climate Change. Macmillan Publishers Limited (published online).

Love, B.J., Nejadhashemi, A.P., 2011. Water quality impact assessment of large-scale biofuel crops expansion in agricultural regions of Michigan. Biomass and Bioenergy 35, 2200–2216.

Martin, F., 2011. Agroecological zoning for biofuels: the Brazilian Experience. IEA Bioenergy Workshop: Quantifying and Managing Land Use Effects of Bioenergy, 19–20 September 2011, Campinas, Brazil.

Martinelli, L.A., Filoso, S., 2008. Expansion of sugarcane ethanol production in Brazil: environmental and social challenges. Ecological Applications 18, 885–898.

Matondi, P.B., Havnevik, K., Beyene, A., 2011. Biofuels, Land Grabbing and Food Security in Africa Zed Books, (London, New York).

McAlpine, C.A., Etter, A., Fearnside, P.M., Seabrook, L., Laurance, W.F., 2009. Increasing world consumption of beef as a driver of regional and global change: a call for policy action based on evidence from Queensland (Australia), Colombia and Brazil. Global Environmental Change 19, 21–33.
Miller, S.A., Landis, A.E., Theis, T.L., August 1, 2007. Environmental trade-offs of biobased production. Environmental Science & Technology, 5176–5182.
Mitchell, D., 2008. Policy Research Working Paper 4682: A Note on Rising Food Prices. The World Bank.
Morton, D.C., DeFries, R.S., Shimabukuro, Y.E., Anderson, L.O., Arai, E., Espirito-Santo, F.d.B., Freitas, R., Morisette, J., 2006. Cropland expansion changes deforestation dynamics in the southern Brazilian Amazon. Proceedings of the National Academy of Sciences of the United States of America 103.
Mueller, C.C., 2003. Expansion and Modernization of Agriculture in the Cerrado: The Case of Soybeans in Brazil's Center-West. Department of Economics, University of Brazilia.
Nantha, H.S., Tisdell, C., 2009. The orangutan-oil palm conflict: economic constraints and opportunities for conservation. Biodiversity and Conservation 18, 487–502.
Naylor, R.L., Liska, A.J., Burke, M.B., Falcon, W.P., Gaskell, J.C., Rozelle, S.D., Cassman, K.G., 2007. The ripple effect: biofuels, food security, and the environment. Environment 49, 30–43.
Nepstad, D.C., Stickler, C.M., Soares, B., Merry, F., 2008. Interactions among Amazon land use, forests and climate: prospects for a near-term forest tipping point. Philosophical Transactions of the Royal Society B-Biological Sciences 363, 1737–1746.
Ng, T.L., Eheart, J.W., Cai, X., Miguez, F., 2010. Modeling miscanthus in the Soil and Water Assessment Tool (SWAT) to simulate its water quality effects as a bioenergy crop. Environmental Science and Technology 44, 7138–7144.
Pacheco, P., 2009. Agrarian reform in the Brazilian Amazon: its implications for land distribution and deforestation. World Development 37, 1337–1347.
Panichelli, L., Dauriat, A., Gnansounou, E., 2009. Life cycle assessment of soybean-based biodiesel in Argentina for export. International Journal of Life Cycle Assessment 14, 144–159.
Payne, W.A., 2010. Are biofuels antithetic to long-term sustainability of soil and water resources? (Chapter 1). Advances in Agronomy 105, 1–46.
Perlack, R.D., Wright, L.L., Turhollow, A.F., Graham, R.L., Stokes, B.J., Erbach, D.C., 2005. Biomass as Feedstock for a Bioenergy and Bioproducts Industry: The Technical Feasibility of a Billion-ton Annual Supply. U.S. Department of Energy.
Pimentel, D., Patzek, T.W., 2005. Ethanol production using corn, switchgrass, and wood; biodiesel production using soybean and sunflower. Natural Resources Research 14, 65–76.
Pineiro, G., Jobbagy, E.G., Baker, J., Murray, B.C., Jackson, R.B., 2009. Set-asides can be better climate investment than corn ethanol. Ecological Applications 19, 277–282.
Pousa, G.P.A.G., Santos, A.L.F., Suarez, P.A.Z., 2007. History and policy of biodiesel in Brazil. Energy Policy 35, 5393–5398.
Powlson, D.S., Riche, A.B., Shield, I., 2005. Biofuels and other approaches for decreasing fossil fuel emissions from agriculture. Annals of Applied Biology 146, 193–201.
Rajagopal, D., 29-31 January, 2007. Rethinking Current Strategies for Biofuel Production in India. Linkages between Energy and Water Management for Agriculture in Developing Countries. ICRISAT campus, Hyderabad, India.
REDD in Indonesia, 2010. Indonesia Moves on Forest Clearing Moratorium. Available from: http://redd-indonesia.org/en/news-room/detail-news/read/indonesia-moves-on-forest-clearing-moratorium/. Accessed 8 November 2010.
REDD (Reducing Emissions from Deforestation and Forest Degradation), 2010. REDD + Partnership. Oslo Forest Climate Conference 2010 (May 2010), Oslo, Norway.
Reijnders, L., 2009. Acute view transport biofuels: can they help limiting climate change without an upward impact on food prices?Journal of Consumer Protection and Food Safety Journal Fur Verbraucherschutz Und Lebensmittelsicherheit 4, 75–78.
Reijnders, L., Huijbregts, M., 2009. Biofuels for Road Transport – A Seed to Wheel Perspective. Springer, London.
Renouf, M.A., Wegener, J.K., Nielsen, L.K., 2008. An environmental life cycle assessment comparing Australian sugarcane with US corn and UK sugar beet as producers of sugars for fermentation. Biomass and Bioenergy 32, 1144–1155.
RFA (Renewable Fuels Association), 2010. Statistics 2010. Available from: http://www.ethanolrfa.org/pages/statistics. Accessed 21 October 2010.
Roberts, M.G., Male, T.D., Toombs, T.P., 2007. Potential Impacts of Biofuels Expansion on Natural Resources: A Case Study of the Ogallala Aquifer Region. Environmental Defence.).
Robertson, G.P., Dale, V.H., Doering, O.C., Hamburg, S.P., Melillo, J.M., Wander, M.M., Parton, W.J., Adler, P.R., Barney, J.N., Cruse, R.M., Duke, C.S., Fearnside, P.M., Follett, R.F., Gibbs, H.K., Goldemberg, J., Mladenoff, D.J., Ojima, D., Palmer, M.W., Sharpley, A., Wallace, L., Weathers, K.C., Wiens, J.A., Wilhelm, W.W., 2008. Agriculture – sustainable biofuels Redux. Science 322, 49–50.
Rounsevell, M.D.A., Reay, D.S., 2009. Land use and climate change in the UK. Land Use Policy 26, S160–S169.
Rounsevell, M.D.A., Reginster, I., Araujo, M.B., Carter, T.R., Dendoncker, N., Ewert, F., House, J.I., Kankaanpaa, S., Leemans, R., Metzger, M.J., Schmit, C., Smith, P., Tuck, G., 2006. A coherent set of future land use change scenarios for Europe. Agriculture Ecosystems & Environment 114, 57–68.
Rowe, R.L., Street, N.R., Taylor, G., 2009. Identifying potential environmental impacts of large-scale deployment of dedicated bioenergy crops in the UK. Renewable & Sustainable Energy Reviews 13, 271–290.
Rowe, R.L., Hanley, M.E., Goulson, D., Clarke, D.J., Doncaster, C.P., Taylor, G., 2011. Potential benefits of commercial willow Short Rotation Coppice (SRC) for farm-scale plant and invertebrate communities in the agri-environment. Biomass and Bioenergy 35, 325–336.
Sawyer, D., 2008. Climate change, biofuels and eco-social impacts in the Brazilian Amazon and Cerrado. Philosophical Transactions of the Royal Society B-Biological Sciences 363, 1747–1752.
Schilling, K.E., Jha, M.K., Zhang, Y.K., Gassman, P.W., Wolter, C.F., 2008. Impact of land use and land cover change on the water balance of a large agricultural watershed: historical effects and future directions. Water Resources Research 44.
Schubert, R., Schellnhuber, H.J., Buchmann, N., Epiney, A., Grießhammer, R., Kulessa, M., Messner, D., Rahmstorf, S., Schmid, J., 2010. Future Bioenergy and Sustainable Land Use. Earthscan, London ; Sterling, VA.
Searchinger, T., Heimlich, R., Houghton, R.A., Dong, F.X., Elobeid, A., Fabiosa, J., Tokgoz, S., Hayes, D., Yu, T.H., 2008. Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land-use change. Science 319, 1238–1240.
Secchi, S., Kurkalova, L., Gassman, P.W., Hart, C., 2011. Land use change in a biofuels hotspot: the case of Iowa, USA. Biomass and Bioenergy 35, 2391–2400.
Sims, R., Taylor, M., Saddler, J., Mabee, W., 2008. From 1st- to 2nd-generation Biofuel Technologies: An Overview of Current Industry and RD&D Activities. International Energy Agency and OECD, Paris.
Singh, A., Olsen, S.I., 2011. A critical review of biochemical conversion, sustainability and life cycle assessment of algal biofuels. Applied Energy 88, 3548–3555.
Sparovek, G., Barretto, A., Berndes, G., Martins, S., Maule, R., 2009. Environmental, land-use and economic implications of Brazilian sugarcane expansion 1996–2006. Mitigation and Adaptation Strategies for Global Change 14, 285–298.
Sparovek, G., Berndes, G., Klug, I.L.F., Barretto, A., 2010. Brazilian agriculture and environmental legislation: status and future challenges. Environmental Science & Technology 44, 6046–6053.
State of California Office of the Governor, 2007. Executive Order S-01–07. State of California Office of the Governor.
The Soybean and Corn Advisor, 2010. 7% of Brazil's Soybeans Used for Biodiesel Production (June 7, 2010). Available from: http://www.soybeansandcorn.com/news/Jun7_10-7-of-Brazils-Soybeans-Used-for-Biodiesel-Production. Accessed 28 June 2011.
Thoenes, P., 2006. Biofuels and commodity markets: palm oil focus. The Impact of biofuels on commodity markets conference. Brussels. 24–25 October 2006.
Tokgoz, S., Elobeid, A., Fabiosa, J., Hayes, D.J., Babcock, B.A., Yu, T.-H., Dong, F., Hart, C.E., Beghin, J.C., 2007. Emerging Biofuels: Outlook of Effects on U.S. Grain, Oilseed, and Livestock Markets. Center for Agricultural and Rural Development (CARD).
Tomei, J., Semino, S., Paul, H., Joensen, L., Monti, M., Jelsoe, E., 2010. Soy production and certification: the case of Argentinean soy-based biodiesel. Mitigation and Adaptation Strategies for Global Change 15, 371–394.
Tyner, W.E., Taheripour, F., Zhuang, Q., Birur, D., Baldos, U., 2010. Land Use Changes and Consequent CO2 Emissions Due to US Corn Ethanol Production: A Comprehensive Analysis: Final Report. Department of Agricultural Economics, Purdue University.
Ugarte, D.G.D., He, L.X., Jensen, K.L., English, B.C., 2010. Expanded ethanol production: implications for agriculture, water demand, and water quality. Biomass & Bioenergy 34, 1586–1596.
UNICA, 2010b. Brazilian Ethanol Production. Available from: http://english.unica.com.br/dadosCotacao/estatistica/. Accessed 28 October 2010.
UNICA (Brazilian Sugarcane Industry Association), 2010a. Brazilian Ethanol Exports. Available from: http://english.unica.com.br/dadosCotacao/estatistica/. Accessed 21 October 2010.
USDA, 2010. Corn for Grain 2009 – Production by Country and Location of Ethanol Plants. U.S. Department of Agriculture.
U.S. EPA, 2009. EPA Proposes New Regulations for the National Renewable Fuel Standard Program for 2010 and beyond (May 2009). U.S. Environmental Protection Agency. Available from: http://www.epa.gov/OMS/renewablefuels/420f09023.htm. Accessed September 17 2010.
U.S. EPA, 2010. Renewable Fuel Standard (RFS). U.S. Environmental Protection Agency. Available from: http://www.epa.gov/otaq/fuels/renewablefuels/index.htm. Accessed January 3 2011.
Van der Horst, D., Vermeylen, S., 2011. Spatial scale and social impacts of biofuel production. Biomass and Bioenergy 35, 2435–2443.
Vanloocke, A., Bernacchi, C.J., Twine, T.E., 2010. The impacts of *Miscanthus x giganteus* production on the Midwest US hydrologic cycle. Global Change Biology Bioenergy 2, 180–191.
von Blottnitz, H., Curran, M.A., 2007. A review of assessments conducted on bio-ethanol as a transportation fuel from a net energy, greenhouse gas, and environmental life cycle perspective. Journal of Cleaner Production 15, 607–619.
Walsh, M.E., De La Torre Ugarte, D.G., Shapouri, H., Slinsky, S.P., 2003. Bioenergy crop production in the United States: potential quantities, land use changes, and economic impacts on the agricultural sector. Environmental and Resource Economics 24, 313–333.
Wicke, B., 2011. Bioenergy Production on Degraded and Marginal Land: Assessing Its Potential, Economic Performance, and Environmental Impacts for Different

Settings and Geographical Scales. PhD. thesis, Faculty of Science, Copernics Institute, Group Science, Technology and Society, Utrecht University, Utrecht.
Wicke, B., Verweij, P., van Meijl, H., van Vuuren, D.P., Faaij, A.P.C., 2012. Indirect land use change: review of existing models and strategies for mitigation. Biofuels 3, 87–100.
Wiegmann, K., Hennenberg, K.J., Fritsche, U.R., 2008. Degraded land and sustainable bioenergy feedstock production: Issue paper. Joint International Workshop on High Nature Value Criteria and Potential for Sustainable Use of Degraded Lands Paris, June 30–July 1, 2008.
Wilcove, D.S., Koh, L.P., 2010. Addressing the threats to biodiversity from oil-palm agriculture. Biodiversity and Conservation 19, 999–1007.
Worldwatch Institute, 2007. Biofuels for Transport: Global Potential and Implications for Sustainable Energy and Agriculture. Earthscan, Sterling, VA.
Wu, Y., Liu, S., 2012. Impacts of biofuels production alternatives on water quantity and quality in the Iowa River Basin. Biomass and Bioenergy 36, 182–191.
Yeh, S., Witcover, J., 2010. Indirect Land-Use Change from Biofuels: Recent Developments in Modeling and Policy Landscapes. International Food and Agricultural Trade Policy Council.

Appendix 2

Table A2.1 Summary of changes in land-use and land cover from the literature reviewed.

Past or current land-use change

Type of land-use change	Study year	Crop	Previous land-use	Location/Country	Purpose of the study	Reference
displacement of other crops/agricultural intensification	-	corn	cropland for soybean & other crops	USA	examining the consequences of future land-use and land cover change from biofuel expansion on the water balance.	Schilling et al. (2008) [1]
displacement of other crops/agricultural intensification	2002-07	corn	cropland for soybean	USA	examining the factors behind the rising global food price since 2002.	Mitchell (2008) [2]
displacement of other crops/agricultural intensification	2006-08	corn, soybean, wheat	grassland/pasture	the Prairie States, the Southeast, USA	drawing forward-looking insights into the likely willingness of US farmers to grow dedicated cellulosic bioenergy crops on marginal lands.	Swinton et al. (2011) [3]
deforestation/cropland expansion	various (1700-)	cropland (e.g. oil palm, sugarcane)	forest	tropical regions	reviewing the major patterns of land-use and land cover change in the tropical region in relation to soil fertility and nutrient management.	Hartemink et al. (2008) [4]
cropland expansion	1981-2000	cropland	various	Bangladesh along the Indus Valley, parts of the Middle East and Central Asia, the region of the Great Lakes of eastern Africa, the south border of the Amazon basin, the Great Plains in the US	producing the current synthesis of documented change cover the period 1981-2000.	Lepers et al. (2005) [5]
deforestation/cropland expansion	1981-2000	cropland	forest	Southeast Asia		

Type of land-use change	Study year	Crop	Previous land-use	Location/Country	Purpose of the study	Reference
deforestation/cropland expansion	1990-2007	oil palm	forest	Borneo, Indonesia	assessing strategies for conservation of orangutan habitat.	Nantha and Tisdell (2009) [6]
deforestation/cropland expansion	1990-2005	oil palm	forest (primary, secondary), plantation	Malaysia & Indonesia	discussing the existing strategies for the oil palm crisis in Southeast Asia.	Wilcove and Koh (2010) [7]
deforestation/cropland expansion/displacement of other crops	1990-2005	oil palm	forest (secondary) and plantation (55-59%), pre-existing cropland (41-45%)	Malaysia & Indonesia	presenting a framework for assessing the impact associated with land conversion to oil palm agriculture on biodiversity.	Koh and Wilcove (2008) [8]
deforestation/cropland expansion	1966, 1981, 1995	oil palm	wetland forest, marchland, inland forest & agricultural land	Selangor, Malaysia	developing a single forest fragmentation index.	Abdullah and Nakagoshi (2007) [9]
displacement of other crops	1966-95	oil palm	forest & rubber plantation	Peninsular of Malaysia, Malaysia	examining the links between development policies and changes in land-use and land cover, and its implication for future development policies.	Abdullah and Hezri (2008) [10]
displacement of other crops	2001-07	rapeseed, sunflower	cropland for wheat	Argentina, Canada, EU, Russia, Ukraine (major wheat exporters)	examining the factors behind the rising global food price since 2002.	Mitchell (2008) [2]
deforestation/cropland expansion	-	soybean	forest (native)	The Yungas, Algentina	investigating whether the social and environmental impacts of soybean production can be mitigated by the Round Table on Responsible Soy (RTRS).	Tomei et al. (2010) [11]
deforestation/cropland expansion	2000-05	soybean	forest	Four Provinces (Salta, Chaco, Tucumán, Santiago del Estero), Argentina	determining the environmental impact of vegetable oil methyl ester (VOME) production from soybean culture in Argentina for export.	Panichelli et al. (2009) [12]
displacement of other crops	2000-05	soybean	cropland (e.g. corn, wheat, sunflower & sorghum)	Northern Provinces, Argentina		

Type of land-use change	Study year	Crop	Previous land-use	Location/Country	Purpose of the study	Reference
pasture abandonment/ cropland expansion	2000-05	soybean	grassland (pasture)	Northern Provinces, Argentina		Panichelli et al. (2009) [12]
deforestation/grassland conversion/cropland expansion	early 1970s 2002	soybean	various (forest, grazing pasture)	Chaco, Argentina	evaluating the efficiency of different land-use practices both in terms of food production and nature conservation.	Grau et al. (2008) [13]
deforestation/cropland expansion	2005	soybean	forest	Amazon, Brazil	assessing the effects of Amazon deforestation on the regional climate.	Sampaio et al. (2007) [14]
deforestation/ crop expansion/ indirect land-use change	2001-04	soybean	primary forest	Brazil	estimating both direct and indirect emissions in the biofuel life cycle.	Liska and Perrin (2009) [15]
deforestation/cropland expansion	2001-04	soybean	forest	Amazon, Brazil	providing satellite-based evidence for the contribution of cropland and pasture to deforestation in Brazilian Amazon.	Morton et al. (2006) [16]
pasture abandonment/ cropland expansion	2001-04	soybean	cattle pasture	Mato Grosso, Brazil		
deforestation/grassland conversion/cropland expansion	2001-04	soybean	savanna (Cerrado)			
deforestation/cropland expansion	2001-04	soybean	forest	Legal Amazon (9 states), Brazil		
pasture abandonment/ cropland expansion	late 1990s to 2006	soybean	cattle pasture	Amazon, Brazil	reviewing trends in Amazon economic, ecological and climatic processes.	Nepstad et al. (2008) [17]
deforestation/cropland expansion	late 1990s to 2006	soybean	forest	Amazon, Brazil		
pasture abandonment/ cropland expansion	2000-06	soybean	cattle pasture	Mato Grosso, Brazil	examining the spatial patterns between deforestation and changes in pasture and soybean areas.	Barona et al. (2010) [18]
pasture abandonment/ cropland expansion	2000-06	soybean	cattle pasture/cleared land	Rondônia, Brazil		
pasture abandonment/ cropland expansion	1980 onwards	soybean	savanna (Cerrado)	Cerrado, Brazil	examining the expansion and the recent modernization of agriculture in Brazil's Cerrado region.	Mueller (2003) [19]

Type of land-use change	Study year	Crop	Previous land-use	Location/Country	Purpose of the study	Reference
grassland conversion/cropland expansion/pasture expansion	1986-2002	soybean & pasture	savanna (Cerrado)	western Bahia, Brazil	comparing land-use and land cover changes in the Brazilian Cerrado emphasising the spatial patterns of its fragmentation.	Brannstrom et al. (2007) [20]
grassland conversion/cropland expansion/pasture expansion	1986-2002	soybean & pasture	savanna (Cerrado)	eastern Mato Grosso, Brazil	comparing land-use and land cover changes in the Brazilian Cerrado emphasising the spatial patterns of its fragmentation.	Brannstrom et al. (2007) [20]
deforestation/cropland expansion/pasture expansion	1976-2007	soybean & pasture	forest	Amazon (Mato Grosso & Rondônia), Brazil	estimating GHG emissions due to deforestation.	Fearnside et al. (2009) [21]
grassland conversion/cropland expansion/pasture expansion	1976-2007	soybean & pasture	savanna (Cerrado)	Cerrado (Mato Grosso & Rondônia), Brazil		
grassland conversion/cropland expansion/pasture expansion	1990 onwards	soybean & pasture	savanna (Cerrado)	Cerrado, Brazil	examining interactions among climate change, political-economic interventions and technical progresses focusing on the impacts of biofuels.	Sawyer (2008) [22]
deforestation/cropland expansion/pasture expansion	1990 onwards	soybean & pasture	forest	Amazon, Brazil		
deforestation/grassland conversion/cropland expansion	1990 onwards	soybean, sugarcane	savanna (Cerrado)	Cerrado, Brazil		
deforestation/cropland expansion/pasture expansion	1990 onwards	soybean & pasture	forest	Amazon, Brazil	examining the spatial patterns between deforestation and changes in pasture and soybean areas.	Barona et al. (2010) [18]
deforestation/crop expansion	2005-08	non-sugarcane crops or cattle pasture	savanna (Cerrado)	Cerrado, Brazil	quantifying the direct climate effects of sugarcane expansion in the Brazilian Cerrado.	Loarie et al (2011) [23]

Type of land-use change	Study year	Crop	Previous land-use	Location/Country	Purpose of the study	Reference
pasture abandonment/ cropland expansion	2007 onwards	soybean, sugarcane	cattle pasture	Amazon, Brazil	examining interactions among climate change, political-economic interventions and technical progresses focusing on the impacts of biofuels.	Sawyer (2007) [22]
deforestation/cropland expansion	by 2003	soybean, sugarcane	forest	Amazon, Brazil	providing better understandings of the social, economic and political aspects related to the deforestation.	Feanside (2005) [24]
pasture abandonment/ cropland expansion	1990-2005	sugarcane	cattle pasture	western region of São Paulo, Brazil	discussing environmental and social issues linked to the expansion of sugarcane for ethanol and providing recommendations on the policy.	Martinelli and Filoso (2008) [25]
cropland expansion	late 1990s to 2006	sugarcane	--	Amazon, Brazil	reviewing trends in Amazon economic, ecological and climatic processes.	Nepstad et al. (2008) [17]
pasture abandonment/ cropland expansion	2005	sugarcane	cattle pasture	São Paulo, Brazil	discussing the sustainability aspects of ethanol production.	Goldemberg et al. (2008) [26]
pasture abandonment/ cropland expansion	2002-08	sugarcane	cattle pasture	Cerrado, Brazil	quantifying the direct climate effects of sugarcane expansion in the Brazilian Cerrado.	Loarie et al (2011) [23]
pasture abandonment/ cropland expansion	1996-2006	sugarcane	cattle pasture	São Paulo and neighbouring states, Brazil	assessing the expansion of sugarcane in Brazil during the period 1996-2006.	Sparovek et al. (2009) [27]
deforestation/cropland expansion	-	sugarcane	forest	Hawaii, USA	investigating the effect of land-use changes on soil carbon.	Osher et al (2003) [28]
pasture abandonment/ cropland expansion	-	sugarcane	cattle pasture			
deforestation/pasture expansion	-	cattle pasture	forest	La Pampa province, Argentina	analysing the socio-economic and environmental performance of two large-scale bioenergy production scenarios.	van Dam et al. (2009) [29]
deforestation/pasture expansion	-	cattle pasture	forest/woodland	central and southern Queensland, Australia	presenting the severity of environmental impacts of rapid increase in the global beef consumption.	McAlpine et al. (2009) [30]

Type of land-use change	Study year	Crop	Previous land-use	Location/Country	Purpose of the study	Reference
deforestation/pasture expansion	2005	cattle pasture	forest	Amazon, Brazil	assessing the effects of Amazon deforestation on the regional climate.	Sampaio et al. (2007) [14]
deforestation/pasture expansion	late 1990s to 2006	cattle pasture	forest	North region (including Amazon states), Brazil	reviewing trends in Amazon economic, ecological and climatic processes.	Nepstad et al. (2008) [17]
deforestation/pasture expansion	2000-06	cattle pasture	forest	Mato Grosso, Brazil	examining the spatial patterns between deforestation and changes in pasture and soybean areas.	Barona et al. (2010) [18]
deforestation/pasture expansion	1978-2007	cattle pasture	forest & savanna	Amazon, Brazil	presenting the severity of environmental impacts of rapid increase in the global beef consumption.	McAlpine et al. (2009) [30]
deforestation/pasture expansion	since 1990	cattle pasture	forest & savanna	lowlands of Amazon and Pacific regions, the foothills of the Andes, Columbia		

Projected land-use change (future scenario/projection)

Type of land-use change	Projected year	Crop	Previous land-use	Location/Country	Purpose of the study	Reference
displacement of other crops/pasture abandonment/cropland expansion	2080	bioenergy crops	pasture & cropland	EU15, Norway, Switzerland	synthesising the methodological aspects of the development of future land-use change scenarios and the key results.	Rounsevell et al. (2006) [31]
displacement of other crops/pasture abandonment/cropland expansion	-	bioenergy crops	surplus or marginal agricultural lands	UK	reviewing the past, present and potential future changes in land-use change in the UK and exploring their implications for GHG fluxes.	Rounsevell & Reay (2009) [32]
indirect land-use change	2020	bioenergy crops	secondary forest, other	EU, CIS region	analysing the impact of possible changes in EU biofuels trade policies on global agricultural production.	Al-Riffai et al. (2010) [33]
			cropland, forest (primary), pasture, savanna grassland	Brazil		
			cropland, forest, pasture	Indonesia and Malaysia		
			cropland, pasture	Latin America		

Type of land-use change	Projected year	Crop	Previous land-use	Location/Country	Purpose of the study	Reference
indirect land-use change	by 2020	bioenergy crops	-	South & Central America (especially Brazil)	assessing the global and sectional implications of the growing demand for bio-based inputs for energy and fuel production.	Banse et al. (2008) [34]
displacement of other crops/cropland expansion	2000-30	all types of bioenergy crops	cropland	EU15+, EU12, Ukraine	estimating land available for biofuel feedstock production without putting food supply or nature conservation at risk.	Fischer et al. (2010) [35]
indirect land-use change	2020	bioenergy crops	pasture, managed forest, savanna & grassland	Brazil, Latin America, CIS, Sub-saharan Africa	updating the previous study on the analysis of the impacts of EU biofuels trade policies (Al-Riffai et al (2010)).	Laborde (2011) [36]
indirect land-use change	2020	bioenergy crops	savanna (58%), forest (primary) (15%), pasture (14%)	Brazil	estimating changes in GHG emissions from soil and above- and belowground biomass resulting from global land use changes caused by the production of biofuels.	Hiederer et al. (2010) [37]
			forest (managed)	EU27		
deforestation/cropland expansion	-	cassava, oil palm	various (mostly forest)	Laos, Cambodia, Malaysia, Indonesia, the Philippines	reviewing the recent expansion of global biofuels production and its consequences.	Naylor et al. (2007) [38]
deforestation/cropland expansion/indirect land-use change	2001-15	corn (US)	forest	global (e.g. EU27, US, East Europe & former USSR, Brazil, Central America)	estimating land-use changes and GHG emissions associated with US corn ethanol production in accordance with the US biofuels policy.	Tyner et al. (2010) [39]
grassland conversion/cropland expansion/indirect land-use change	2001-15	corn (US)	grassland	global (e.g. East Europe & former USSR, Russia, Sub-Saharan Africa, USA, South America, Canada)		

Type of land-use change	Projected year	Crop	Previous land-use	Location/Country	Purpose of the study	Reference
deforestation/cropland expansion/indirect land-use change	2016	corn (US)	forest & grassland	US, Brazil, India, China, Canada, Europe	calculating GHG emissions from worldwide land-use change resulting from the expansion of corn ethanol in the US.	Searchinger et al. (2008) [40]
crop expansion/displacement of other crops	-	corn	land under Conservation Reserve Program (CRP)	the Raccoon River watershed, Iowa, USA	examining the consequences of future land-use and land cover change from biofuel expansion on the water balance.	Schilling et al. (2008) [1]
crop expansion/grassland conversion	-	corn (and soybean as a rotation crop)	lands under CRP	Iowa, USA	examining the land-use impacts of biofuels expansion on both the intensive and extensive margin, and its environmental consequences (sediment, nitrogen and phosphorous, and soil carbon).	Secchi et al. (2011) [41]
grassland conversion/crop expansion	2007 onwards	corn (irrigated)	lands under CRP & native grassland	the Ogallala Aquifer in the Great Plains, USA	providing recommendations for fuel and natural resource policy in the US.	Roberts et al. (2007) [42]
agricultural intensification	-	corn (and soybean as a rotation crop)	cropland	Iowa, USA	examining the land-use impacts of biofuels expansion on both the intensive and extensive margin, and its environmental consequences (sediment, nitrogen and phosphorous, and soil carbon).	Secchi et al. (2011) [41]
crop expansion/ grassland conversion	-	corn (and rotation with other crops)	pasture and rangeland	the Corn Belt and the Lake States, USA	examining how corn price increase will affect land use and analysing how the land use change will affect nitrate runoff, percolation, soil erosion and carbon sequestration in the US Midwest.	Langpap and Wu (2011) [43]
deforestation/crop expansion	-	corn, soybean	forest	the Mississippi river basin, USA	evaluating land surface energy and water balance changes resulting from a change in land cover.	Twine et al. (2004) [44]
grassland conversion/ crop expansion	-	corn, soybean	grassland/savanna			

Type of land-use change	Projected year	Crop	Previous land-use	Location/Country	Purpose of the study	Reference
displacement of other crops	-	grass (warm season grass, cool season grass)	cropland for corn	the Raccoon River watershed, Iowa, USA	examining the consequences of future land-use and land cover change from biofuel expansion on the water balance.	Schilling et al. (2008) [1]
crop expansion	2070	Indian mustard	marginal agricultural land	Australia	assessing the potential suitability for the production of biodiesel crops under different IPCC emission scenarios and conducting a preliminary economic analysis of the profitability of these crops.	Odeh et al. (2011) [45]
crop expansion	-	Jatropha curcas	wasteland	India	arguing the drawbacks of India's biofuel policy.	Rajagopal (2007) [46]
displacement of other crops	-	lignocellulosic crops (miscanthus)	wheat	Cambridgeshire, UK	quantifying the effects on farmland birds at the field-scale of switching land-use from intensive annual cereal cropping to commercial production of miscanthus biomass crops.	Bellamy et al. (2009) [47]
displacement of other crops	late 1990s -2006	lignocellulosic crops (willow short rotation coppice)	cereal crops	North Nottinghamshire, UK	comparing biodiversity impacts of willow SRC plantations with that observed in arable and set-aside in the UK.	Rowe et al. (2011) [48]
deforestation/displacement of other crops	2004-07	lignocellulosic crop (miscanthus)	forest & cropland for corn	Midwest, USA	evaluating hydrological changes resulting from large-scale conversion to one that contains miscanthus.	Vanloocke et al. (2010) [49]
displacement of other crops	2005 2050	lignocellulosic crops (miscanthus and switchgrass)	corn	Midwest, USA	exploring potential hydrologic change associated with simultaneous land use conversion to lignocellulosic crops.	Le et al (2011) [50]

Type of land-use change	Projected year	Crop	Previous land-use	Location/Country	Purpose of the study	Reference
displacement of other crops	-	lignocellulosic crops (miscanthus and switchgrass)	corn	the Iowa River Basin, USA	evaluating the effects of potential land cover change from a portion of current cornfield or native grassland to dedicated bioenergy crops on water quality at the watershed scale.	Wu & Liu (2012) [51]
grassland conversion	-		native grassland			
displacement of other crops (2030)	2012 2015	lignocellulosic crops (switchgrass)	pasture	the southeast, USA	estimating land use changes of expanding ethanol production in the USA and ascertaining the impacts on water use and water quality.	Ugarte et al. (2010) [52]
crop expansion	-	lignocellulosic crops	set-side, cropland & grassland (pasture)	UK	exploring land-use and biomass supply potential and providing recommendations for further research.	Rowe et al. (2009) [53]
displacement of other crops	late 1990s -2006	lignocellulosic crops (willow short rotation coppice)	set-aside	North Nottinghamshire, UK	comparing biodiversity impacts of willow SRC plantations with that observed in arable and set-aside in the UK.	Rowe et al. (2011) [48]
pasture abandonment/ cropland expansion	2000-30	lignocellulosic crops	‘surplus’ pasture	EU15+, EU12, Ukraine	estimating land available for biofuel feedstock production without putting at risk food supply or nature conservation.	Fischer et al. (2010) [35]
grassland conversion/displacement of other crops/ crop expansion	2007 onwards	lignocellulosic crops	land under CRP	USA	reviewing the recent expansion of global biofuels production and its consequences.	Naylor et al. (2007) [38]
grassland conversion/displacement of other crops/ crop expansion	-	lignocellulosic crops	land under CRP	Upper Midwest, USA	presenting overview of biofuels and sustainability issues.	Payne (2010) [54]
displacement of other crops	-	miscanthus	cropland (e.g. corn, soybean)	Salt Creek watershed, Illinois, USA	estimating the potential effects on riverine nitrate load of miscanthus in place of conventional crops.	Ng et al. (2010) [55]

Type of land-use change	Projected year	Crop	Previous land-use	Location/Country	Purpose of the study	Reference
deforestation/crop expansion	-	oil palm	forest	Borneo, Indonesia	reviewing the recent expansion of global biofuels production and its consequences.	Naylor et al. (2007) [38]
crop expansion/ indirect land-use change	2020	oil palm	forest, grassland, mixed, savanna, shrub, wetland	Indonesia, Malaysia, Columbia	developing an understanding of the chain of causes and effects that lead from an increased demand for biofuel feedstock to indirect land use change, and provides a framework for capturing and quantifying those relationships.	Bauen et al. (2010) [56]
deforestation/displacement of other crops/crop expansion	1990-2007	oil palm	forest	Indonesia, Malaysia	assessing strategies for conservation of orangutan habitat.	Nantha and Tisdell (2009) [6]
crop expansion	2070	pongamia	marginal agricultural land	Australia	assessing the potential suitability for the production of biodiesel crops under different IPCC emission scenarios and conducting a preliminary economic analysis of the profitability of these crops.	Odeh et al. (2011) [45]
displacement of other crops	2020	rapeseed	break crops (flax fibre, linseed, lupins, dry peas and soybeans)	Europe	developing an understanding of the chain of causes and effects that lead from an increased demand for biofuel feedstock to indirect land use change, and providing a framework for capturing and quantifying those relationships.	Bauen et al. (2010) [56]
displacement of other crops	2030	soybean	cropland for annual crops (e.g. wheat, sunflower, maize and soybean)	La Pampa, Argentina	analysing the socio-economic and environmental performance of two large-scale bioenergy production scenarios.	van Dam et al. (2009) [29]
cropland expansion	2030	soybean	degraded grassland			
grassland conversion/ cropland expansion	2030	soybean	native grassland			

Type of land-use change	Projected year	Crop	Previous land-use	Location/Country	Purpose of the study	Reference
crop expansion	2020	soybean	cattle pasture	South-western states (Mato Grosso, Mato Grosso do Sul, Goiás and Minas Gerais), Brazil	projecting land-use changes caused by bioenergy crop expansion in 2020.	Lapola et al. (2010) [57]
cropland expansion	2020	sugarcane	cattle pasture	South-eastern states (São Paulo, Minas Gerais and Paraná), Brazil		
pasture abandonment/ cropland expansion	-	sugarcane	cattle pasture	Amazon, Brazil	reviewing the recent expansion of global biofuels production and its consequences.	Naylor et al (2007) [38]
displacement of other crops	2030	switchgrass	cropland for annual crops (e.g. wheat, sunflower, maize and soybean)	La Pampa, Argentina	analysing the socio-economic and environmental performance of two large-scale bioenergy production scenarios.	van Dam et al. (2009) [29]
cropland expansion	2030	switchgrass	degraded grassland			
grassland conversion/ crop expansion	2030	switchgrass	native grassland			
cropland expansion	-	various (sorghum, miscanthus, switchgrass, native grasses)	marginal land	three watersheds in Michigan, USA	evaluating the long-term water quality implications of large-scale bioenergy cropping system expansion.	Love & Nejadhashemi (2011) [58]
cropland expansion/ displacement of other crops	-	various (corn, canola, rye, sorghum, soybean, miscanthus, corn, switchgrass, native grasses)	cropland for other crops			

Type of land-use change	Projected year	Crop	Previous land-use	Location/Country	Purpose of the study	Reference
deforestation/grassland conversion/ indirect land-use change	2020	cattle ranching	savanna grassland	Cerrado, Brazil	projecting land-use changes caused by bioenergy crop expansion in 2020.	Lapola et al. (2010) [57]

Appendix 3: Minutes of interviews

The following minutes document a summary of discussions that took places as part of visits to the Burnett region (Bundaberg and Kingaroy). The main objectives of those interviews were: to obtain regional specific data and information; and to confirm the information received from the datasets.

1. **Mr. Robert Doyle (Burnett Mary Field Officer, Growcom Australia)**

   Date: 17 February, 2012

- Mr. Doyle is responsible for helping horticultural farms in the region to apply for Reef Rescue funding assistance. The money (AUD 560,000) is equally distributed among 35 40 farmers in this region (AUD 14,000).
- The financial assistance under the Reef Rescue supports up to 50% of farmers' investments, which will benefit water quality, such as controlled traffic farming utilising GPS, fertilisers and irrigation system, and weed seeker technology.
- Neither Department of Environment and Resource Management (DERM) nor Burnett Mary Regional Group (BMRG) is currently conducting water quality monitoring to evaluate the effectiveness of the financial assistance under the Reef Rescue program. However, the improvements made by the farmers are checked by the field officer.
- In Bundaberg, not much land clearing has happened since the mid-1990s. During the 1970s and 80s, massive amounts of land were cleared for agricultural expansion. In the mid-1990s, the rate of land clearing slowed down.
- In the Bundaberg district, irrigation was introduced in the 1970s (i.e. the Bundaberg Irrigation Area). Until then, dryland farming was conducted in this region.
- In recent years, sugarcane land has been converted into horticulture as it provides a higher profit margin. Cane farmers utilise the fallow period of sugarcane (20% of total lands) to produce vegetable and fruits. In 1993, the total area of horticulture in the Bundaberg Irrigation Area (36,000 ha) was 7,000 ha, while it reached 16,440 ha in 2009 (see Table 1). Mostly, macadamia nuts and avocados are replacing sugarcane land.
- The difference in the environmental impacts between sugarcane and horticulture is still unknown.

- DERM has a set of data including farm resource map, and soil data, which is provided for farmers.

Table 1. Production, estimated area grown and gross value of horticultural crops in the Bundaberg district – 2009 (Complied by Jerry L. Lovatt, DEEDI)

Crop	**Estimated area planted (ha)**	**Production p = packages t = tonnes**	**Estimated gross value ($)**
Fruit & Nuts			
Avocados	1,510	1,741,560 p	31,256,930
Bananas	65	94,980 p	2,541,670
Citrus	530	1,174,470 p	39,139,190
Custard apple	55	21,850 p	361,250
Lychee	160	25,620 p	868,290
Macadamia nuts	5,790	8,400 t	15,537,760
Mangos	545	179,920 p	3,587,340
Mangos (process)		107 t	53,700
Passionfruit	90	140,940 p	3,431,790
Pineapple (fresh)		131,320 p	2,491,090
Pineapple (process)	500	7,145 t	2,748,890
Stone fruit	80	30,290 p	772,140
Vegetables			
Beans	140	69,680 p	3,017,130
Button squash	110	107,930 p	2,241,790
Capsicum	320	1,296,300 p	17,449,450
Chilli	50	279,810 p	10,584,510
Cherry tomatoes	150	895,660 p	25,940,670
Cucumber	70	618,730 p	8,406,360
Egg fruit	40	164,110 p	2,813,490
Potatoes	120	2,940 t	2,912,770
Potatoes (process)	310	11,525 t	4,797,250
Pumpkins (large)	310	6,280 t	3,763,350
Rockmelon	170	341,520 p	5,398,570
Snow peas	680	512,150 p	19,396,190
Sweet corn	35	35,170 p	496,060
Sweet potatoes	1,260	2,529,640 p	52,351,030
Tomatoes	1,510	7,688,170 p	130,176,950
Watermelon	660	19,710 t	13,010,380
Zucchini	1,180	1,767,330 p	29,728,270
Miscellaneous Crops, Nurseries, Ornamentals etc.			18,608,590
TOTAL	**16,440** (sugarcane: 24,000)		**453,882,850**

2. **Mr. Neil Halpin (Agronomist, Queensland Department of Employment, Economic Development and Innovation [DEEDI], [Bundaberg])**

   Date: 17 February, 2012

- Mr Halpin is responsible for the sustainable sugarcane production system, including legume crops in rotation with sugarcane, application of fertilisers, soil compaction from excessive soil degradation, maintenance of soil organics using cane trash blanket.
- Horticulture started to occur in the Bundaberg region in the mid-1980s due to its high growth margins. Cultivation of tree crops (e.g. macadamia nuts) has gradually expanded over the past few decades.
- Horticulture must have much more serious environmental consequences than sugarcane production. It requires ten times more chemical pesticide and three times more fertiliser inputs than sugarcane production.
- International commodity price is the main driver for agricultural land-use change in the region.
- Currently, the sugarcane price is high so that sugarcane production is profitable. Increased demand for sugar is driven by the Chinese market.
- Spatial data sets including land suitability study/soil study may be available from Andrew Robson (Kingaroy DEEDI) and Andrew Dougall (local DEEDI offices).
- Currently, ethanol production from sugarcane has not been discussed in this region. (However, he mentioned the Pentland scheme in the Burdekin. In this scheme, grazing land is proposed to be converted to sugarcane production with the construction of a dam on the upper Burdekin River).

3. **Ian Crosthwaite (Agronomist, BGA AgriServices)**

   Date: 12 March, 2012

- Marginal agricultural land can be characterised by poor soil quality (e.g. sandy soil) (Plate A1), rockiness, acidity, wet soil, slope (<5%) (Plate A2), and limited depth of soil. In the South Burnett region, cropping production on many of these lands was abandoned in the past and converted to cattle grazing (Plate A3, A4, A5 & A6).

- Major clearing occurred between the late 19th century and 1960s. The land was cleared mainly for dairy and subdivided into 160 acre (65 hectare) blocks in the past century. However, the dairy industry has nearly disappeared from the region over the past two decades.

- The South Burnett presents such a diverse agricultural land-use: grazing, forest plantation for wood chipping (Plate A7), horticulture (tree crops, such as avocado and nectarine), peanut, cereals, corn, sorghum (Plate A8), alfalfa, hay etc.

- Cropping has been replaced gradually by cattle pasture since the 1980s due to decreased dependence on the agriculture sector and decreased economic profitability. Cropping requires intensive labour and investments when compared to grazing. Grazing doesn't require much labour. However, it is not profitable if you do not own the block of the land. Family members on most grazing properties tend to have income outside of the agricultural sector.

- The North and Central Burnett show similar land use change trends, but the loss of cropping lands has been much more prominent. The blocks of land in these regions are generally much larger than the South Burnett region.

- The farmer whom I visited first (Peter) owns 65 ha to produce mung beans, hay, alfalfa etc. (Plate A9).

- The marginal lands which we saw were categorised in class VI on the South Burnett land capability study.

Plate A1. Poor soil quality

Plate A2. Agricultural land with slope (<5%) used for grazing

Plate A3. Abandoned cropping land converted to cattle grazing

Plate A4. Abandoned cropping land converted to cattle grazing

Plate A5. Abandoned cropping land converted to cattle grazing

Plate A6. Abandoned cropping land converted to cattle grazing

Plate A7. Forest plantation for wood chipping

Plate A8. Sorghum field near Kingaroy

Plate A9. A farm in the South Burnett near Kingaroy

4. Damien O'Sullivan (Agronomist, DEEDI)

Date: 13 March, 2012

- Forest plantations (paulownia, blue gum, spotted gum, corymbia, chinchilla white gum) were established under Managed Investment Schemes (MIS) in the South Burnett region. Many plantations have taken up good cropping lands (Plate A10). Some of the forests have been abandoned after the managed investment companies collapsed (Plate A11).
- The forest owned by Forestry Plantations Queensland is leased by graziers. The land presents an example of utilising the land for both tree plantation and grazing (forested grazing) (Plate A12).

- Tarong power station supplies around 15% of the power supply in QLD (Plate A13). The power station purchased 1,200 ha of land in the region (Edenvale South Road) for coal mining/underground coal gasification. However, this project has been held up as this process causes significant environmental impacts. The land has been left unused for a while.
- Duboisia is grown for pharmaceutical purpose (e.g. eye drops) by a German pharmaceutical company in the region. Residue after the processing such as the extraction of the leaves (e.g. stems etc.) can be a potential source of bioenergy (heat and electricity).
- In the South Burnett region, cropping lands has decreased over the past 10 to 15 years due to farm population decline and the decline of the agricultural sector. Especially, farmers in the South Burnett region have moved out of cropping as they could not survive economically.
- North Burnett has less population, broad-scale cropping, and more grazing than the South Burnett region.
- In the region, cropping was abolished on large areas due to low productivity and marginality. However, most of these lands are still in use for cattle grazing.
- Regrowth of trees is a significant issue for farmers. They are using chemicals to control and prevent vegetation regrowth.
- Now only 5 to 10% of the population is engaging in the agriculture sector in the region.
- The region can produce a wide variety of crops from tropical fruits such as bananas to temperate crops, because the topography of the region provides different regional climate. Therefore, the location of crop production requires specific mapping.

Plate A10. Example of a forest plantation on good agricultural land

Plate A11. Abandoned forest

Plate A12. Forest owned by Forestry Plantations Queensland

Plate A13. Land purchased for coal mining/underground coal gasification

Table A4.1 A description of the eight land capability classes used in classifying land in the Burnett region (Source: Donnollan and Searle, 1999; Kent, 2002)

Land class	Description
Class I	• Land suitable for all agriculture and pastoral uses. • Land is suited to a wide range of crops and is highly productive. • Land presents no limitations to use of machinery or choice of implements. • Wind and water erosion hazard are low even under intensive cultivation.
Class II	• Land suitable for all agricultural uses but with slight restrictions to use for cultivation in one or more of the following categories: • Land with some limitation to the choice of crops and/or slight restrictions to productivity. • Land with some impediment to the use of cultivation machinery which limits the choice of implements or restricts the conditions for successful operation. • Land which under cultivation requires simple conservation practices to reduce soil loss to an acceptable level. These include agronomic practices such as contour working, strip cropping, stubble mulching.
Class III	• Land suitable for all agricultural uses but with moderate restrictions to use for cultivation in one or more of the following categories: • Land with moderate limitations to the choice of crops and/or moderate restrictions to productivity. • Land with moderate impediment to the use of cultivation machinery which limits the choice of implements or restricts the conditions for successful operation. • Land which under cultivation requires intensive conservation practices to reduce soil loss to an acceptable level. These include contour banking systems and intensive residue management involving specialised machinery.
Class IV	• Land primarily suited to pastoral use but which may be safely used for occasional cultivation with careful management. • Land on which the choice of crops is severely restricted and/or conditions is such that productivity under cropping is severely limited. • Land with severe impediment to the use of cultivation machinery which limits the choice of implements or severely restricts the conditions for successful operation. • Land which cannot be used safely for permanent cultivation. If cropped, a pasture phase must be the major component in the cropping program to limit soil loss to an acceptable level.
Class V	• Land which in all other characteristics would be arable but has limitations which, unless removed, make cultivation impractical and/or uneconomical.
Class VI	• Land which is not suitable for cultivation but is well suited to pastoral use and on which pasture improvement involving the use of machinery is practicable.
Class VII	• Land which is not suitable for cultivation but on which pastoral use is possible only with careful management. Pasture improvement involving the use of machinery is not practicable.
Class VIII	• Land which has such severe limitations that it is unsuited for either cultivation or grazing.

Table A5.1 FU class corresponding to Australian Land Use and Management Classification (ALUM) (Source: Fentie, in press) (1/2).

ALUM				FU class
LU_CODE	PRIMARY	SECONDARY	TERTIARY	
1.1.3	Conservation and natural environments	Nature conservation	Natural park	Conservation
1.1.4			Natural feature protection	
1.1.5			Habitat/species management area	
1.1.7			Other conserved area	
1.3.0		Other minimal use	-	
1.3.1			Defence	
1.3.3			Remnant native cover	
2.1.0	Production from relatively natural environments	Grazing natural vegetation	-	Grazing Forested
2.1.0		Grazing natural vegetation		Grazing Open
2.2.0		Production forestry	-	Forestry
3.1.0	Production from dryland agriculture and plantations	Plantation forestry	-	
3.1.2			Softwood production	
3.1.3			Other forest production	
3.3.0		Cropping	-	Dryland Cropping
3.3.5			Sugar	Sugarcane
3.4.0		Perennial horticulture		Horticulture
3.4.1			Tree fruits	
3.4.5			Shrub nuts, fruits & berries	
4.3.0	Production from irrigated agriculture and plantations	Irrigated cropping	-	Irrigated Cropping
4.3.5			Irrigated sugar	Sugarcane
4.4.0		Irrigated perennial horticulture	-	Horticulture
4.4.1			Irrigated tree fruits	
4.4.2			Irrigated oleaginous fruits	
4.4.3			Irrigated tree nuts	
4.4.4			Irrigated vine fruits	
4.4.5			Irrigated shrub nuts, fruits & berries	
4.4.7			Irrigated vegetables & herbs	
4.5.0		Irrigated seasonal horticulture	-	
4.5.4			Irrigated vegetables & herbs	

Table A5.1 FU class corresponding to Australian Land Use and Management Classification (ALUM) (Source: Fentie, in press) (2/2).

ALUM				FU class
LU_CODE	PRIMARY	SECONDARY	TERTIARY	
5.1.0	Intensive uses	Intensive horticulture	-	Horticulture
5.1.3			Glasshouses (hydroponic)	
5.2.0		Intensive animal production	Intensive animal production	Other
5.2.1			Dairy	Grazing Open
5.2.2			Cattle	Other
5.2.4			Poultry	
5.2.5			Pigs	
5.2.6			Aquaculture	
5.3.0		Manufacturing and industrial	-	
5.4.0		Residential	-	Urban
5.4.1			Urban residential	
5.4.2			Rural residential	
5.5.0		Services	-	Other
5.5.1			Commercial services	
5.5.2			Public services	
5.5.3			Recreation and culture	
5.5.4			Defence facilities	
5.5.5			Research facilities	
5.6.0		Utilities	-	
5.6.1			Electricity generation/transmission	
5.7.1		Transport and communication	Airports/aerodromes	
5.8.0		Mining	-	
5.8.2		Mining	Quarries	
5.9.0		Waste treatment and disposal	-	
6.1.0	Water	Lake	-	Water
6.2.0		Reservoir/dam		
6.3.0		River		
6.4.0		Channel/aqueduct		
6.5.0		Marsh/wetland		
6.5.1		Marsh/wetland	Marsh/wetland - conservation	

Table A5.2 The rainfall-runoff model (Simhyd) parameters used for the Burnett River catchment (Source: Fentie, in press; Queensland Department of Natural Resources and Mines) (1/4)

	Conservation Forestry Grazing forested	**Grazing open Urban Other**	**Dryland Cropping Sugarcane Horticulture Irrigated cropping**
B1 (SC#1)			
Baseline coeff.	0.370936	0.377202	0.3
Impervious Threshold	1	1	1
Infiltration coeff.	135	164	200
Infiltration shape	5.892438	5.063659	3
Interflow Coeff.	0.107159	0.107272	0.1
Perv. Fraction	0.9	0.9	0.9
RISC	1.518922	1.578897	1.5
Recharge coefficient	0.196045	0.19179	0.2
SMSC	257.6511	279.1241	320
B2 (SC#2)			
Baseline coeff.	0.243996	0.296474	0.298738
Impervious Threshold	1	1	1
Infiltration coeff.	161	127	170
Infiltration shape	4.756243	6.928832	3.793091
Interflow Coeff.	0.116666	0.146042	0.110995
Perv. Fraction	0.9	0.9	0.9
RISC	1.21235	1.111096	1.374125
Recharge coefficient	0.386888	0.471959	0.254938
SMSC	218.222	176.1891	255.0601
B3 (SC#3)			
Baseline coeff.	0.415829	0.562298	0.319501
Impervious Threshold	1	1	1
Infiltration coeff.	239	170	223
Infiltration shape	3.604445	3.224696	2.452801
Interflow Coeff.	0.068355	0.067351	0.09962
Perv. Fraction	0.9	0.9	0.9
RISC	2.782948	2.858893	1.584813
Recharge coefficient	0.088916	0.118254	0.195622
SMSC	421.1494	123.528	336.4143
B4 (SC#4)			
Baseline coeff.	0.243996	0.296474	0.298738
Impervious Threshold	1	1	1
Infiltration coeff.	161	127	170
Infiltration shape	4.756243	6.928832	3.793091
Interflow Coeff.	0.116666	0.146042	0.110995
Perv. Fraction	0.9	0.9	0.9
RISC	1.21235	1.111096	1.374125
Recharge coefficient	0.386888	0.471959	0.254938
SMSC	218.222	176.1891	255.0601
B5 (SC#5)			
Baseline coeff.	0.29466	0.292521	0.299432
Impervious Threshold	1	1	1
Infiltration coeff.	224	231	201
Infiltration shape	2.767503	2.756014	3.001013
Interflow Coeff.	0.08447	0.222935	0.099144
Perv. Fraction	0.9	0.9	0.9
RISC	1.859437	1.997645	1.524482
Recharge coefficient	0.546768	0.496899	0.196575
SMSC	414.0625	482.6698	325.9859

Table A5.2 The rainfall-runoff model (Simhyd) parameters used for the Burnett River catchment (Source: Fentie, in press; Queensland Department of Natural Resources and Mines) (2/4)

	Conservation Forestry Grazing forested	**Grazing open Urban Other**	**Dryland Cropping Sugarcane Horticulture Irrigated cropping**
B6 (SC#6)			
Baseline coeff.	0.123269	0.109846	0.29986
Impervious Threshold	1	1	1
Infiltration coeff.	208	146	232
Infiltration shape	2.158132	1.21411	2.417733
Interflow coeff.	0.067347	0.061473	0.0992
Perv. Fraction	0.9	0.9	0.9
RISC	1.513741	1.844144	1.526362
Recharge coefficient	0.44318	0.321059	0.19648
SMSC	500	500	363.1484
B7 (SC#7)			
Baseline coeff.	0.325238	0.342599	0.300799
Impervious Threshold	1	1	1
Infiltration coeff.	220	178	200
Infiltration shape	2.460556	3.454713	2.987021
Interflow coeff.	0.103114	0.010639	0.100304
Perv. Fraction	0.9	0.9	0.9
RISC	1.617674	1.526911	1.507571
Recharge coefficient	0.173724	0.070662	0.199896
SMSC	293.6891	318.8488	319.835
B8 (SC#8)			
Baseline coeff.	0.372122	1	0.312314
Impervious Threshold	1	1	1
Infiltration coeff.	213	303	207
Infiltration shape	2.792884	4.403711	2.894095
Interflow coeff.	0.083064	0.022503	0.09721
Perv. Fraction	0.9	0.9	0.9
RISC	2.158071	5	1.587484
Recharge coefficient	0.133131	0.06033	0.187179
SMSC	333.1011	173.693	319.6815
B9 (1) (SC#9)			
Baseline coeff.	0.191058	0.265985	0.298775
Impervious Threshold	1	1	1
Infiltration coeff.	122	193	198
Infiltration shape	3.866437	3.425146	3.07226
Interflow coeff.	0.130364	0.14814	0.09743
Perv. Fraction	0.9	0.9	0.9
RISC	1.339944	1.207802	1.51924
Recharge coefficient	0.114784	0.219015	0.189671
SMSC	500	437.2037	320.1703
B9 (2) (SC#10)			
Baseline coeff.	0.191058	0.265985	0.298775
Impervious Threshold	1	1	1
Infiltration coeff.	122	193	198
Infiltration shape	3.866437	3.425146	3.07226
Interflow coeff.	0.130364	0.14814	0.09743
Perv. Fraction	0.9	0.9	0.9
RISC	1.339944	1.207802	1.51924
Recharge coefficient	0.114784	0.219015	0.189671
SMSC	500	437.2037	320.1703

Table A5.2 The rainfall-runoff model (Simhyd) parameters used for the Burnett River catchment (Source: Fentie, in press; Queensland Department of Natural Resources and Mines) (3/4)

	Conservation Forestry Grazing forested	**Grazing open Urban Other**	**Dryland Cropping Sugarcane Horticulture Irrigated cropping**
B10 (SC#11)			
Baseline coeff.	1	0.496764	0.306334
Impervious Threshold	1	1	1
Infiltration coeff.	163	187	200
Infiltration shape	4.227282	3.498354	3.010662
Interflow coeff.	0.060423	0.058561	0.098276
Perv. Fraction	0.9	0.9	0.9
RISC	4.727772	1.652928	1.479093
Recharge coefficient	0.026573	0.059259	0.191328
SMSC	463.6926	241.4253	315.1342
B11 (SC#12)			
Baseline coeff.	0.339618	0.429594	0.300888
Impervious Threshold	1	1	1
Infiltration coeff.	98	400	200
Infiltration shape	1.755267	1.411709	3.001597
Interflow coeff.	0.026411	0.128883	0.09969
Perv. Fraction	0.9	0.9	0.9
RISC	2.846744	3.433038	1.506217
Recharge coefficient	0.037507	0.027792	0.198511
SMSC	314.4235	500	320.242
B12 (SC#13)			
Baseline coeff.	0.339618	0.429594	0.300888
Impervious Threshold	1	1	1
Infiltration coeff.	98	400	200
Infiltration shape	1.755267	1.411709	3.001597
Interflow coeff.	0.026411	0.128883	0.09969
Perv. Fraction	0.9	0.9	0.9
RISC	2.846744	3.433038	1.506217
Recharge coefficient	0.037507	0.027792	0.198511
SMSC	314.4235	500	320.242
B13 (SC#14)			
Baseline coeff.	0.339618	0.429594	0.300888
Impervious Threshold	1	1	1
Infiltration coeff.	98	400	200
Infiltration shape	1.755267	1.411709	3.001597
Interflow coeff.	0.026411	0.128883	0.09969
Perv. Fraction	0.9	0.9	0.9
RISC	2.846744	3.433038	1.506217
Recharge coefficient	0.037507	0.027792	0.198511
SMSC	314.4235	500	320.242
B14 (SC#15)			
Baseline coeff.	0.191058	0.265985	0.298775
Impervious Threshold	1	1	1
Infiltration coeff.	122	193	198
Infiltration shape	3.866437	3.425146	3.07226
Interflow coeff.	0.130364	0.14814	0.09743
Perv. Fraction	0.9	0.9	0.9
RISC	1.339944	1.207802	1.51924
Recharge coefficient	0.114784	0.219015	0.189671
SMSC	500	437.2037	320.1703

Table A5.2 The rainfall-runoff model (Simhyd) parameters used for the Burnett River catchment (Source: Fentie, in press; Queensland Department of Natural Resources and Mines) (4/4)

	Conservation Forestry Grazing forested	**Grazing open Urban Other**	**Dryland Cropping Sugarcane Horticulture Irrigated cropping**
B15 (SC#16)			
Baseline coeff.	0.129686	0.248639	0.29917
Impervious Threshold	1	1	1
Infiltration coeff.	177	196	200
Infiltration shape	0.995291	0.723993	2.999374
Interflow coeff.	0.087264	0.037415	0.099919
Perv. Fraction	0.9	0.9	0.9
RISC	1.073553	2.757754	1.496529
Recharge coefficient	0.109485	0.200002	0.199655
SMSC	499.034	500	322.2333
B16 (SC#17)			
Baseline coeff.	1	0.826481	0.3
Impervious Threshold	1	1	1
Infiltration coeff.	114	182	200
Infiltration shape	3.258124	2.429847	3
Interflow coeff.	0.058838	0.08676	0.1
Perv. Fraction	0.9	0.9	0.9
RISC	3.168472	1.463884	1.5
Recharge coefficient	0.02405	0.115138	0.2
SMSC	287.5676	394.6693	320
B17 (SC#19)			
Baseline coeff.	0.146624	0.132156	0.185732
Impervious Threshold	1	1	1
Infiltration coeff.	252	234	209
Infiltration shape	2.912345	5.294912	2.88662
Interflow coeff.	0.031346	0.042795	0.09526
Perv. Fraction	0.9	0.9	0.9
RISC	0.967419	1.550027	1.454753
Recharge coefficient	0.131722	0.052531	0.220236
SMSC	340.5281	256.8889	335.296
B18 (SC#18)			
Baseline coeff.	0.354192	0.238194	0.404707
Impervious Threshold	1	1	1
Infiltration coeff.	358	140	323
Infiltration shape	3.684624	1.274662	1.845544
Interflow coeff.	0.059972	0.043277	0.096891
Perv. Fraction	0.9	0.9	0.9
RISC	2.90299	1.972105	1.055957
Recharge coefficient	0.042144	0.059688	0.338626
SMSC	318.9051	240.2174	500

Table A5.3 The Event Mean Concentration (EMC) and Dry Weather Concentrations (DWC) values used for the Burnett River catchment (Source: Fentie, in press; Queensland Department of Natural Resources and Mines).

FU class	Constituent	Runoff Element	Conc_mg/l	FU class	Constituent	Runoff Element	Conc_mg/l
Conservation	TSS	EMC	33	**Horticulture**	TSS	EMC	275
		DWC	16.5			DWC	137.5
	TN	EMC	0.37		TN	EMC	3.01
		DWC	0.18			DWC	1.5
	TP	EMC	0.07		TP	EMC	0.54
		DWC	0.03			DWC	0.28
Forestry	TSS	EMC	44	**Irrigated cropping**	TSS	EWC	1,507.83
		DWC	22			DWC	138
	TN	EMC	0.48		TN	EWC	4.18
		DWC	0.24			DWC	0.85
	TP	EMC	0.08		TP	EWC	1.04
		DWC	0.04			DWC	0.09
Grazing forested	TSS	EMC	235.7	**Urban**	TSS	EWC	220
		DWC	100			DWC	110
	TN	EMC	0.6		TN	EWC	2.4
		DWC	0.17			DWC	1.2
	TP	EMC	0.17		TP	EWC	0.42
		DWC	0.02			DWC	0.22
Grazing open	TSS	EMC	324.12	**Water**	TSS	EWC	0
		DWC	100			DWC	0
	TN	EMC	1.02		TN	EWC	0
		DWC	0.34			DWC	0
	TP	EMC	0.22		TP	EWC	0
		DWC	0.03			DWC	0
Dryland cropping	TSS	EMC	2,522.54	**Other**	TSS	EWC	110
		DWC	165			DWC	55
	TN	EMC	7.23		TN	EWC	1.2
		DWC	1.02			DWC	0.6
	TP	EMC	1.84		TP	EWC	0.22
		DWC	0.1			DWC	0.11
Sugarcane	TSS	EMC	693.23				
		DWC	0				
	TN	EMC	13.92				
		DWC	0				
	TP	EMC	0.9				
		DWC	0				

Table A5.4 Biodiversity Status (BS) and Vegetation Management Class (Source: Queensland Department of Environment and Heritage Protection, 2013).

Class	Description
Endangered	A regional ecosystem is listed as 'Endangered' *under the Vegetation Management Act 1999* if: • remnant vegetation is less than 10 per cent of its pre-clearing extent across the bioregion; or 10–30% of its pre-clearing extent remains and the remnant vegetation is less than 10,000 hectares. In addition to the criteria listed for an 'Endangered' regional ecosystems under *the Vegetation Management Act 1999*, for biodiversity planning purposes a regional ecosystem is listed with a Biodiversity Status of 'Endangered' if: • less than 10 per cent of its pre-clearing extent remains unaffected by severe degradation and/or biodiversity loss*; or • 10–30 per cent of its pre-clearing extent remains unaffected by severe degradation and/or biodiversity loss and the remnant vegetation is less than 10,000 hectares; or • it is a rare regional ecosystem** subject to a threatening process***.
Of concern	A regional ecosystem is listed as 'Of concern' under *the Vegetation Management Act 1999* if: • remnant vegetation is 10–30 per cent of its pre-clearing extent across the bioregion; or more than 30 per cent of its pre-clearing extent remains and the remnant extent is less than 10,000 hectares. In addition to the criteria listed for an 'Of concern' regional ecosystem under *the Vegetation Management Act 1999*, for biodiversity planning purposes a regional ecosystem is listed with a Biodiversity Status 'Of concern' if: • 10–30 per cent of its pre-clearing extent remains unaffected by moderate degradation and/or biodiversity loss****.
No concern at present /Least concern	A regional ecosystem is listed as 'Least concern' under *the Vegetation Management Act 1999* if: • remnant vegetation is over 30 per cent of its pre-clearing extent across the bioregion, and the remnant area is greater than 10,000 hectares. In addition to the criteria listed for 'Least concern' regional ecosystems under *the Vegetation Management Act 1999*, for biodiversity planning purposes a regional ecosystem is listed with a Biodiversity Status of 'No concern at present' if: • the degradation criteria listed above for 'Endangered' or 'Of concern' regional ecosystems are not met.

* severe degradation and/or biodiversity loss is defined as: floristic and/or faunal diversity is greatly reduced but unlikely to recover within the next 50 years even with the removal of threatening processes; or soil surface is severely degraded.
** Rare regional ecosystem refers to pre-clearing extent (1000 ha); or patch size (100 ha and of limited total extent across its range)
*** Threatening processes are those that are reducing or will reduce the biodiversity and ecological integrity of a regional ecosystem. For example, clearing, weed invasion, fragmentation, inappropriate fire regime or grazing pressure, or infrastructure development.
**** Moderate degradation and/or biodiversity loss is defined as: floristic and/or faunal diversity is greatly reduced but unlikely to recover within the next 20 years even with the removal of threatening processes; or soil surface is moderately degraded.

Table A5.5 Land use areas of each Functional Unit (FU) and sub-catchment on 1992/93, 2001/02 and 2005/06 FU maps (Figure 5.2) (1/3) (Unit: ha).

Sub-catchment #	Conservation			Forestry			Grazing forested			Grazing open		
	1992/93	2001/02	2005/06	1992/93	2001/02	2005/06	1992/93	2001/02	2005/06	1992/93	2001/02	2005/06
B1 (SC#1)	97,102	98,093	42,756	22,311	23,506	10,231	109,699	93,844	163,983	110,290	118,871	124,636
B2 (SC#2)	7,052	7,726	11,007	2,729	2,872	-	39,364	38,233	38,449	30,456	31,434	28,982
B3 (SC#3)	9,196	9,945	18,504	11,993	11,685	218	70,405	56,938	68,012	53,053	57,375	60,049
B4 (SC#4)	33,053	35,335	41,652	48,998	49,832	33,243	111,038	82,856	109,835	137,533	144,265	144,945
B5 (SC#5)	1,711	2,688	5,503	7,309	6,649	5,352	89,912	77,468	82,929	78,506	91,202	85,300
B6 (SC#6)	433	497	6,344	17,090	17,368	17,057	84,901	82,350	78,780	47,431	54,878	53,438
B7 (SC#7)	7,056	8,223	9,556	3,971	4,247	3,718	50,246	41,071	49,188	87,042	100,139	84,853
B8 (SC#8)	2,892	3,134	8,585	20,133	19,668	20,337	94,241	83,199	92,824	79,931	100,581	94,885
B9 (SC#9 and 10)	470	335	5,539	2,831	2,617	2,734	20,471	21,134	16,144	8,044	7,516	7,487
B10 (SC#11)	10,767	8,527	12,546	72,159	73,610	73,245	67,350	56,648	58,945	28,188	25,937	34,033
B11 (SC#12)	12,886	12,491	20,243	21,952	21,460	20,921	78,417	63,378	64,233	67,669	79,217	76,024
B12 (SC#13)	871	744	3,533	72,706	73,152	71,965	33,929	25,572	26,246	29,115	25,011	34,180
B13 (SC#14)	48,767	45,916	9,216	56,771	55,102	55,988	49,634	38,384	79,397	51,364	52,615	62,673
B14 (SC#15)	2,676	2,428	13,645	9,115	10,135	9,863	137,585	121,255	126,896	60,548	76,084	61,239
B15 (SC#16)	6,300	6,517	4,071	34,299	34,823	35,369	101,169	66,804	90,801	103,604	137,105	118,928
B16 (SC#17)	3,346	3,818	12,969	13,324	13,376	6,257	34,958	26,143	32,244	19,940	28,331	20,082
B17 (SC#19)	9,207	8,174	31,302	38,385	38,736	11,977	58,932	54,170	66,624	53,532	59,881	51,502
B18 (SC#18)	34,974	33,515	30,427	20,995	20,433	15,634	50,945	43,052	64,705	49,905	58,106	44,600
TOTAL	**288,759**	**288,106**	**287,398**	**477,071**	**479,271**	**394,109**	**1,282,196**	**1,072,499**	**1,310,235**	**1,096,151**	**1,248,548**	**1,187,836**

Table A5.5 Land use areas of each FU and sub-catchment on 1992/93, 2001/02 and 2005/06 FU maps (Figure 5.2) (2/3)

(Unit: ha)

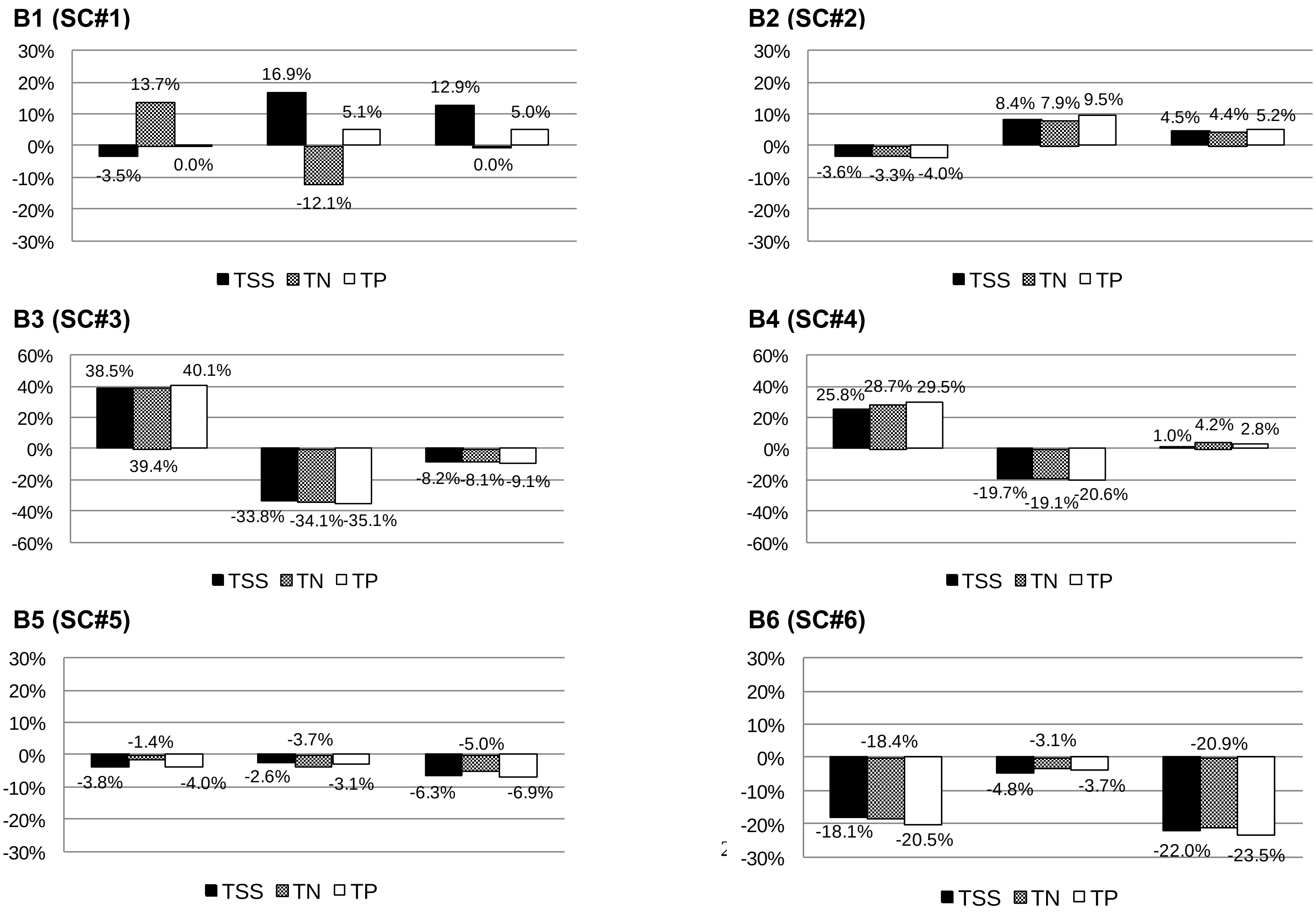
Sub-catchment #	Dryland cropping			Sugarcane			Horticulture			Irrigated cropping		
	1992/93	2001/02	2005/06	1992/93	2001/02	2005/06	1992/93	2001/02	2005/06	1992/93	2001/02	2005/06
B1 (SC#1)	3,823	780	412	18,302	26,628	17,546	2,803	2,506	2,653	-	113	1,169
B2 (SC#2)	961	447	1,597	-	-	-	204	113	200	-	-	587
B3 (SC#3)	2,596	10,199	300	-	-	-	-	332	-	1,249	1,361	1,144
B4 (SC#4)	11,944	26,610	8,926	-	-	-	-	558	300	2,700	4,832	5,032
B5 (SC#5)	2,455	807	681	-	-	-	1,112	1,513	1,172	110	224	-
B6 (SC#6)	4,621	409	183	-	-	-	873	222	200	530	178	29
B7 (SC#7)	18,059	11,316	17,580	-	-	-	-	663	99	627	1,149	1,393
B8 (SC#8)	23,136	14,251	4,629	-	-	-	-	222	-	1,000	441	298
B9 (SC#9 and 10)	10	110	25	-	-	-	110	222	-	-	-	-
B10 (SC#11)	410	13,811	-	-	-	-	-	224	99	-	110	-
B11 (SC#12)	517	4,576	-	-	-	-	-	201	-	-	-	-
B12 (SC#13)	210	11,446	-	-	-	-	100	-	-	-	-	-
B13 (SC#14)	752	15,232	-	-	-	-	-	21	99	-	-	-
B14 (SC#15)	648	412	-	-	-	-	551	1,047	132	463	221	100
B15 (SC#16)	3,019	1,763	201	-	-	-	-	334	-	816	1,784	-
B16 (SC#17)	-	-	101	-	-	-	-	-	-	-	-	-
B17 (SC#19)	751	502	50	-	-	-	-	-	-	440	-	-
B18 (SC#18)	1,191	2,969	3,050	-	-	-	-	227	-	930	337	398
TOTAL	**75,103**	**115,640**	**37,735**	**18,302**	**26,628**	**17,546**	**5,753**	**8,405**	**4,954**	**8,865**	**10,750**	**10,150**

Table A5.5 Land use areas of each FU and sub-catchment on 1992/93, 2001/02 and 2005/06 FU maps (Figure 5.2) (3/3)

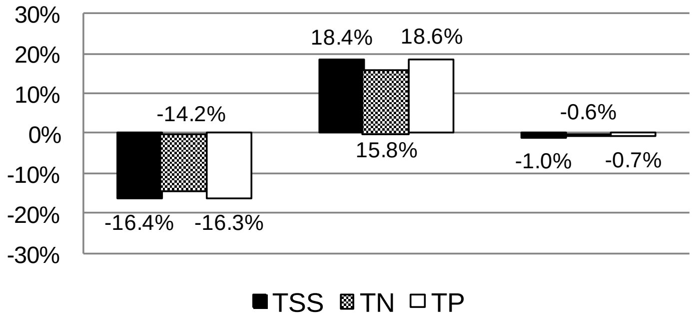
(Unit: ha)

Sub-catchment #	**Urban**			**Water**			**Other**			**TOTAL**		
	1992/93	**2001/02**	**2005/06**	**1992/93**	**2001/02**	**2005/06**	**1992/93**	**2001/02**	**2005/06**	**1992/93**	**2001/02**	**2005/06**
B1 (SC#1)	5,685	5,249	6,785	544	1,476	1,443	1,965	1,458	910	372,524	372,524	372,524
B2 (SC#2)	-	-	-	-	62	65	121	-	-	80,887	80,887	80,887
B3 (SC#3)	526	867	805	110	185	285	200	441	11	149,328	149,328	149,328
B4 (SC#4)	6,183	6,174	6,348	2,452	3,177	2,873	2,164	2,426	2,911	356,065	356,065	356,065
B5 (SC#5)	520	446	599	-	291	309	210	557	-	181,845	181,845	181,845
B6 (SC#6)	-	-	-	519	681	607	341	156	101	156,739	156,739	156,739
B7 (SC#7)	5,562	5,113	5,306	415	632	713	129	554	701	173,107	173,107	173,107
B8 (SC#8)	1,247	1,342	1,593	578	697	650	643	266	-	223,801	223,801	223,801
B9 (SC#9 and 10)	-	-	-	-	2	7	-	-	-	31,936	31,936	31,936
B10 (SC#11)	-	-	-	-	7	6	-	-	-	178,874	178,874	178,874
B11 (SC#12)	-	-	-	-	6	20	-	112	-	181,441	181,441	181,441
B12 (SC#13)	-	-	-	-	6	7	-	-	-	135,931	135,931	135,931
B13 (SC#14)	-	-	-	-	7	15	100	111	-	207,388	207,388	207,388
B14 (SC#15)	1,481	1,564	1,503	-	2	20	431	350	100	213,498	213,498	213,498
B15 (SC#16)	-	-	-	1,181	1,258	1,128	110	110	-	250,498	250,498	250,498
B16 (SC#17)	-	-	-	-	-	15	100	-	-	71,668	71,668	71,668
B17 (SC#19)	-	-	-	-	4	12	220	-	-	161,467	161,467	161,467
B18 (SC#18)	-	-	-	-	66	126	100	335	100	159,040	159,040	159,040
TOTAL	**21,204**	**20,755**	**22,939**	**5,799**	**8,559**	**8,301**	**6,834**	**6,876**	**4,834**	**3,286,037**	**3,286,037**	**3,286,037**

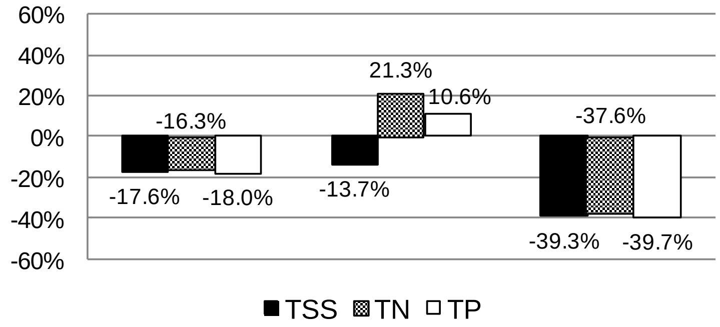
Figure A.5.1 Percentage change in average annual sub-catchment load over three time periods (1992/93-2001/02, 2001/02-2005/06 and 1992/93-2005/06) for Total Suspended Solids (TSS), Total Nitrogen (TN) and Total Phosphorus (TP).



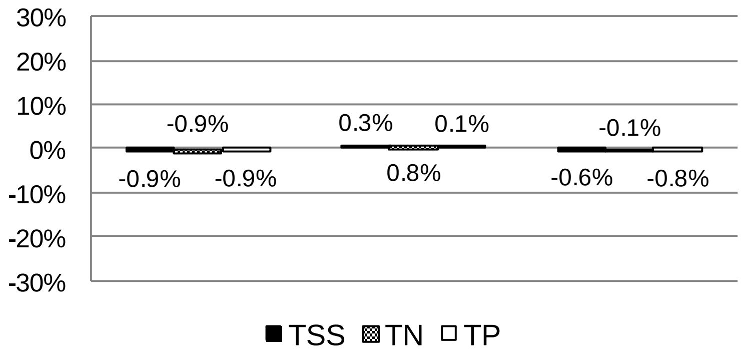
B7 (SC#7)



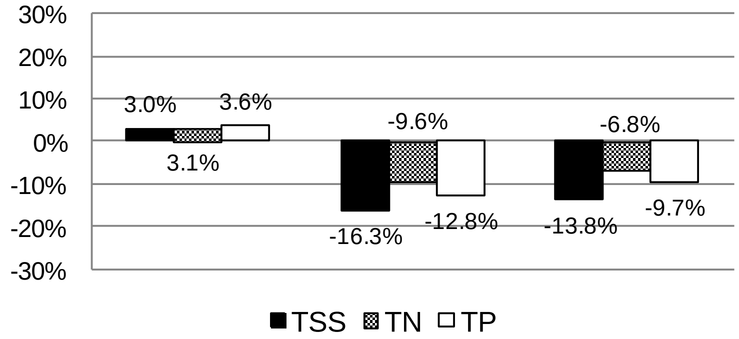
B8 (SC#8)



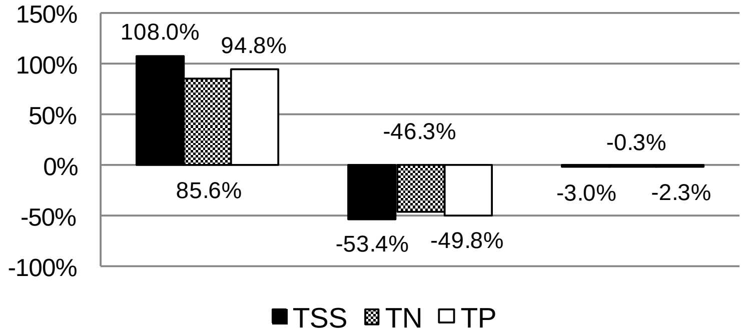
B9 (1) (SC#9)



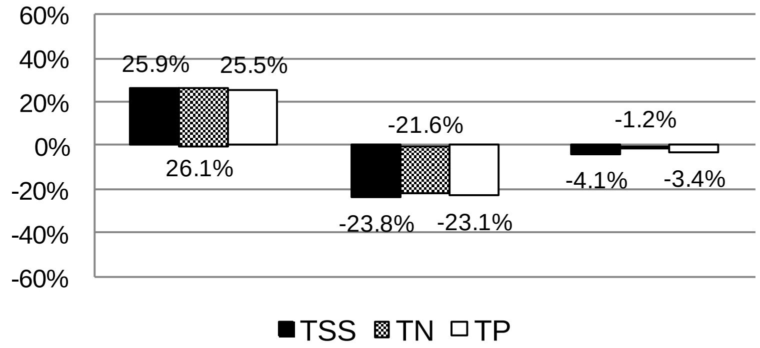
B9 (2) (SC#10)



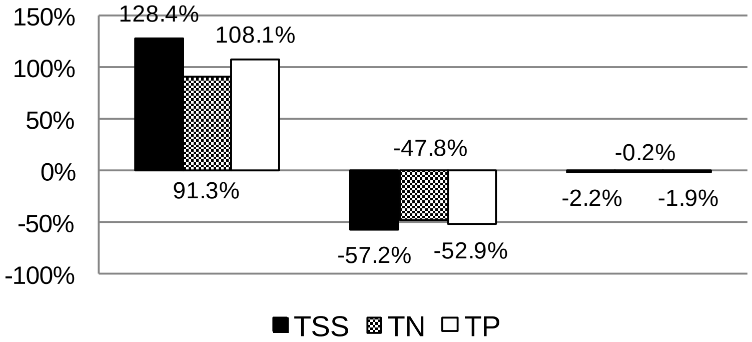
B10 (SC#11)



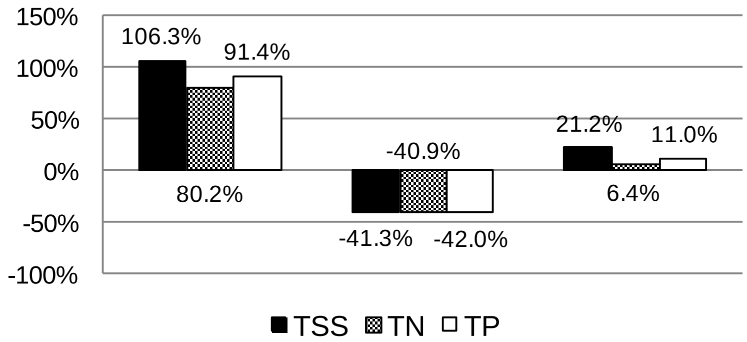
B11 (SC#12)



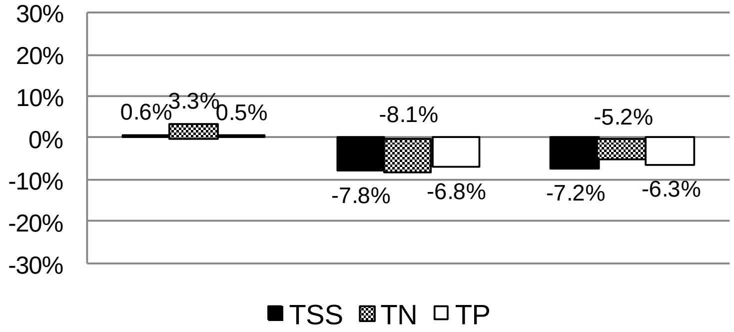
B12 (SC#13)



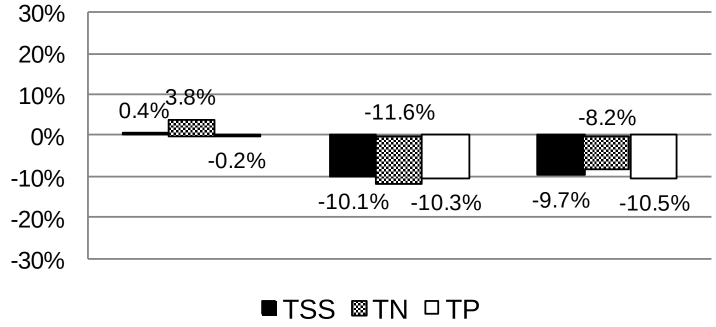
B13 (SC#14)



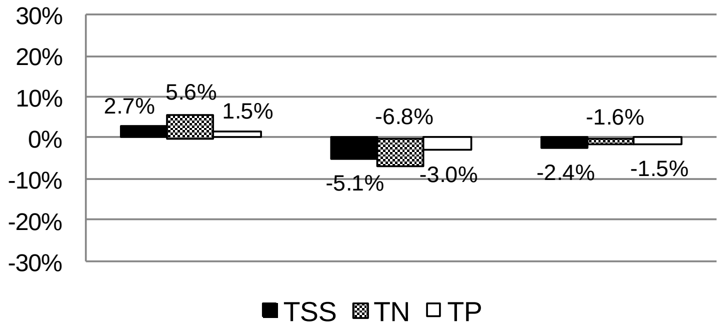
B14 (SC#15)



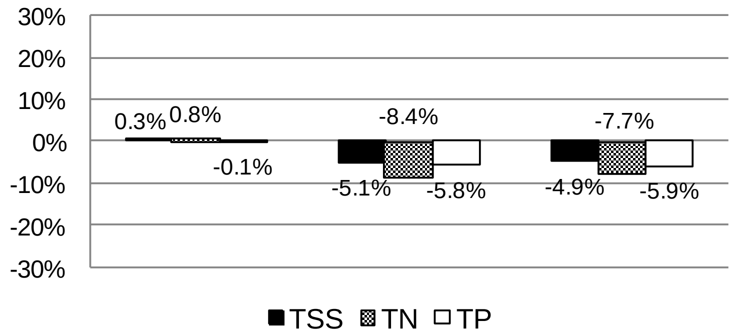
B15 (SC#16)



B16 (SC#17)



B17 (SC#19)



B18 (SC#18)

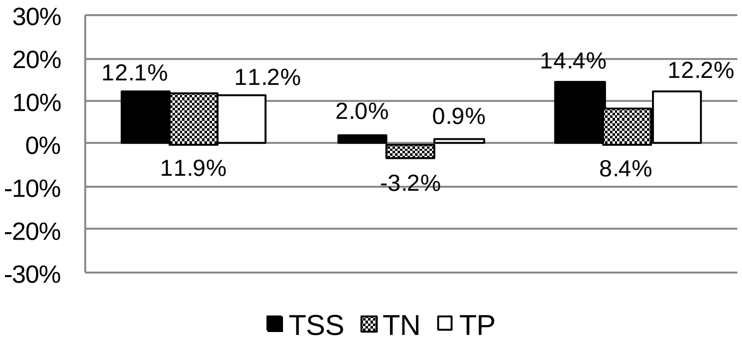
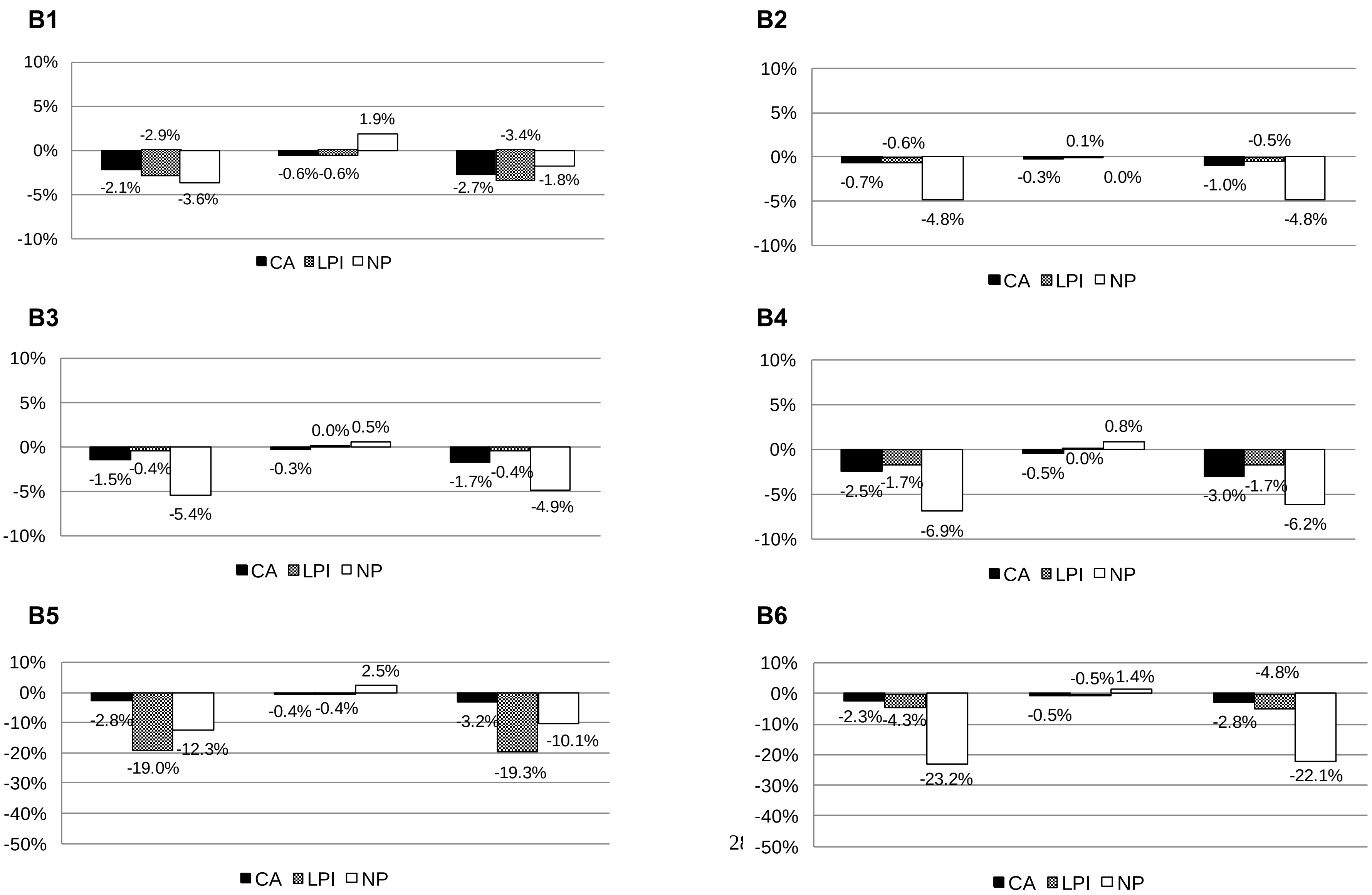
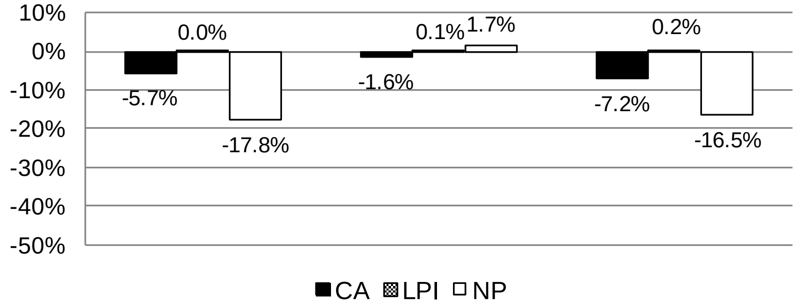


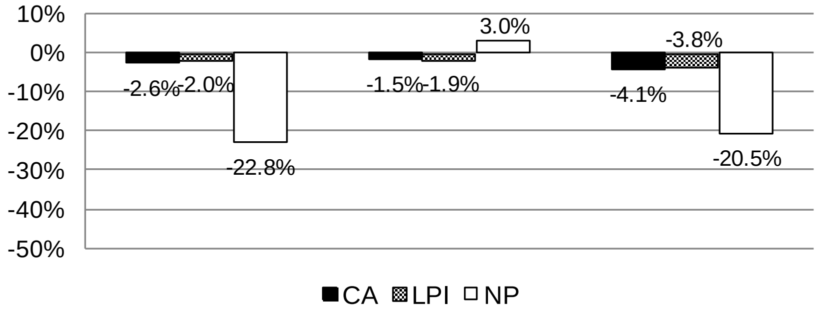
Figure A.5.2 Percentage changes in class metrics over three time periods (1991-2001, 2001-2005 and 1991-2005) for Class Area (CA), Largest Patch Index (LPI) and number of patches (NP) (%).



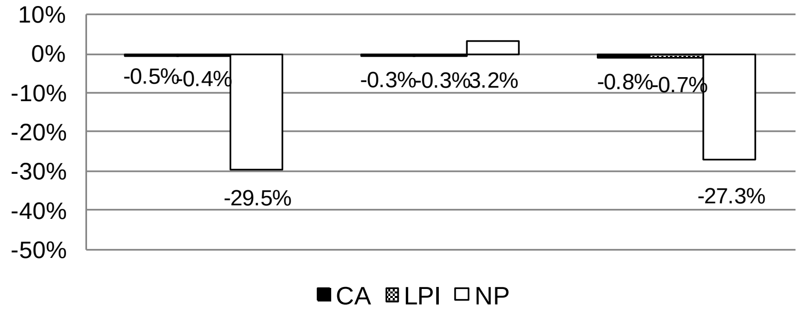
B7



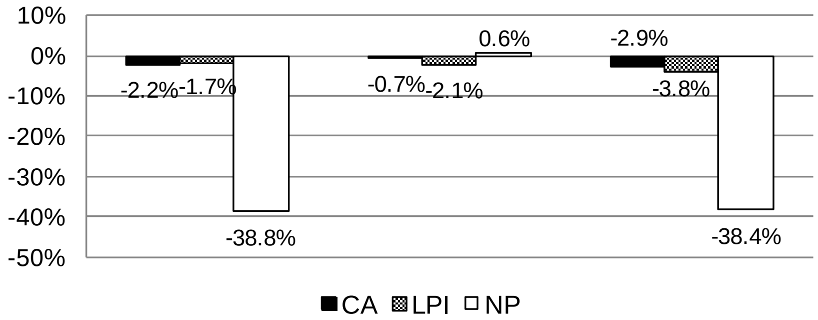
B8



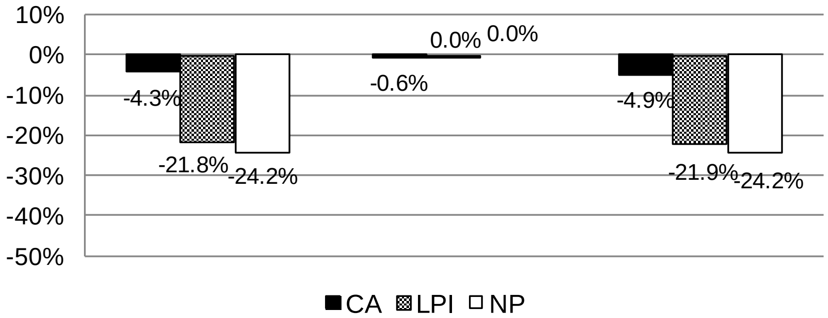
B9



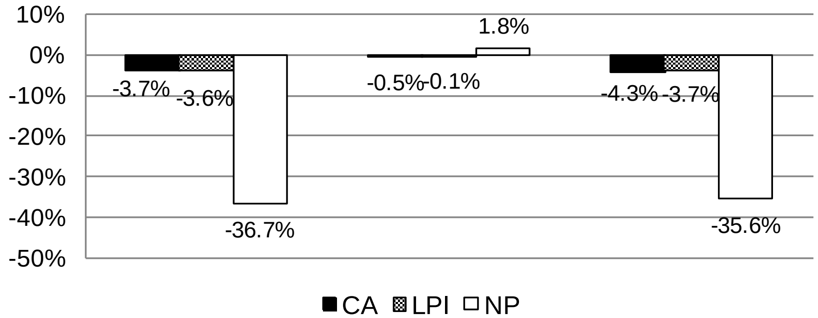
B10



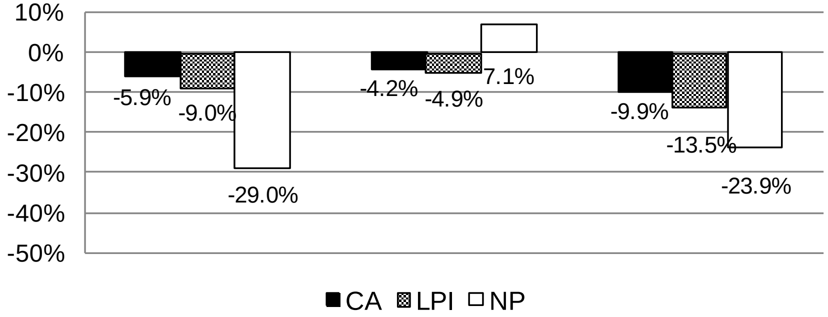
B11



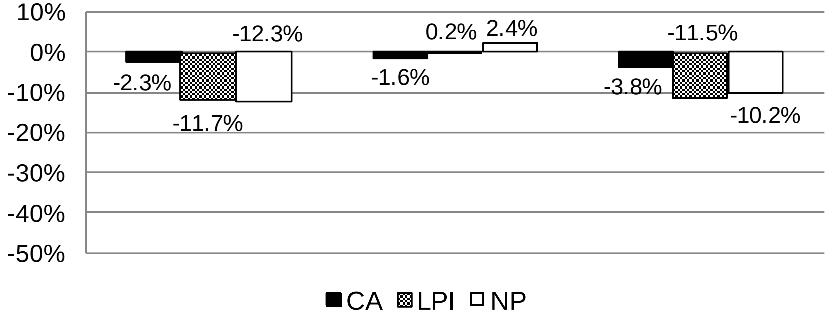
B12



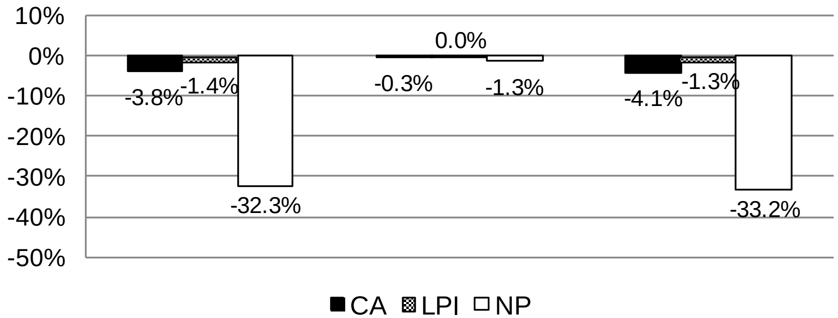
B13



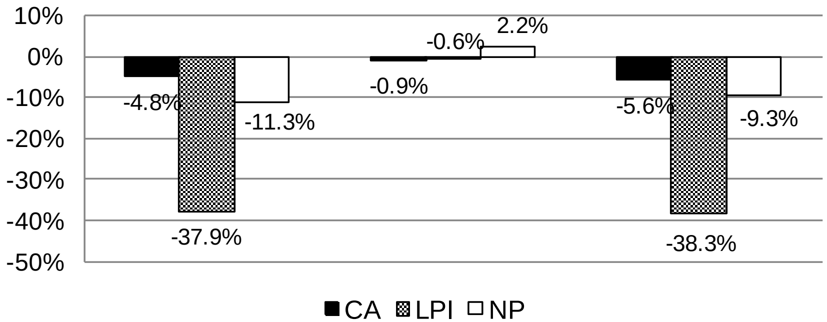
B14



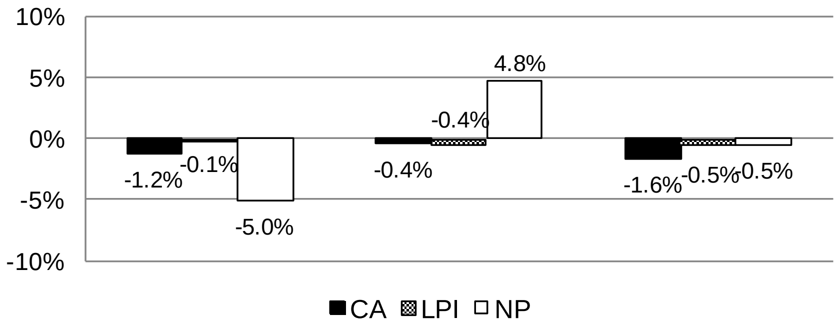
B15



B16



B17



B18

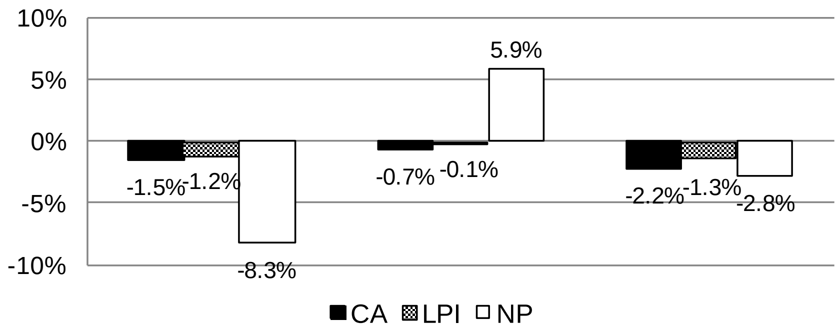


Table A5.6 Actual habitat amount of the Burnett River catchment (ha).

Native vegetation/FU categories		1991		2001		2005	
		BCV_{LU} (low)	BCV_{LU} (high)	BCV_{LU} (low)	BCV_{LU} (high)	BCV_{LU} (low)	BCV_{LU} (high)
1	Wet forest	59,610	74,413	56,540	70,675	56,330	70,413
2	Dry eucalypt forest	862,987	1,232,839	838,956	1,198,508	829,936	1,185,623
3	Brigalow	4,985	7,122	3,927	5,610	3,812	5,447
4	Non-eucalypt woodland	5,528	7,898	5,383	7,691	5,381	7,687
5	Grassland	-	-	296	422	294	420
6	Coastal communities and wetlands	6,570	9,386	7,223	10,318	7,202	10,288
7	Other conservation	43,937	70,300	44,074	70,519	27,026	43,242
8	Other plantation forestry	11,099	22,198	11,136	22,271	10,185	20,370
9	Grazing forested	203,211	474,160	155,308	362,385	200,752	468,422
10	Grazing open	105,010	210,020	119,364	238,729	117,656	235,313
11	Dryland cropping	3,431	10,294	5,508	16,523	1,853	5,558
12	Sugarcane	1,004	3,010	1,334	4,003	1,005	3,014
13	Horticulture	268	803	368	1,102	238	715
14	Irrigated cropping	397	1,984	505	2,527	489	2,447
15	Urban	1,605	4,013	1,593	3,982	1,843	4,609
17	Other	314	941	335	1,006	225	675
	TOTAL	**1,309,958**	**2,129,483**	**1,251,850**	**2,016,273**	**1,264,230**	**2,064,243**

Table A5.7 Percentage changes in habitat amount of the Burnett River sub-catchments (%).

	1991-2001		2001-05		1991-2005	
	BCV_{LU} (low)	BCV_{LU} (high)	BCV_{LU} (low)	BCV_{LU} (high)	BCV_{LU} (low)	BCV_{LU} (high)
B1	-2.9	-3.6	-3.5	-1.1	-6.2	-4.6
B2	-0.6	-0.9	-0.2	-0.3	-0.8	-0.6
B3	-5.4	-7.0	2.3	4.0	-3.2	-3.3
B4	-5.4	-6.5	1.6	3.6	-3.9	-3.2
B5	-5.2	-6.6	1.8	2.6	-3.4	-4.2
B6	-1.5	-1.6	0.6	0.6	-0.9	-1.0
B7	-4.4	-5.2	2.2	4.5	-2.3	-1.0
B8	-3.9	-5.1	4.5	6.1	0.5	0.6
B9	-0.4	-0.3	0.6	0.5	0.2	0.2
B10	-4.2	-4.7	0.8	1.5	-3.4	-3.2
B11	-5.9	-6.8	2.0	2.4	-4.0	-4.6
B12	-4.7	-4.7	0.9	0.8	-3.9	-4.0
B13	-6.1	-6.0	-1.6	-0.3	-7.6	-6.3
B14	-4.3	-5.4	2.4	3.7	-2.0	-1.9
B15	-8.8	-11.3	3.9	5.9	-5.2	-6.0
B16	-6.1	-7.1	2.4	3.9	3.9	-3.5
B17	-2.5	-3.1	1.7	2.9	-0.9	-0.3
B18	-3.3	-3.9	0.9	3.0	-2.3	-1.0
Total	-4.4	-5.3	1.0	2.4	-3.5	-3.1

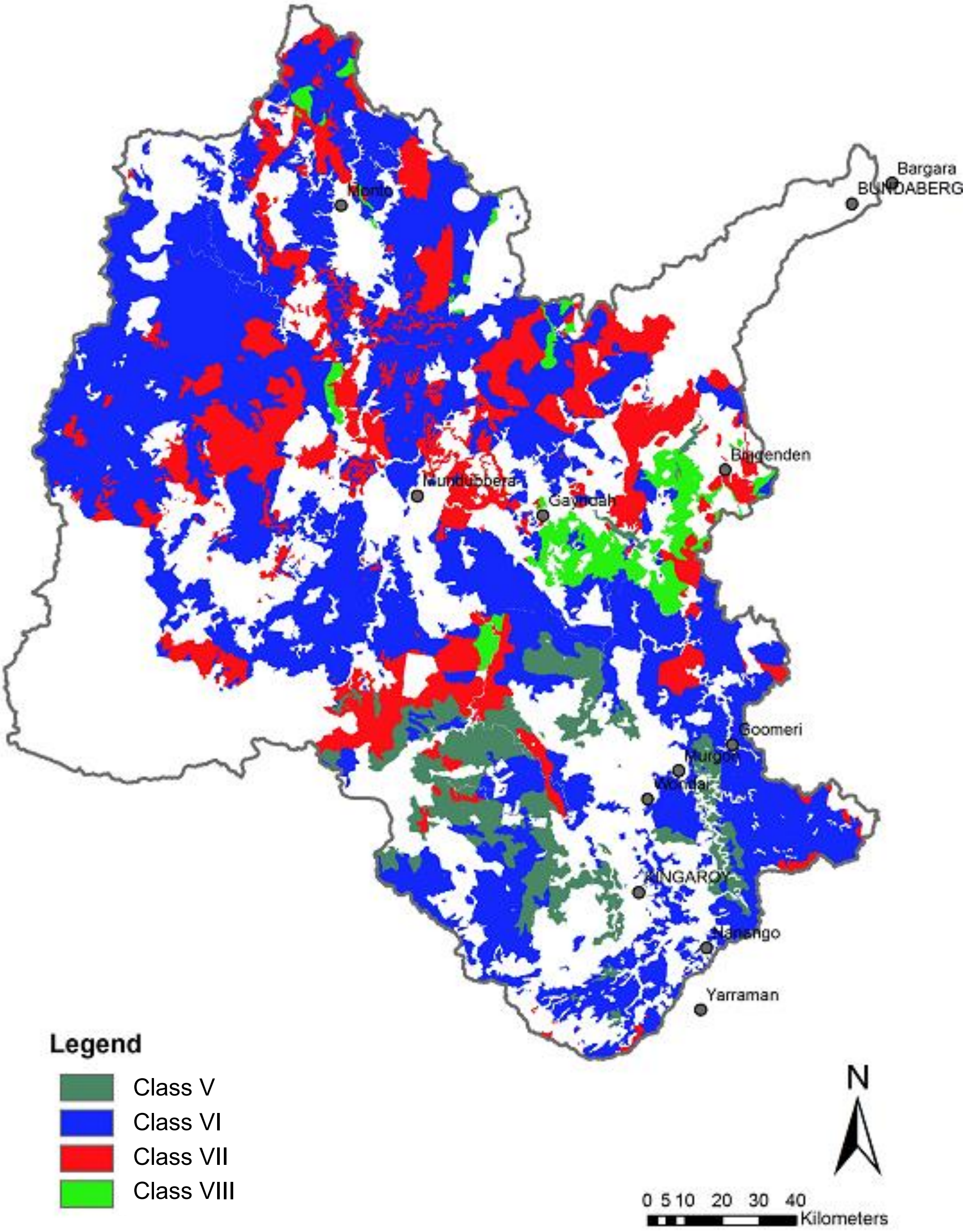


Figure A6.1 Lands in Class V and greater, classified by land capability studies. (source: Donnollan and Searle, 1999; Kent, 2002; Vandersee and Kent; 1983)

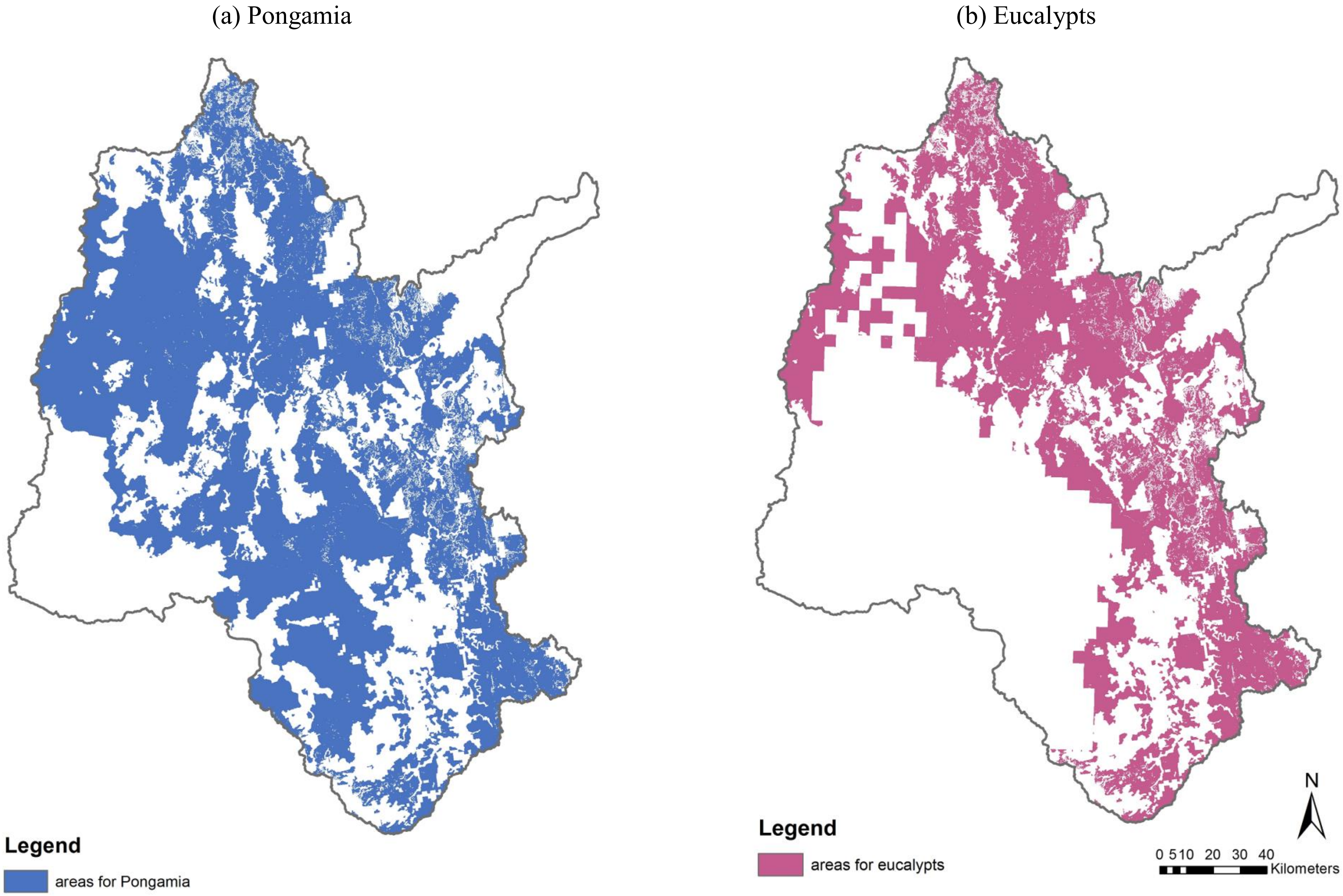


Figure A6.2 'Underutilised agricultural land' available for bioenergy production within the Burnett River catchment.

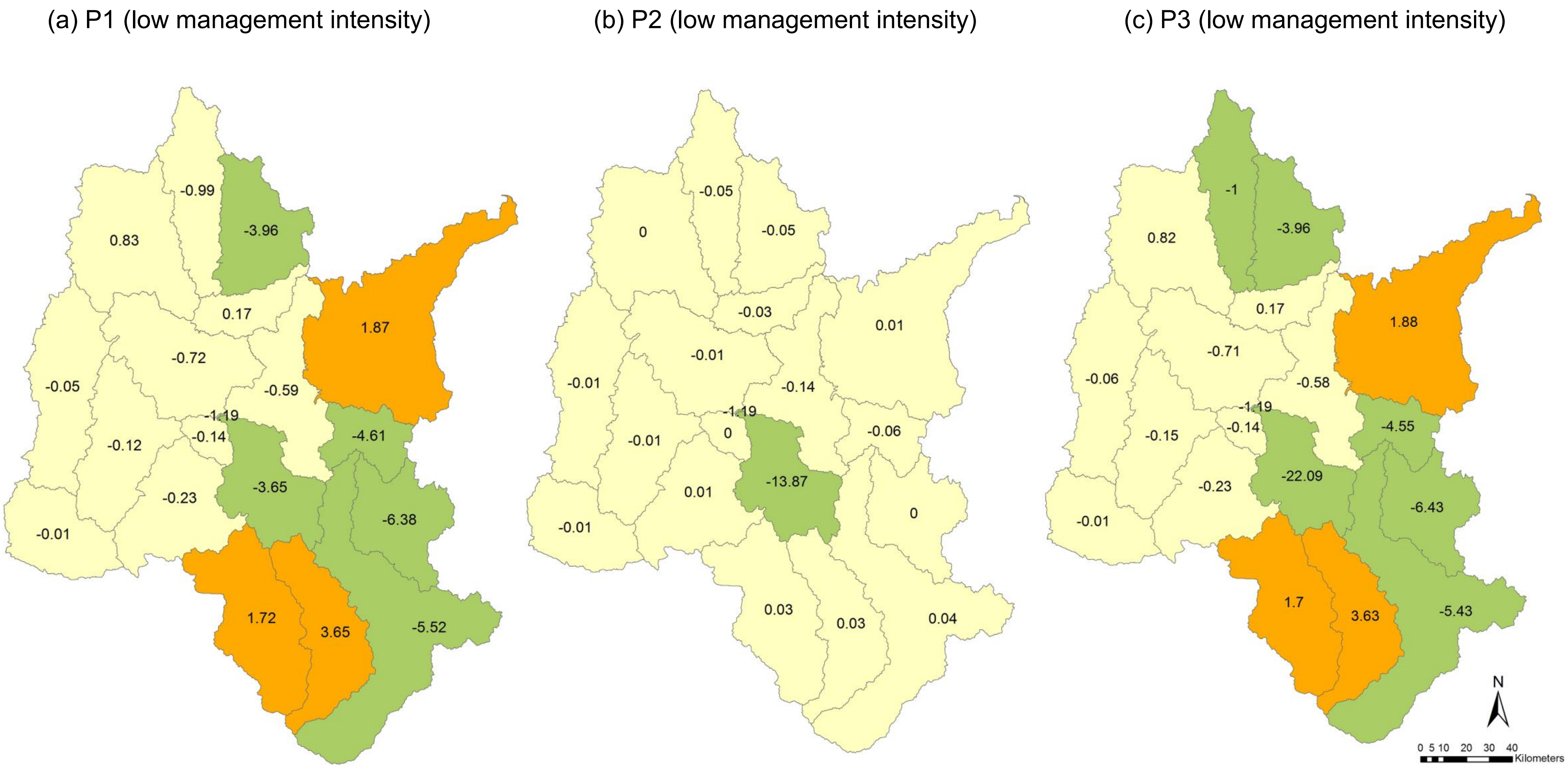


Figure A6.3 [a] Change in estimated average annual runoff-volume by sub-catchment (%): Pongamia (low management intensity).

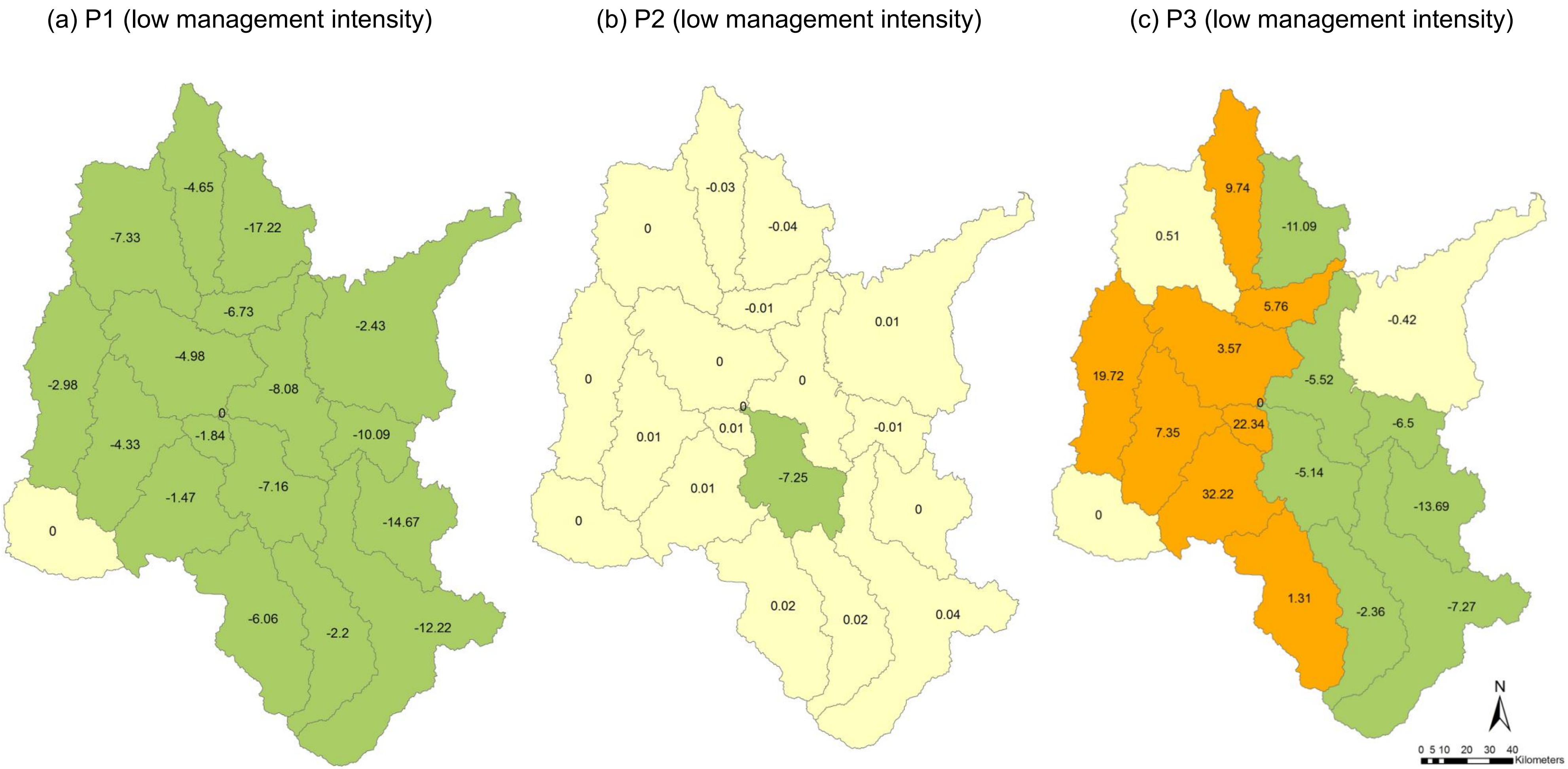


Figure A6.3 [b] Change in estimated average annual TSS load by sub-catchment (%): Pongamia (low management intensity).

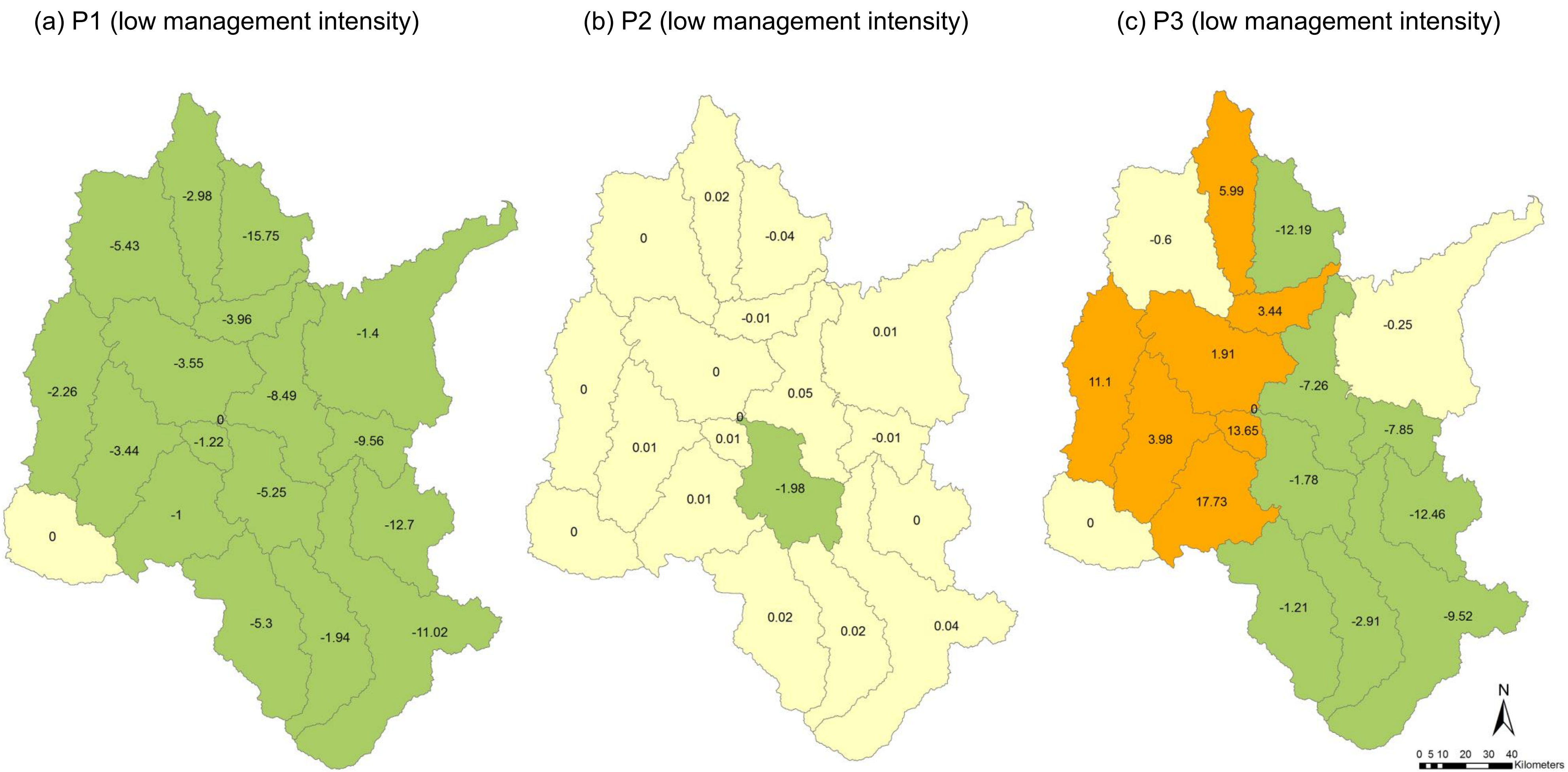


Figure A6.3 [c] Change in estimated average annual TP load by sub-catchment (%): Pongamia (low management intensity).

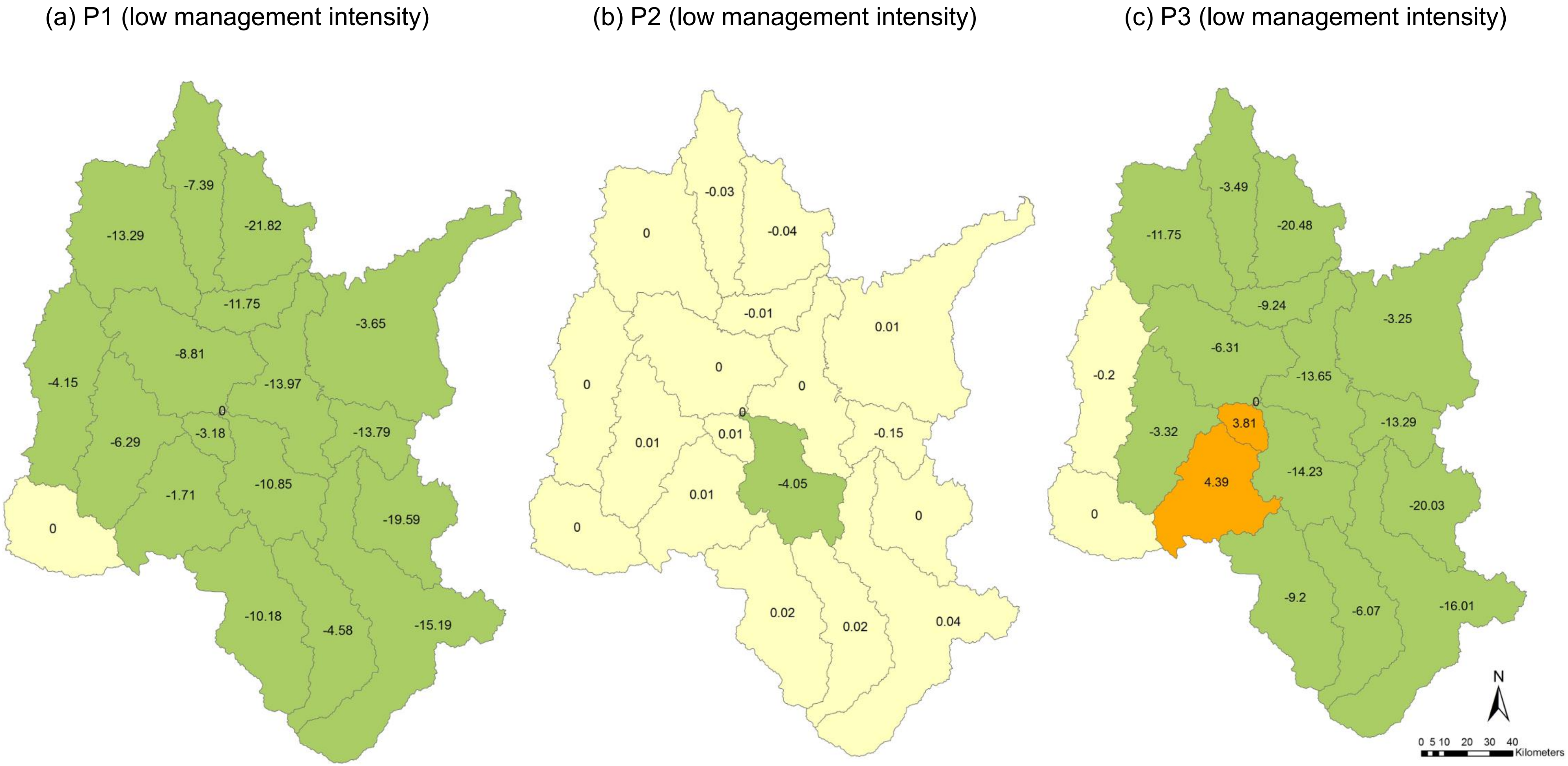


Figure A6.3 [d] Change in estimated average annual TN load by sub-catchment (%): Pongamia (low management intensity).

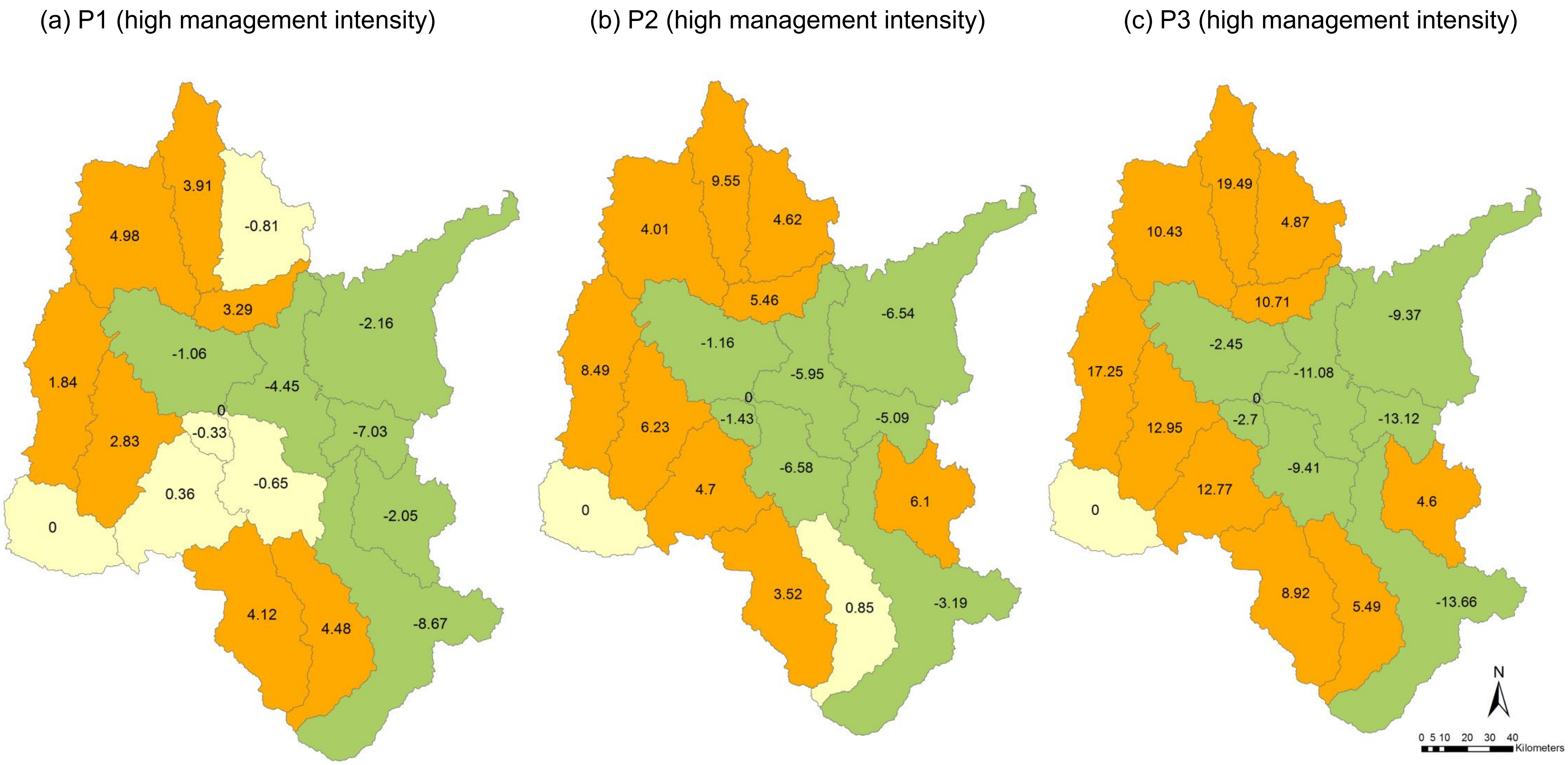


Figure A6.3 [e] Change in estimated average annual runoff-volume by sub-catchment (%): Pongamia (high management intensity).

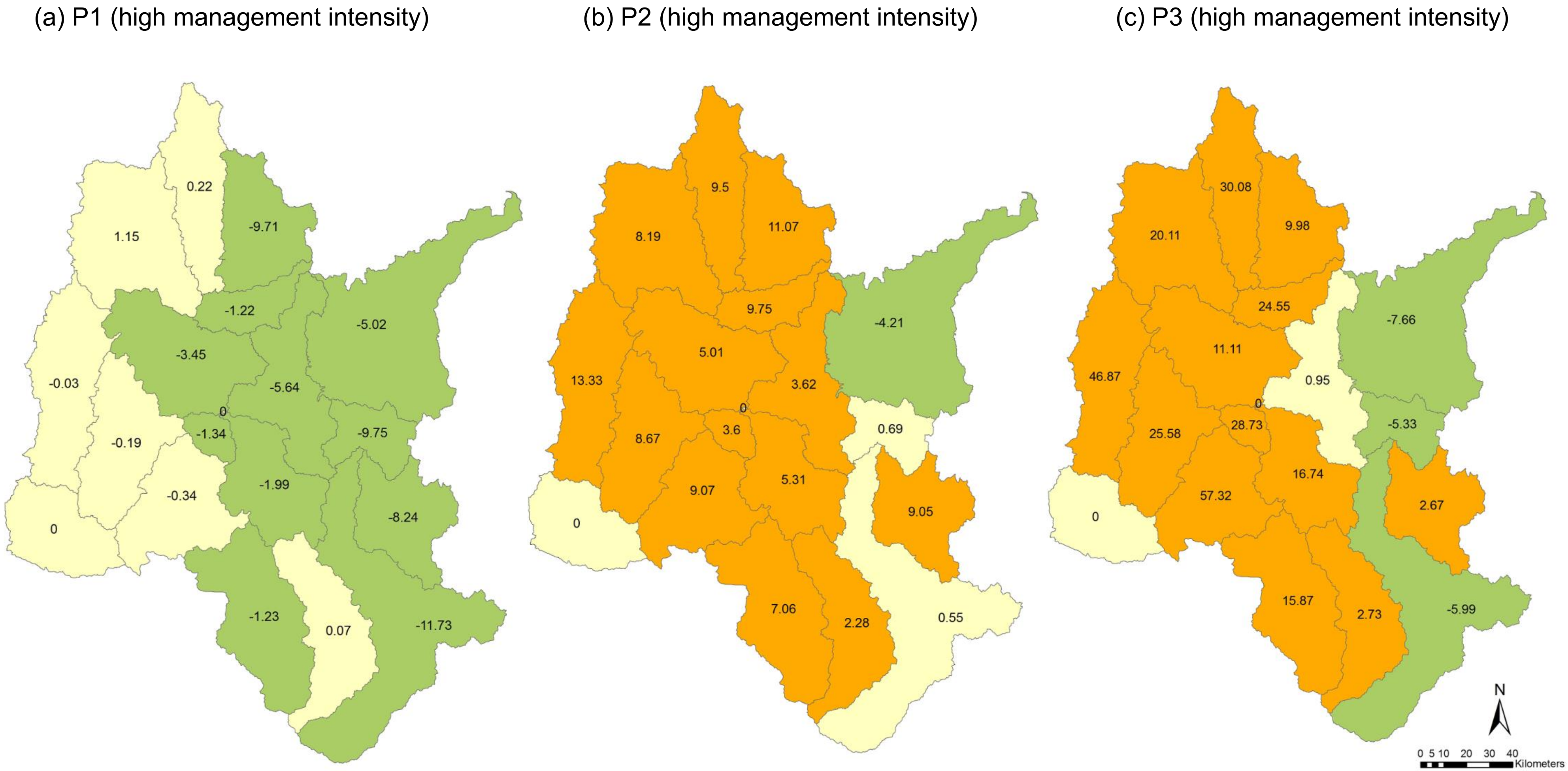


Figure A6.3 [f] Change in estimated average annual TSS load by sub-catchment (%): Pongamia (high management intensity).

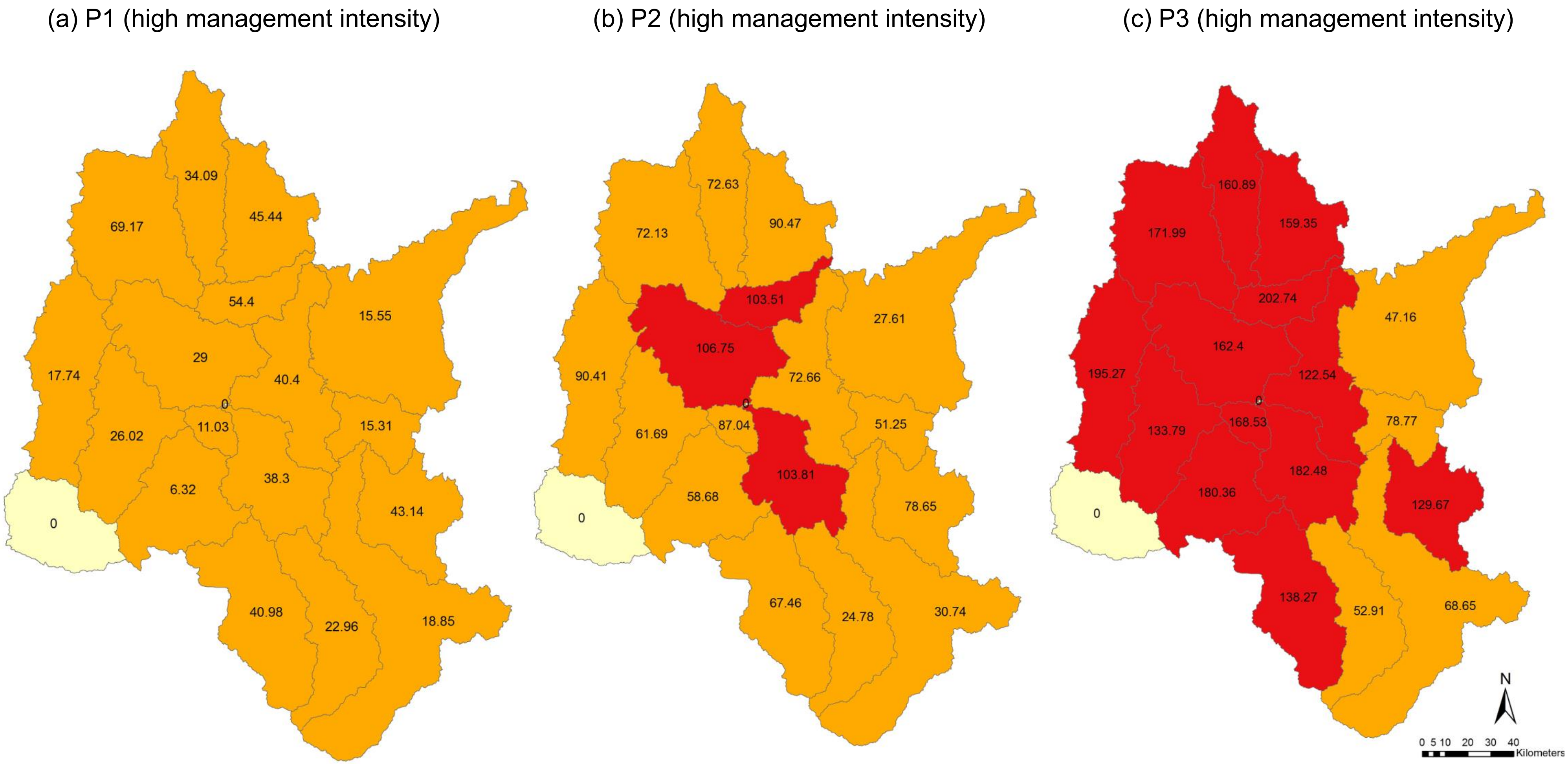


Figure A6.3 [g] Change in estimated average TP load by sub-catchment (%): Pongamia (high management intensity).

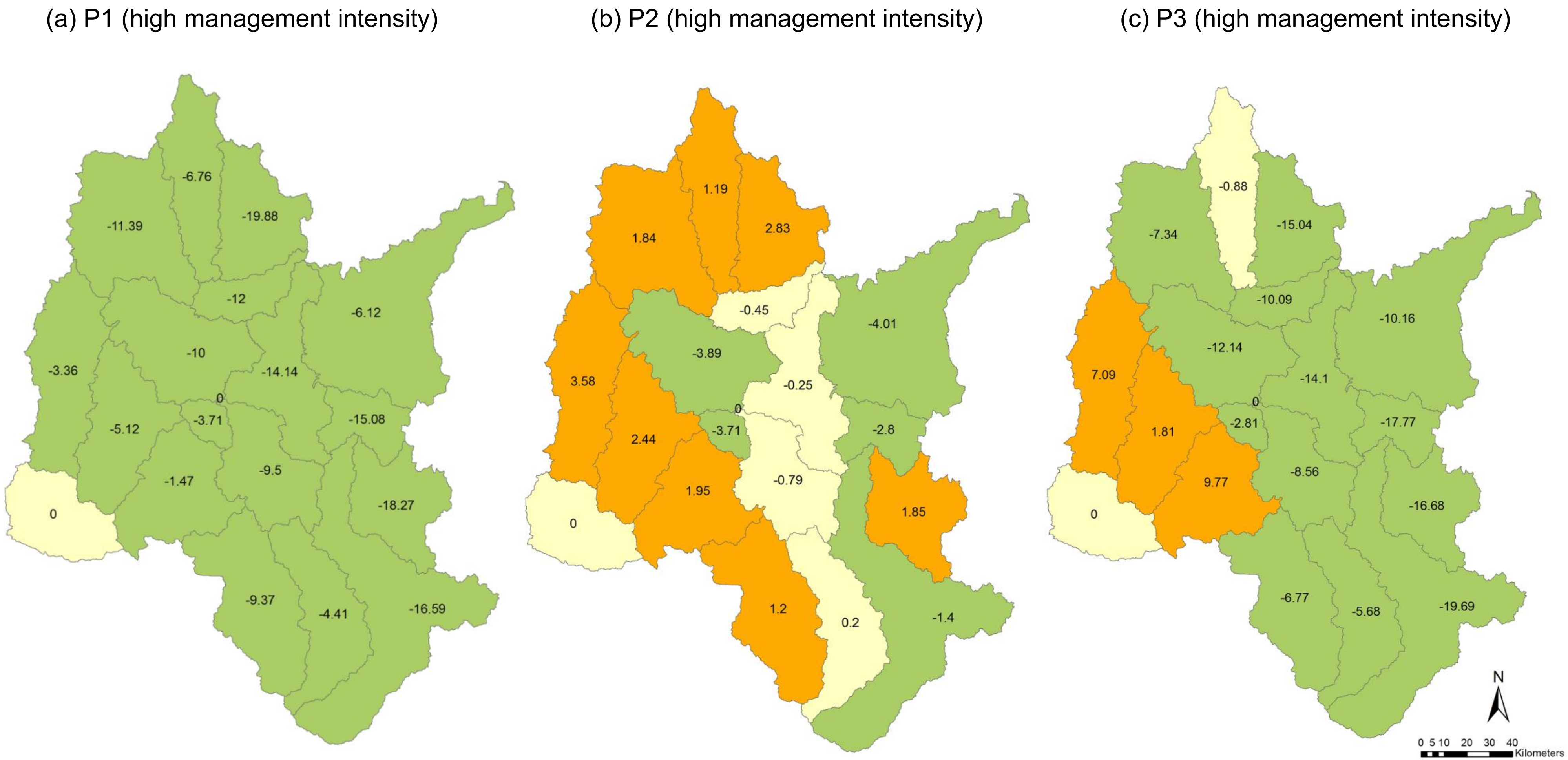


Figure A6.3 [h] Change in estimated average TN load by sub-catchment (%): Pongamia (high management intensity).

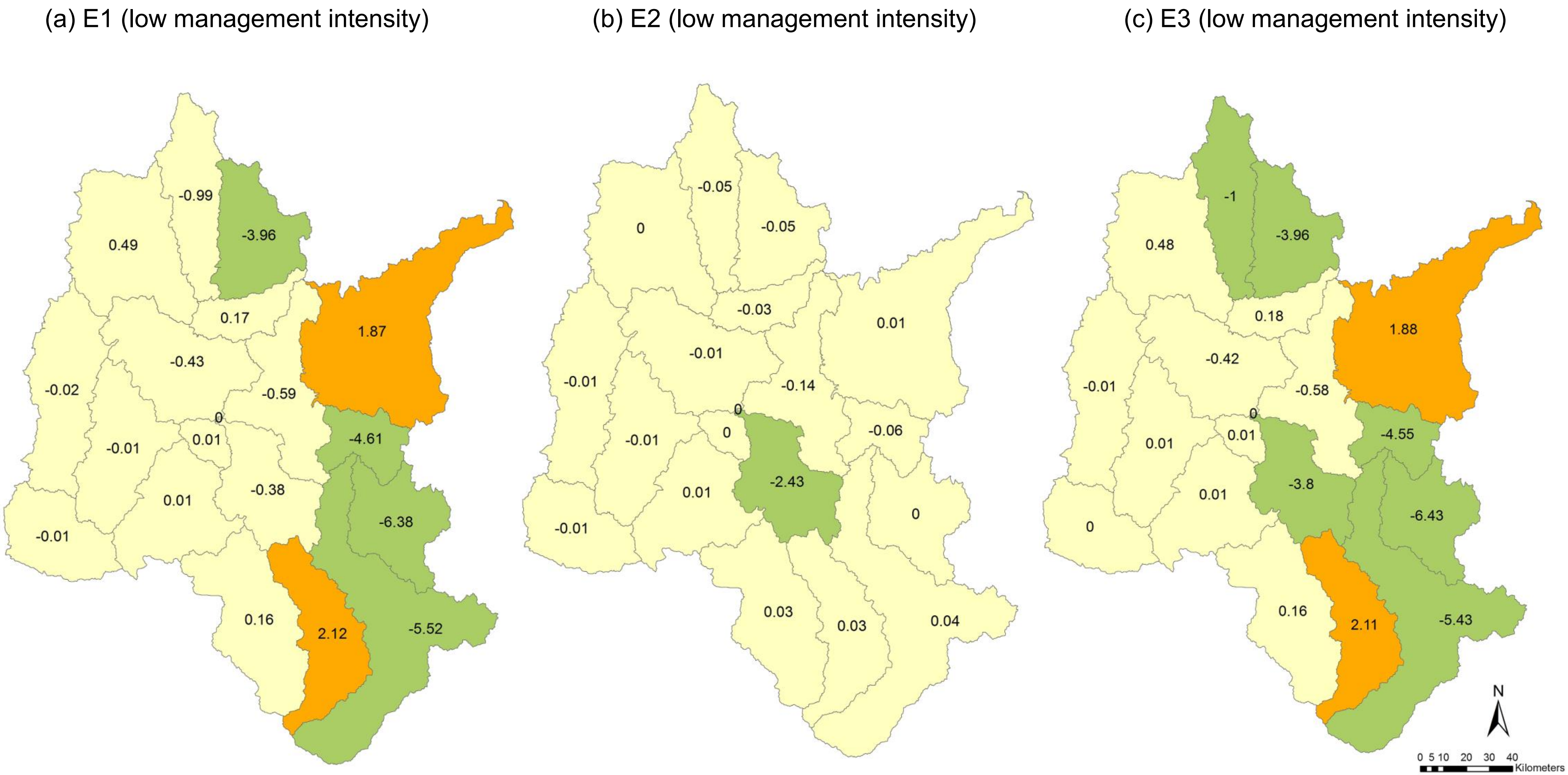


Figure A6.4 [a] Change in estimated average annual runoff-volume by sub-catchment (%): Eucalypts (low management intensity)

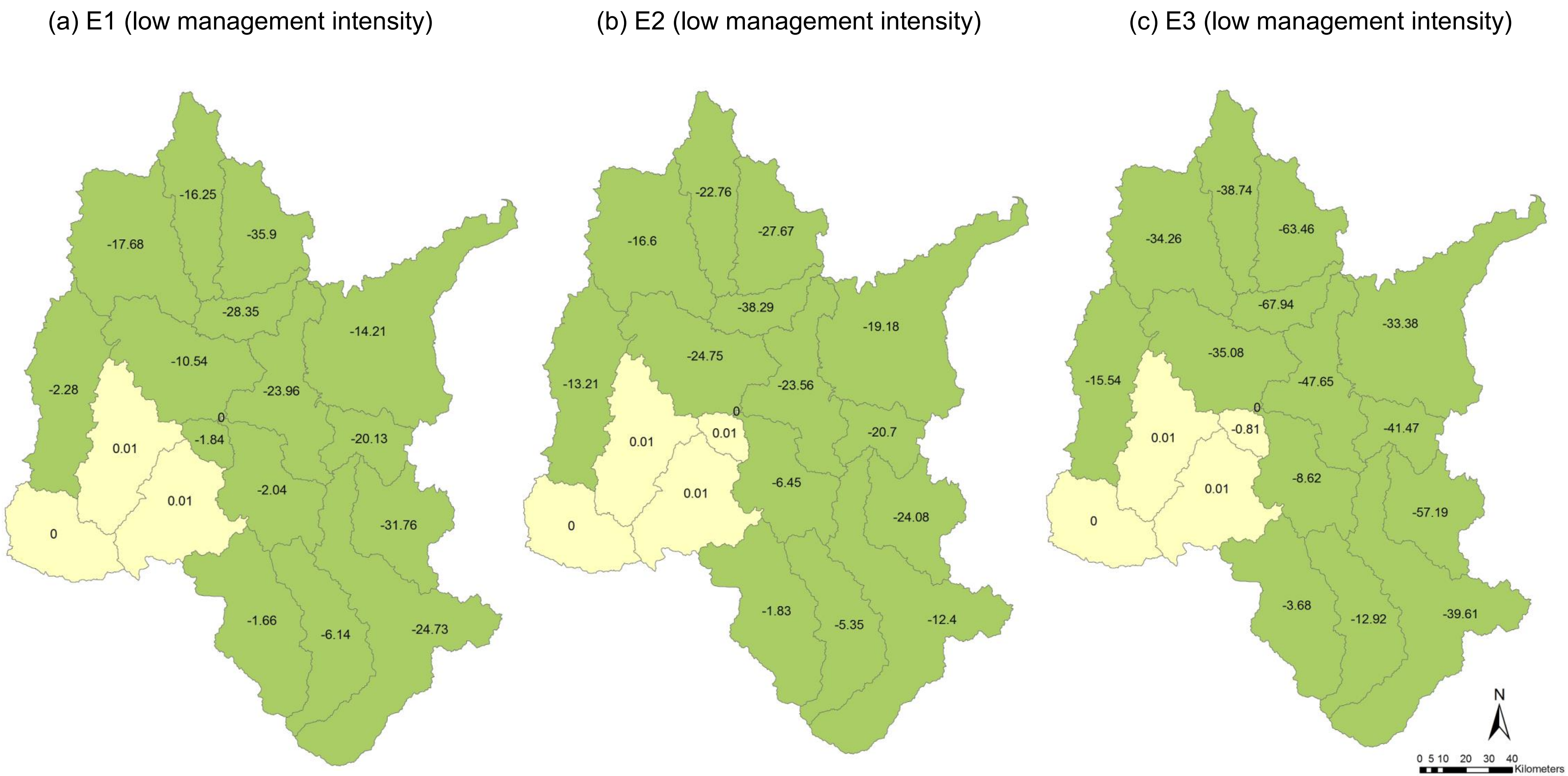


Figure A6.4 [b] Change in estimated average annual TSS load by sub-catchment (%): Eucalypts (low management intensity).

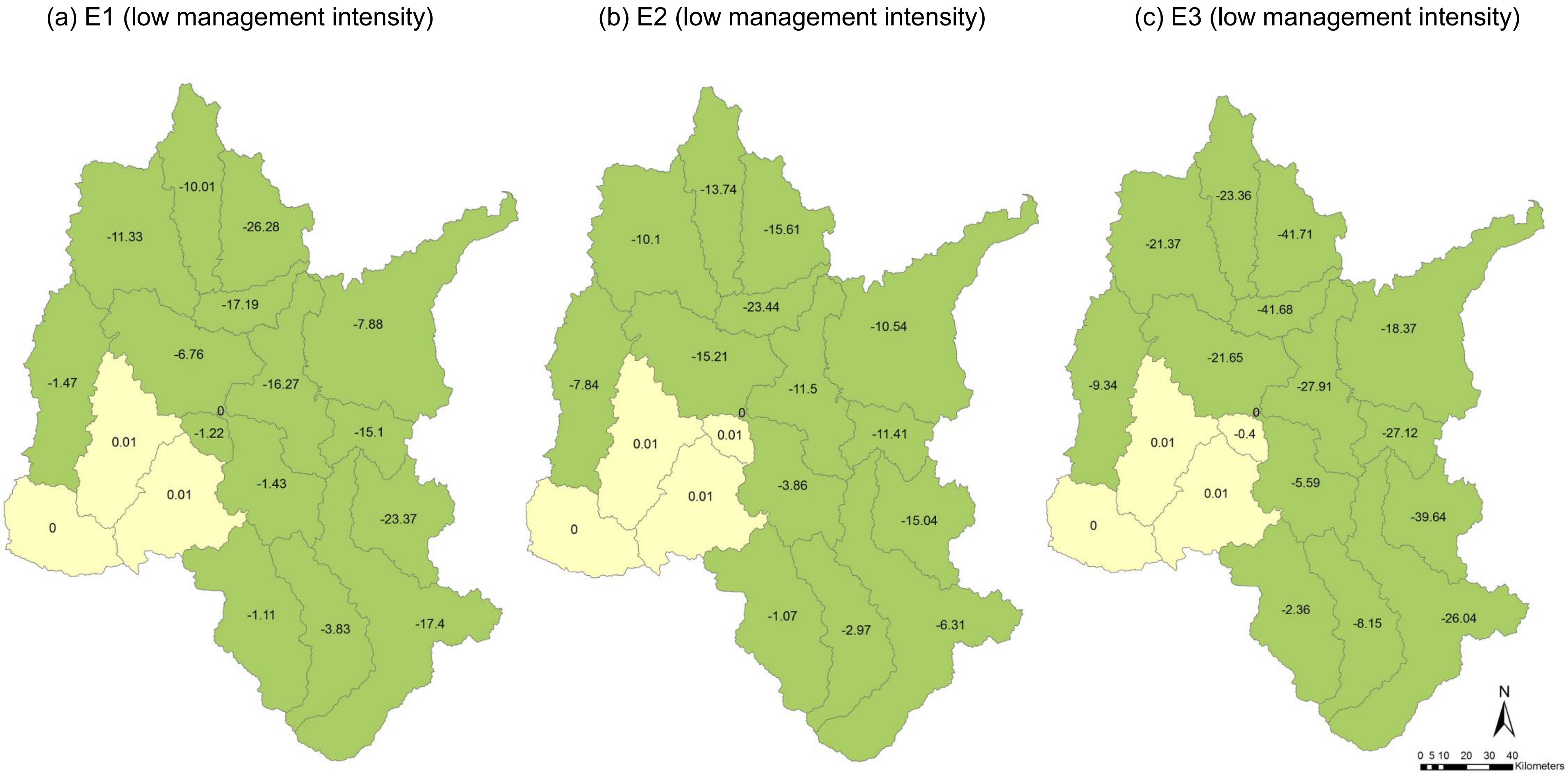


Figure A6.4 [c] Change in estimated average annual TP load by sub-catchment (%): Eucalypts (low management intensity).

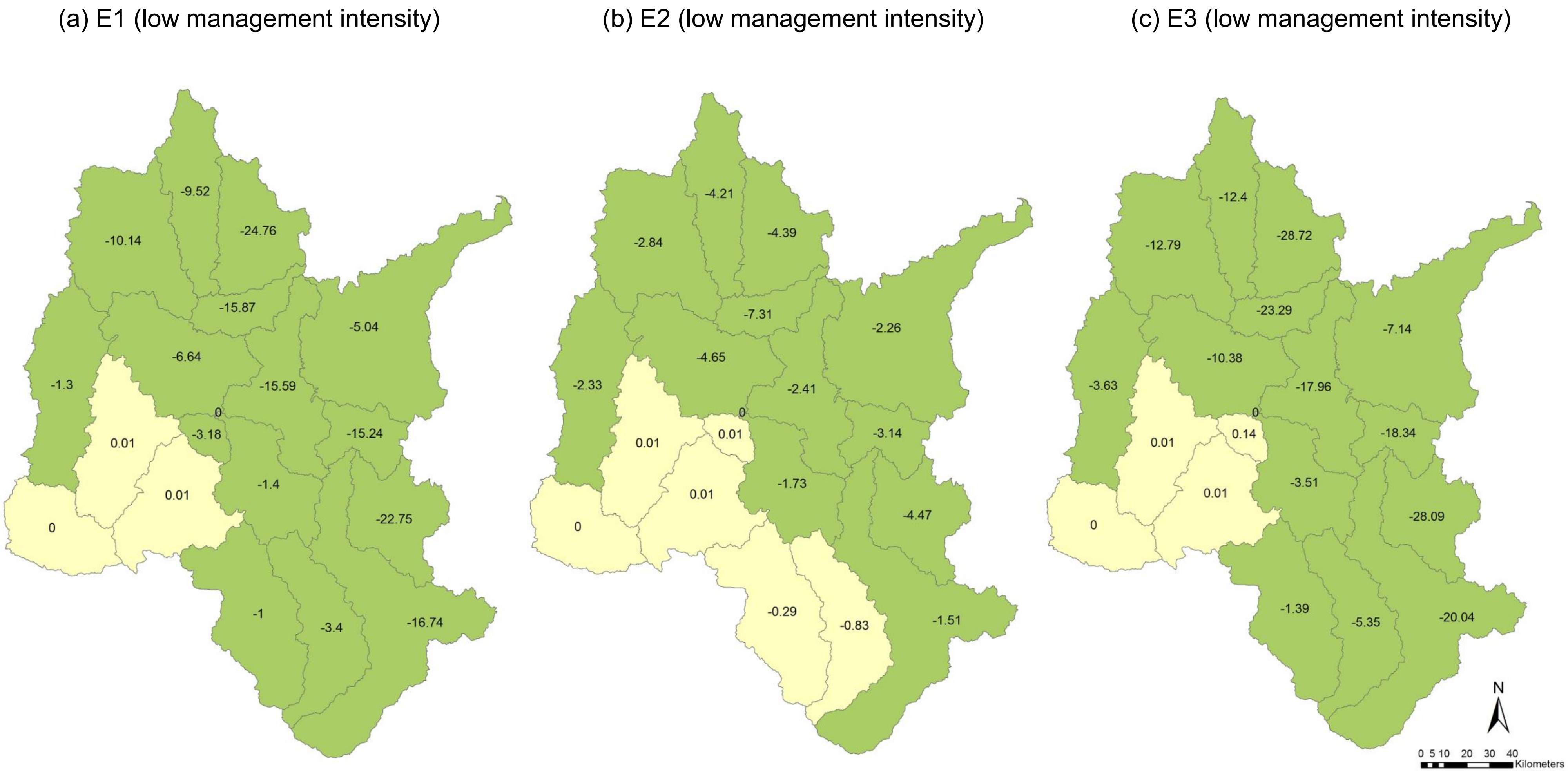


Figure A6.4 [d] Change in estimated average annual TN load by sub-catchment (%): Eucalypts (low management intensity).

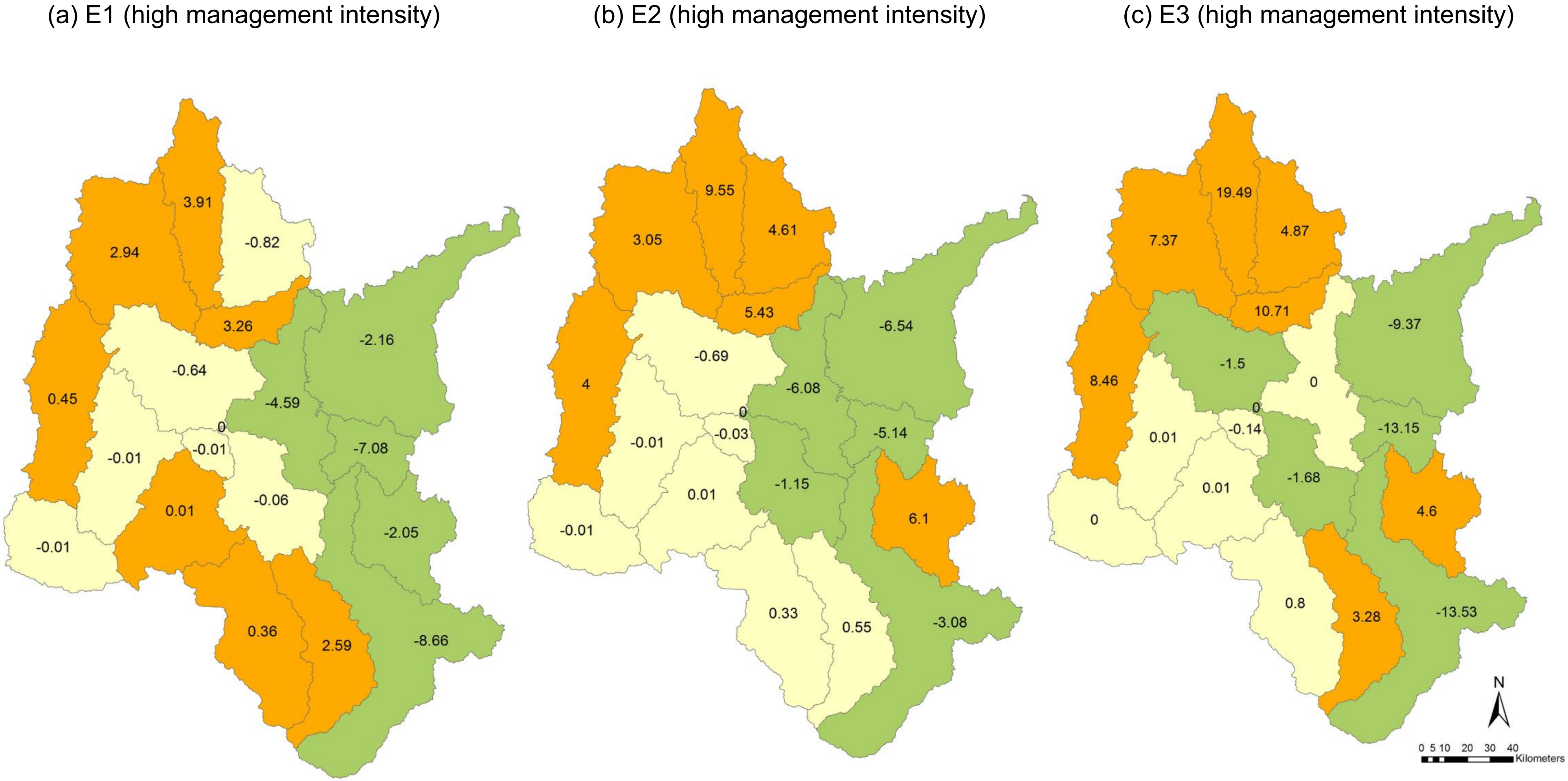


Figure A6.4 [e] Change in estimated average annual runoff-volume by sub-catchment (%): Eucalypts (high management intensity).

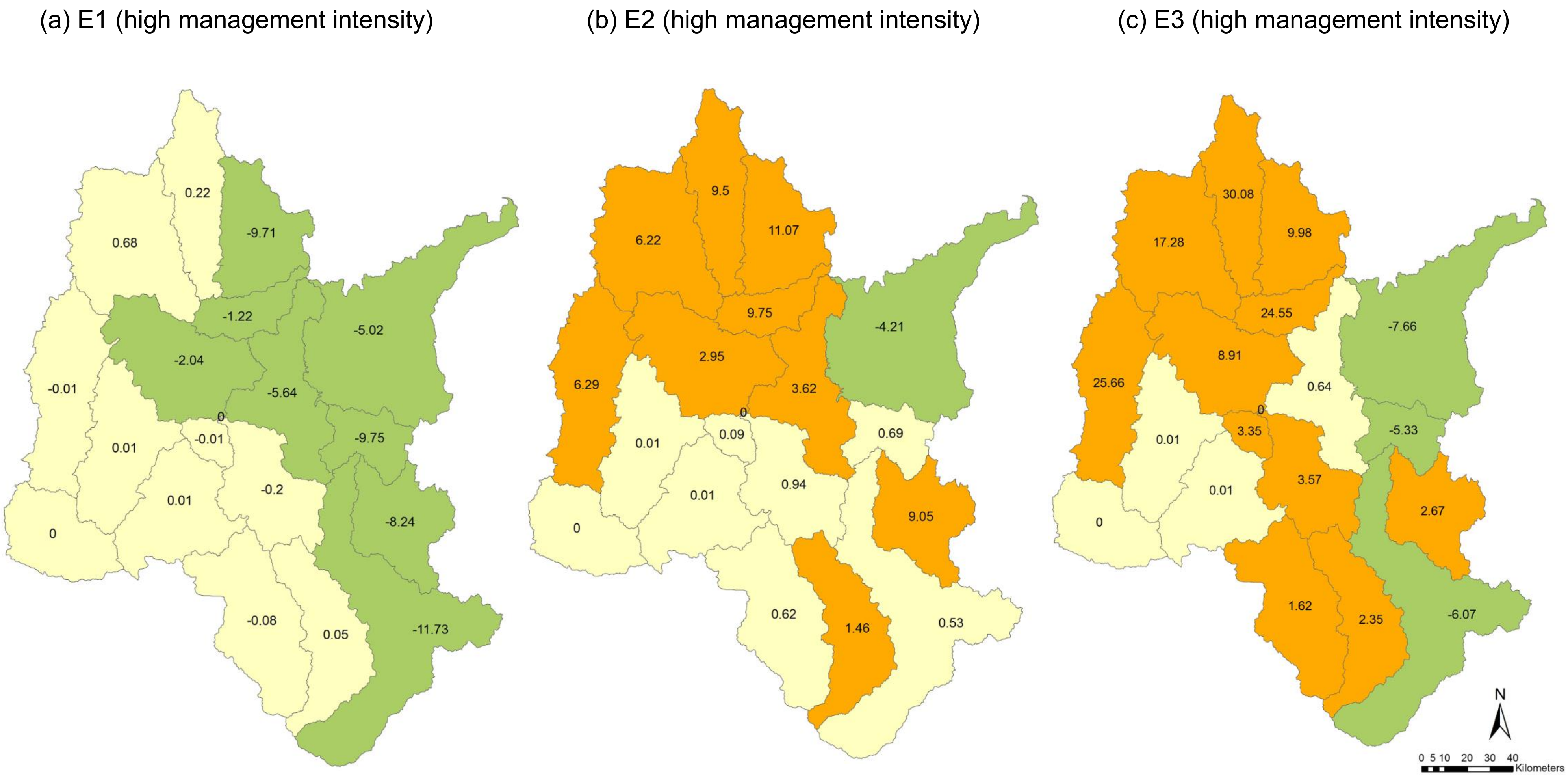


Figure A6.4 [f] Change in estimated average annual TSS load by sub-catchment (%): Eucalypts (high management intensity).

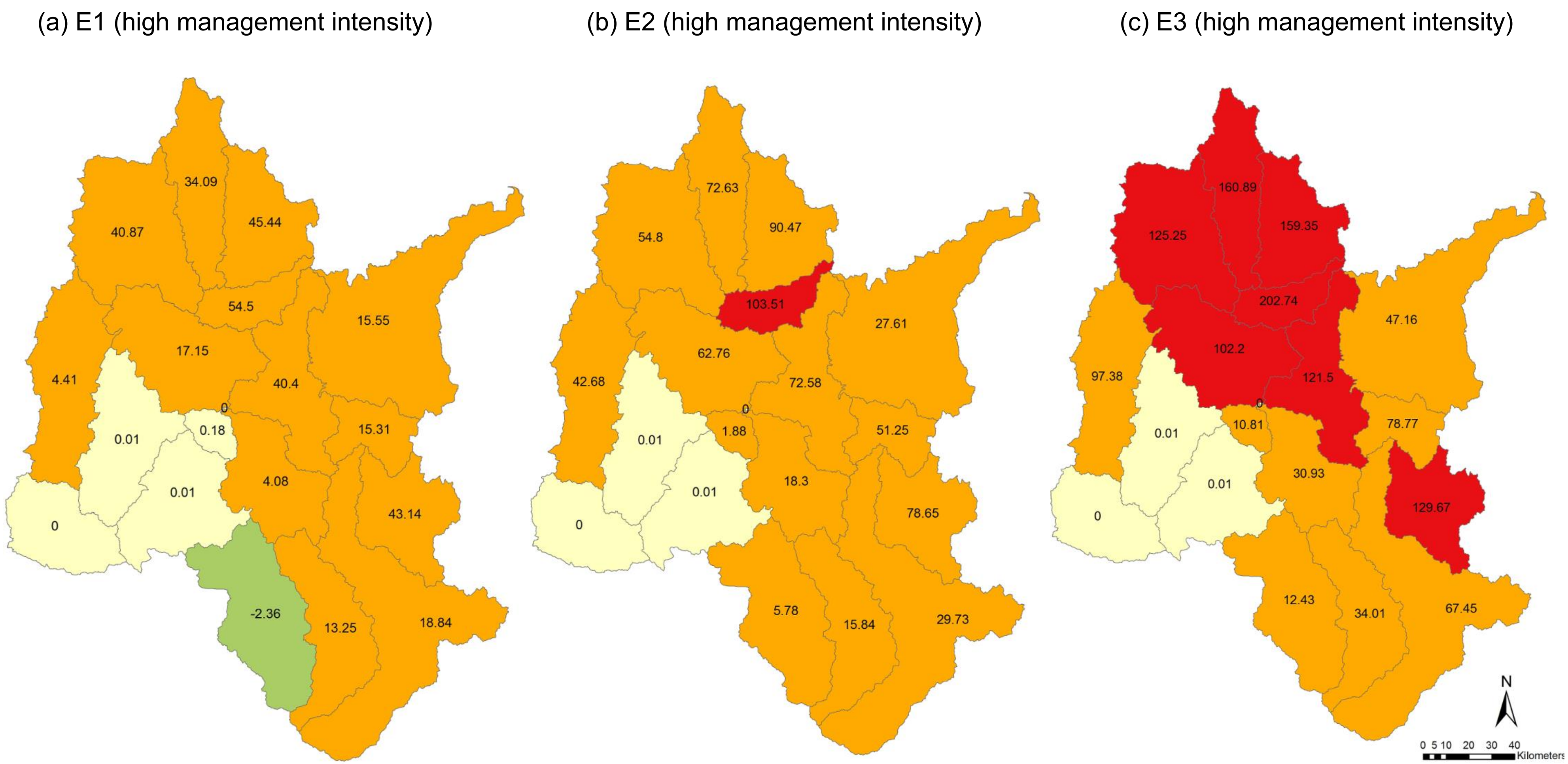


Figure A6.4 [g] Change in estimated average TP load by sub-catchment (%): Eucalypts (high management intensity).

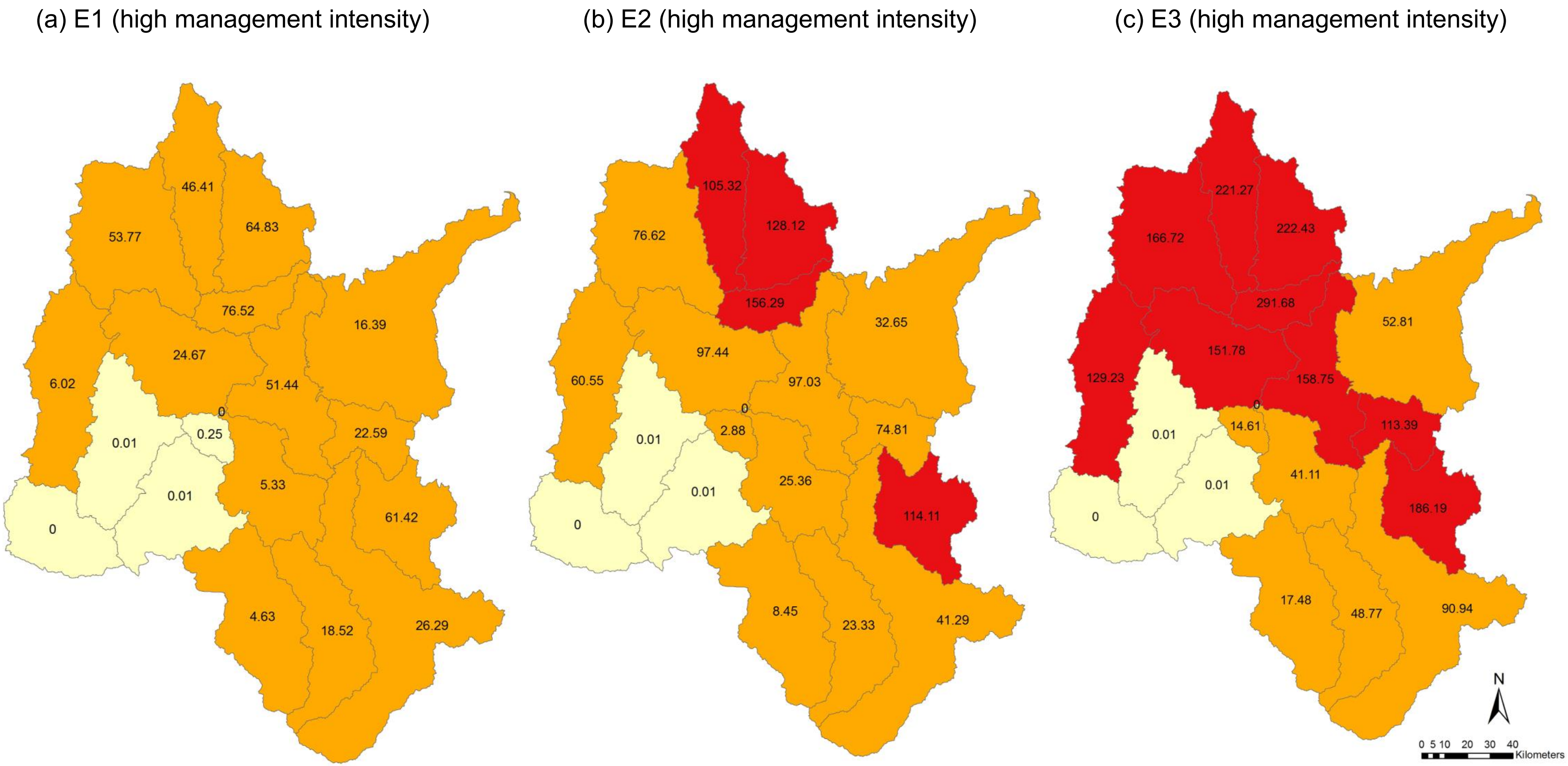


Figure A6.4 [h] Change in estimated average TN load by sub-catchment (%): Eucalypts (high management intensity).

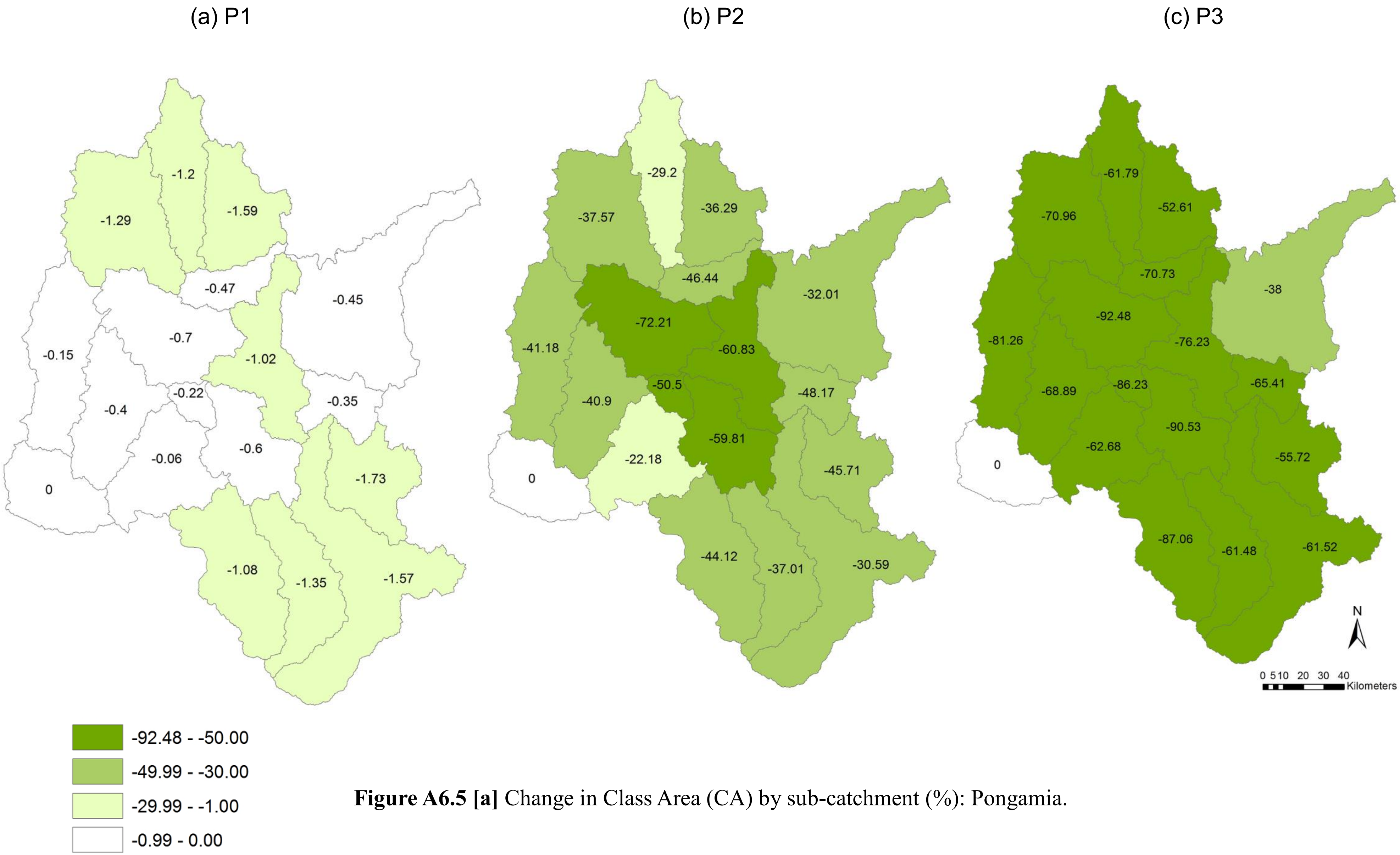


Figure A6.5 [a] Change in Class Area (CA) by sub-catchment (%): Pongamia.

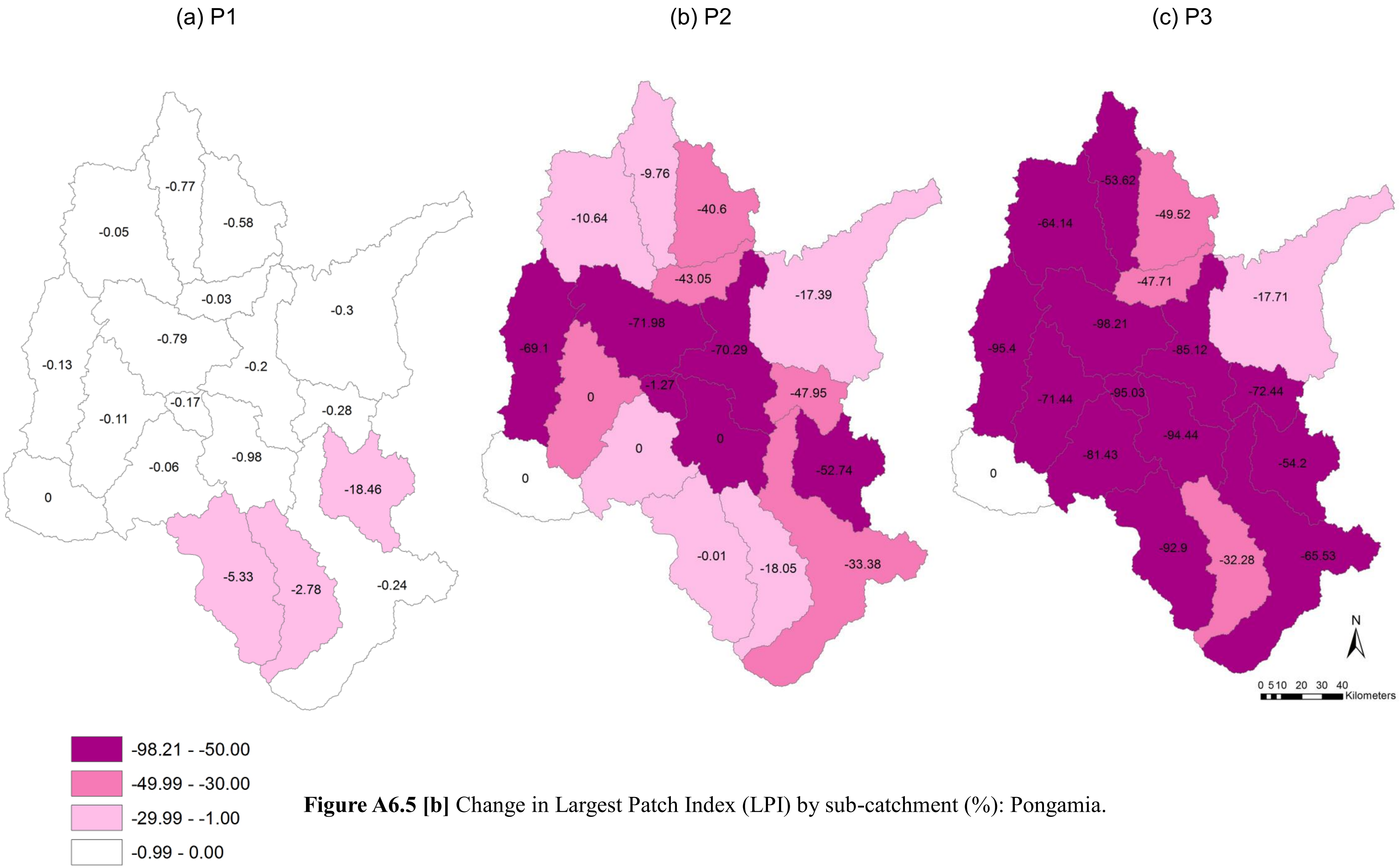


Figure A6.5 [b] Change in Largest Patch Index (LPI) by sub-catchment (%): Pongamia.

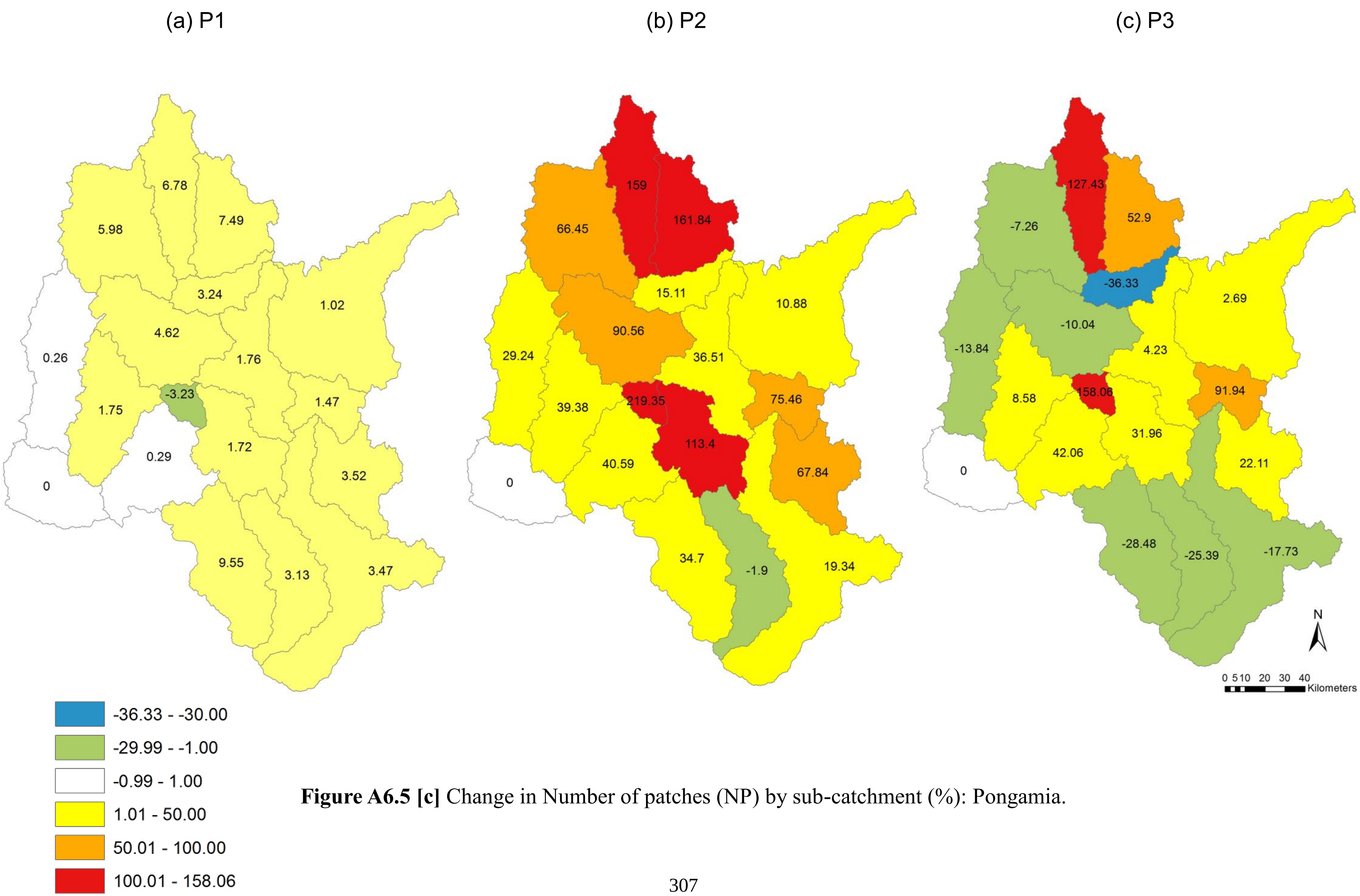


Figure A6.5 [c] Change in Number of patches (NP) by sub-catchment (%): Pongamia.

(a) P1 (low management intensity) (b) P2 (low management intensity) (c) P3 (low management intensity)

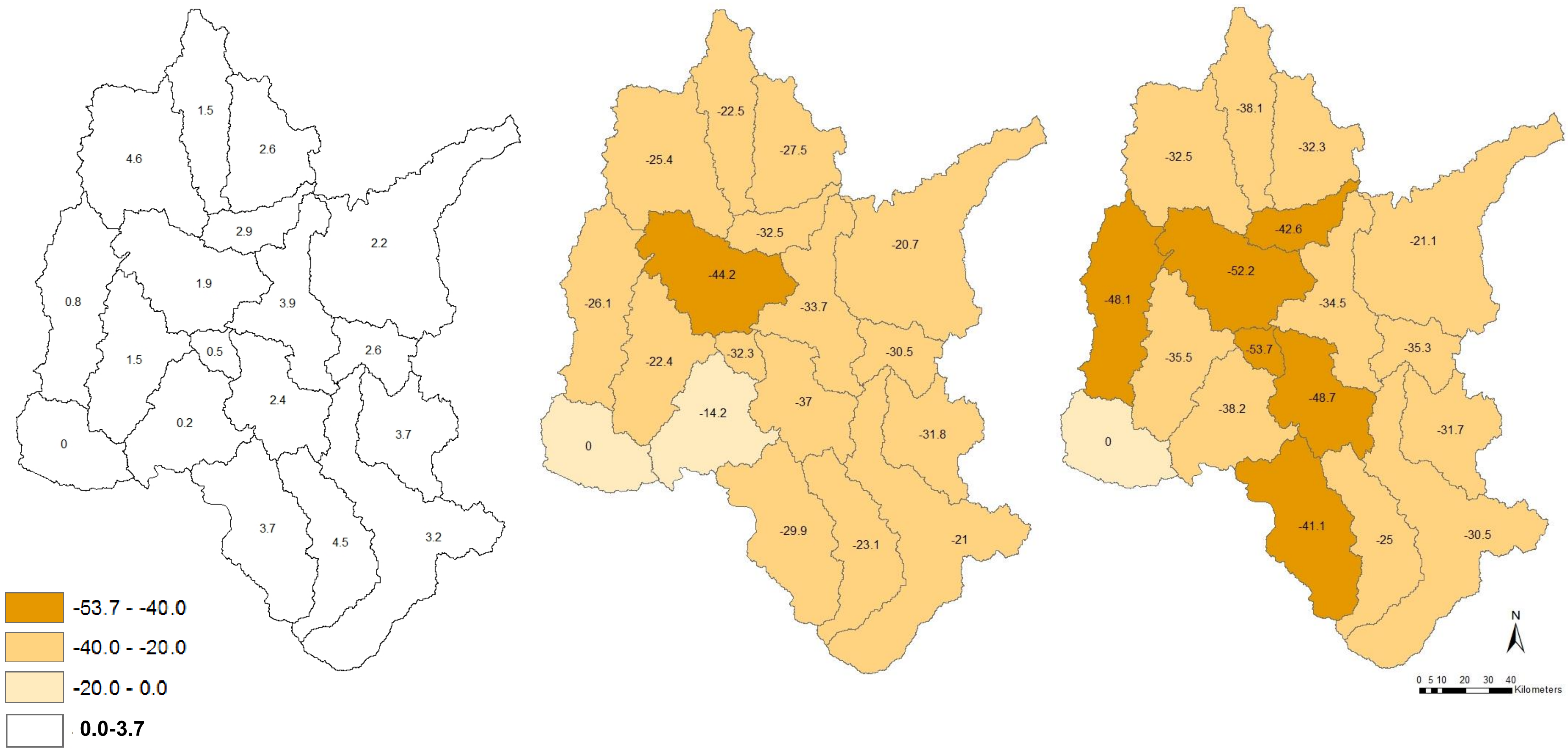


Figure A6.5 [d] Change in actual habitat amount by sub-catchment (%): Pongamia (low management intensity).

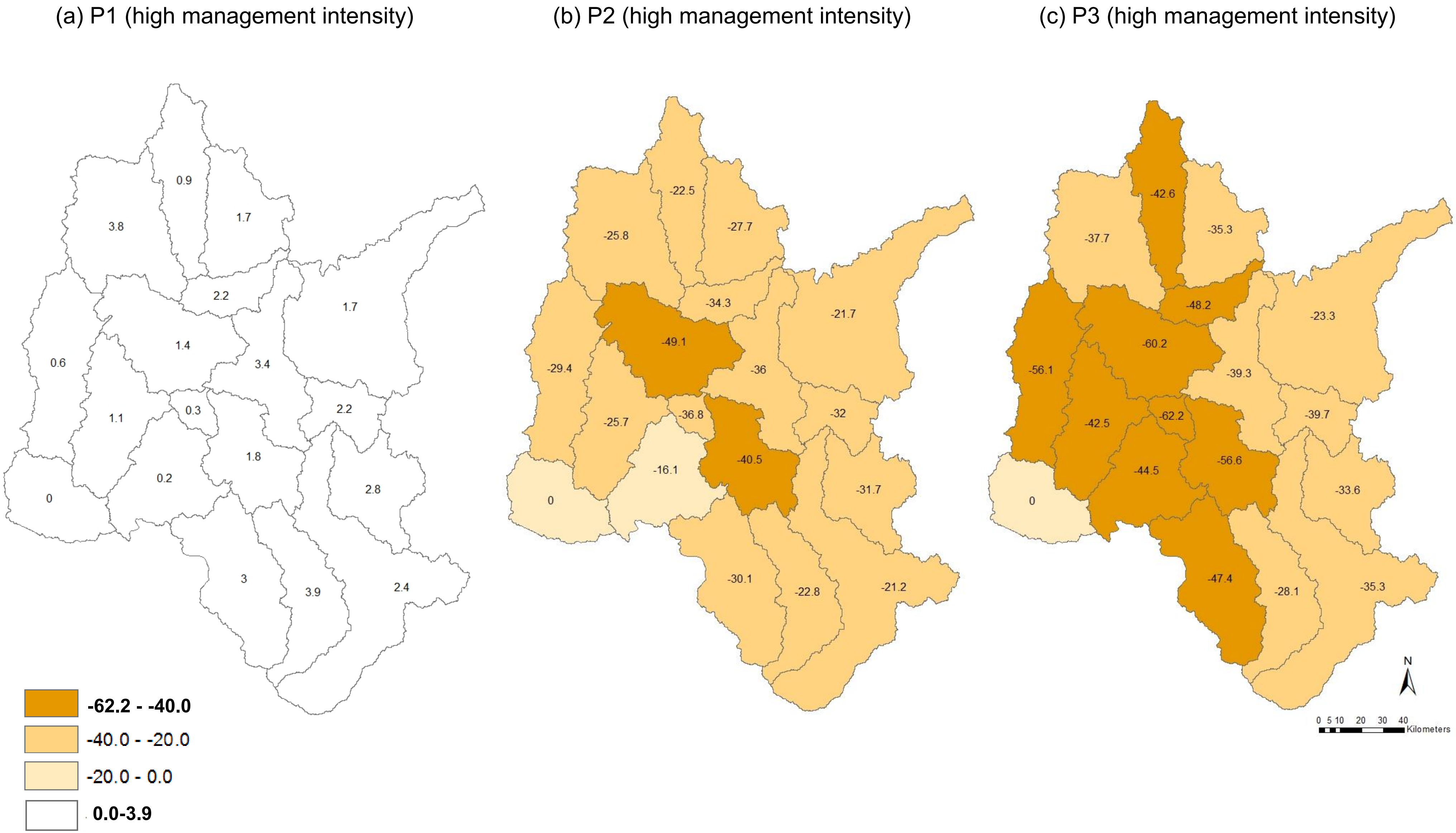


Figure A6.5 [e] Change in actual habitat amount by sub-catchment (%): Pongamia (high management intensity).

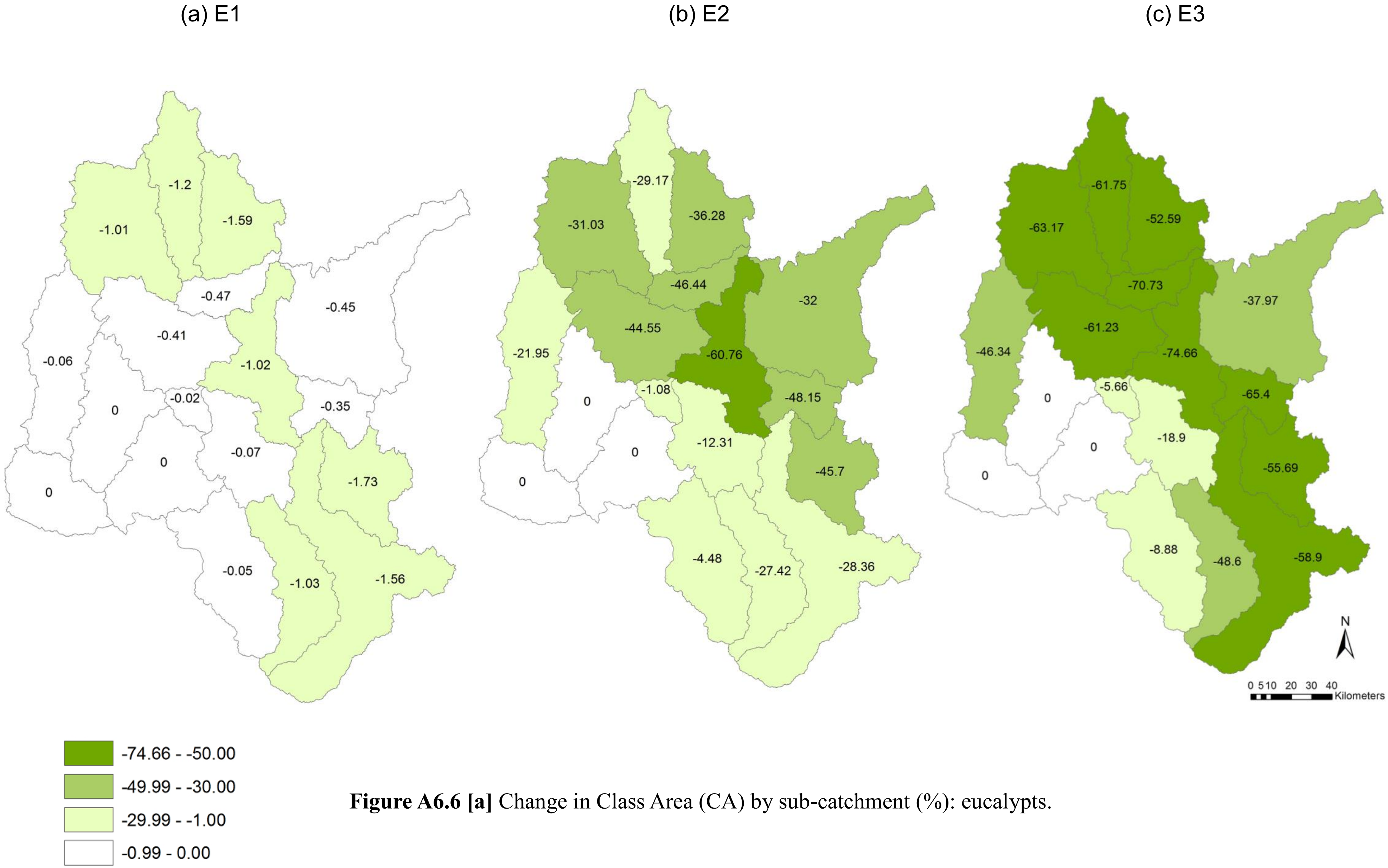


Figure A6.6 [a] Change in Class Area (CA) by sub-catchment (%): eucalypts.

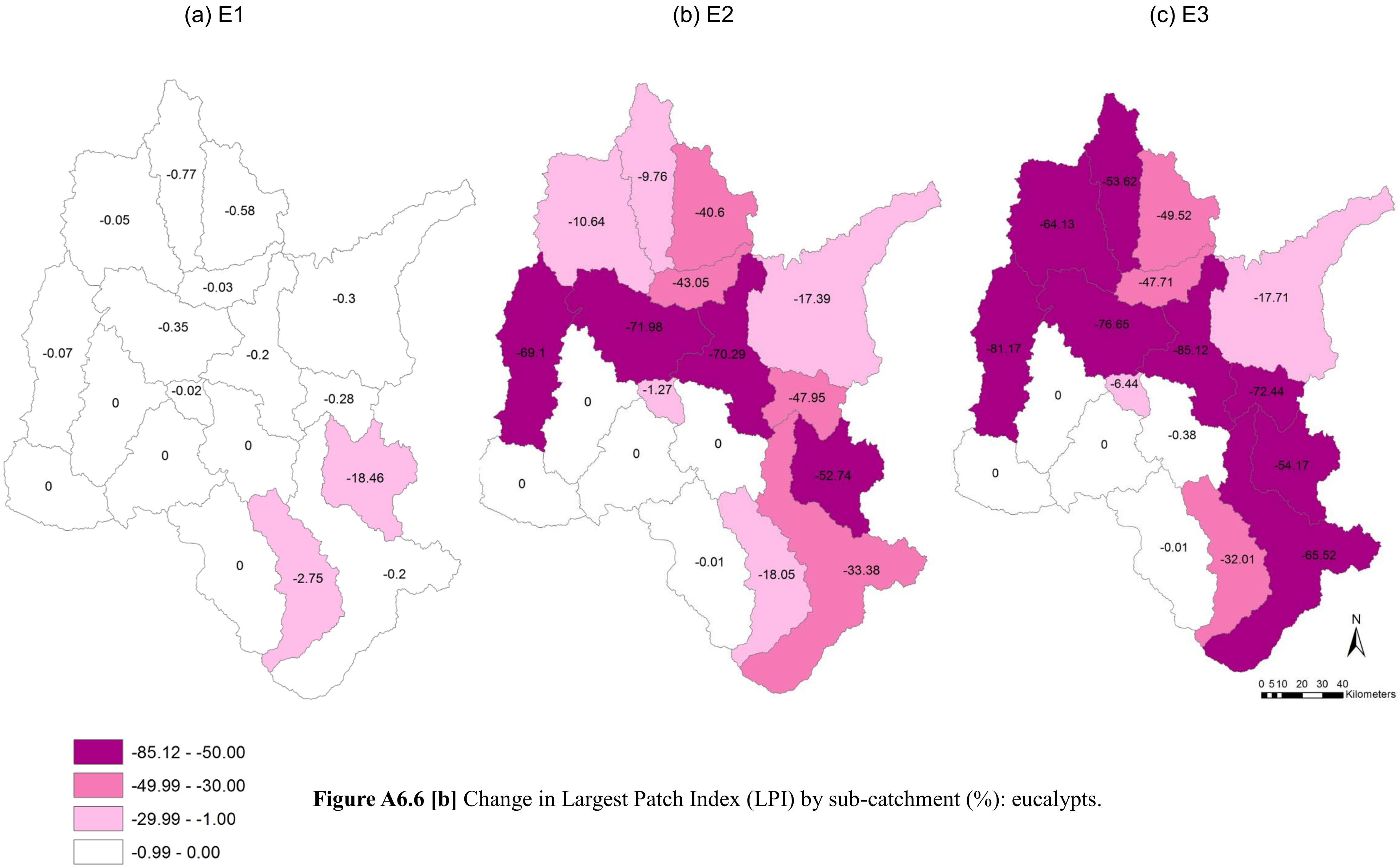


Figure A6.6 [b] Change in Largest Patch Index (LPI) by sub-catchment (%): eucalypts.

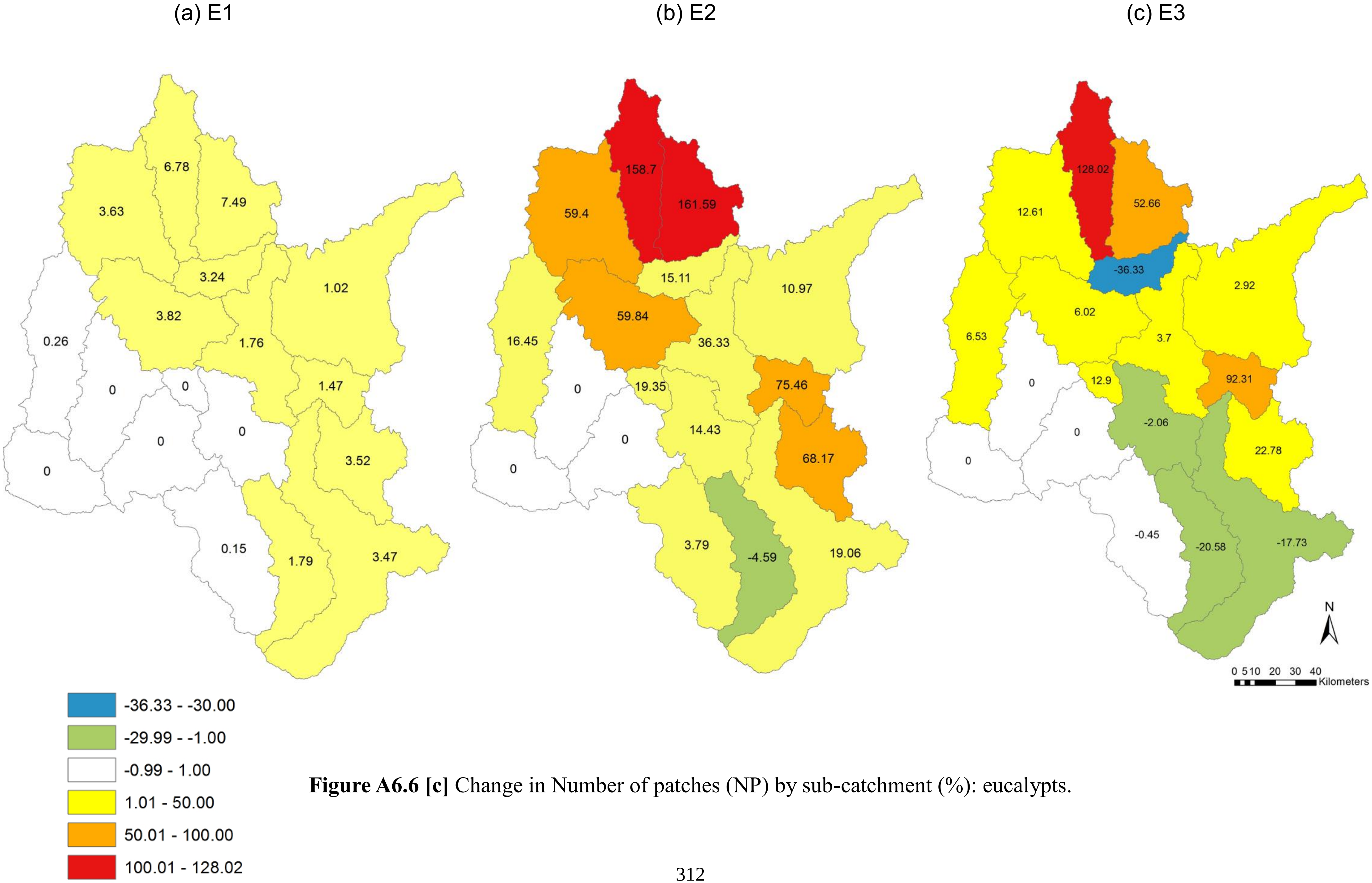


Figure A6.6 [c] Change in Number of patches (NP) by sub-catchment (%): eucalypts.

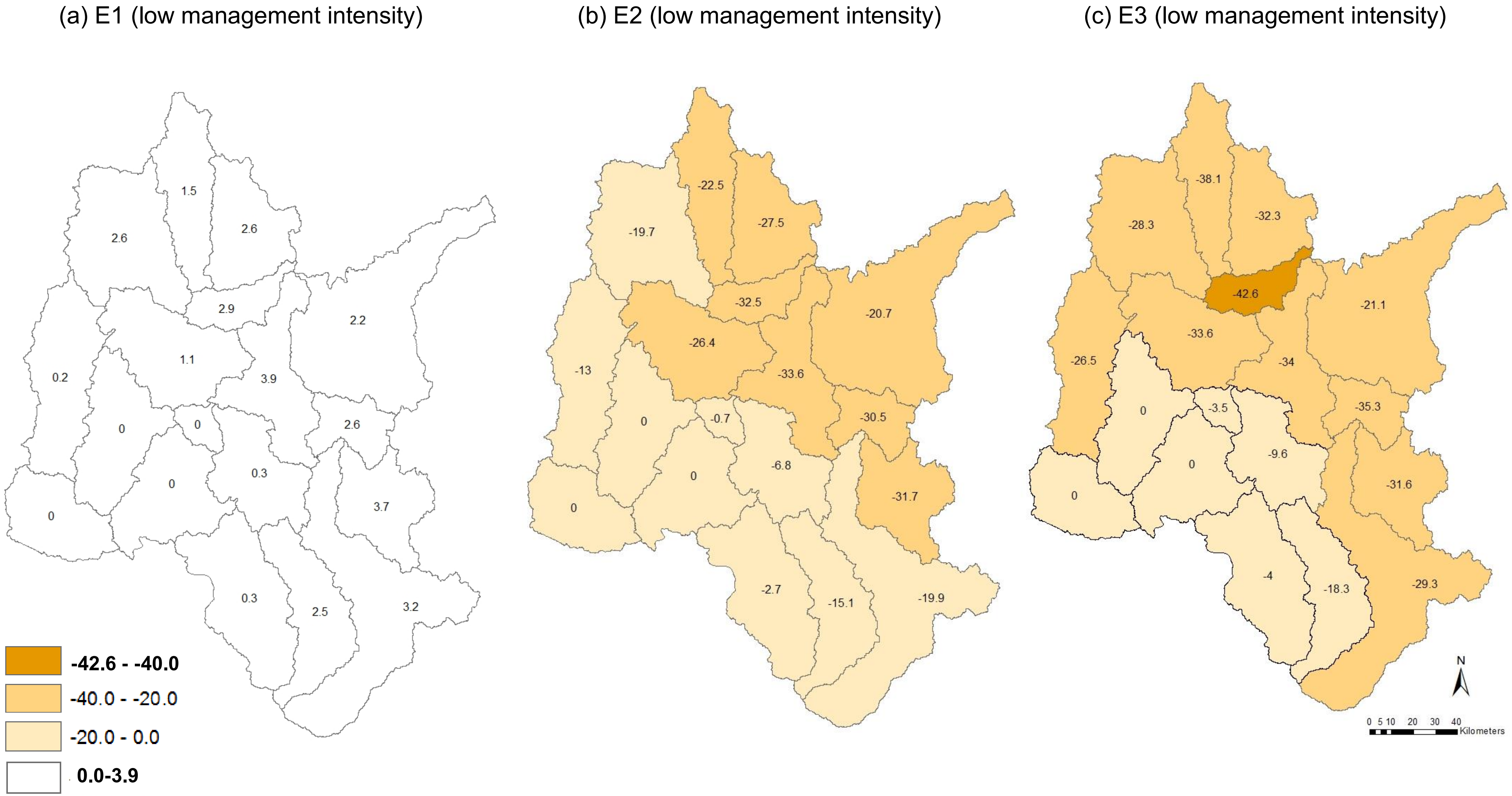


Figure A6.6 [d] Change in actual habitat amount by sub-catchment (%) (low management intensity): eucalypts.

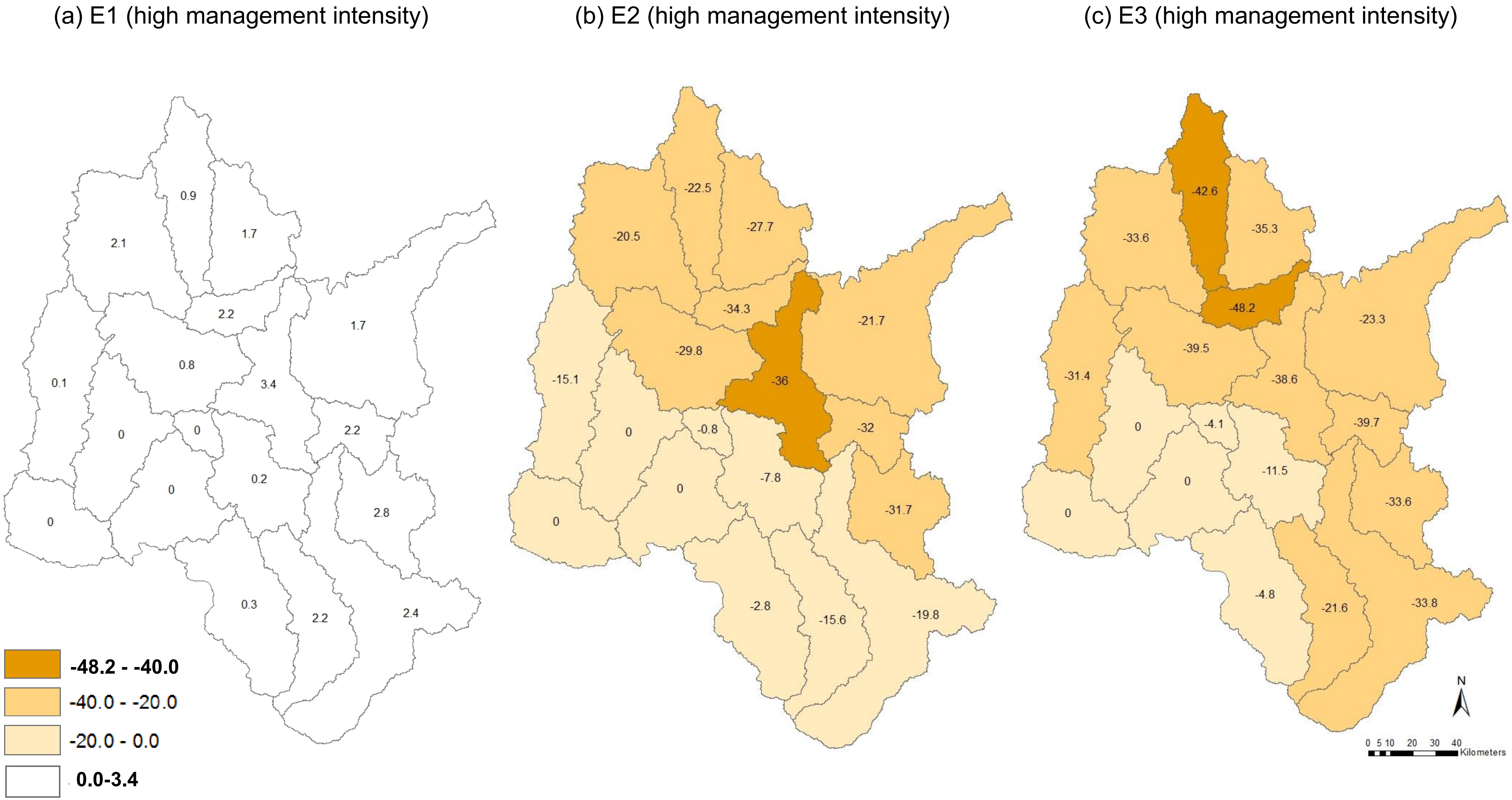


Figure A6.6 [e] Change in actual habitat amount by sub-catchment (%) (high management intensity): eucalypts.